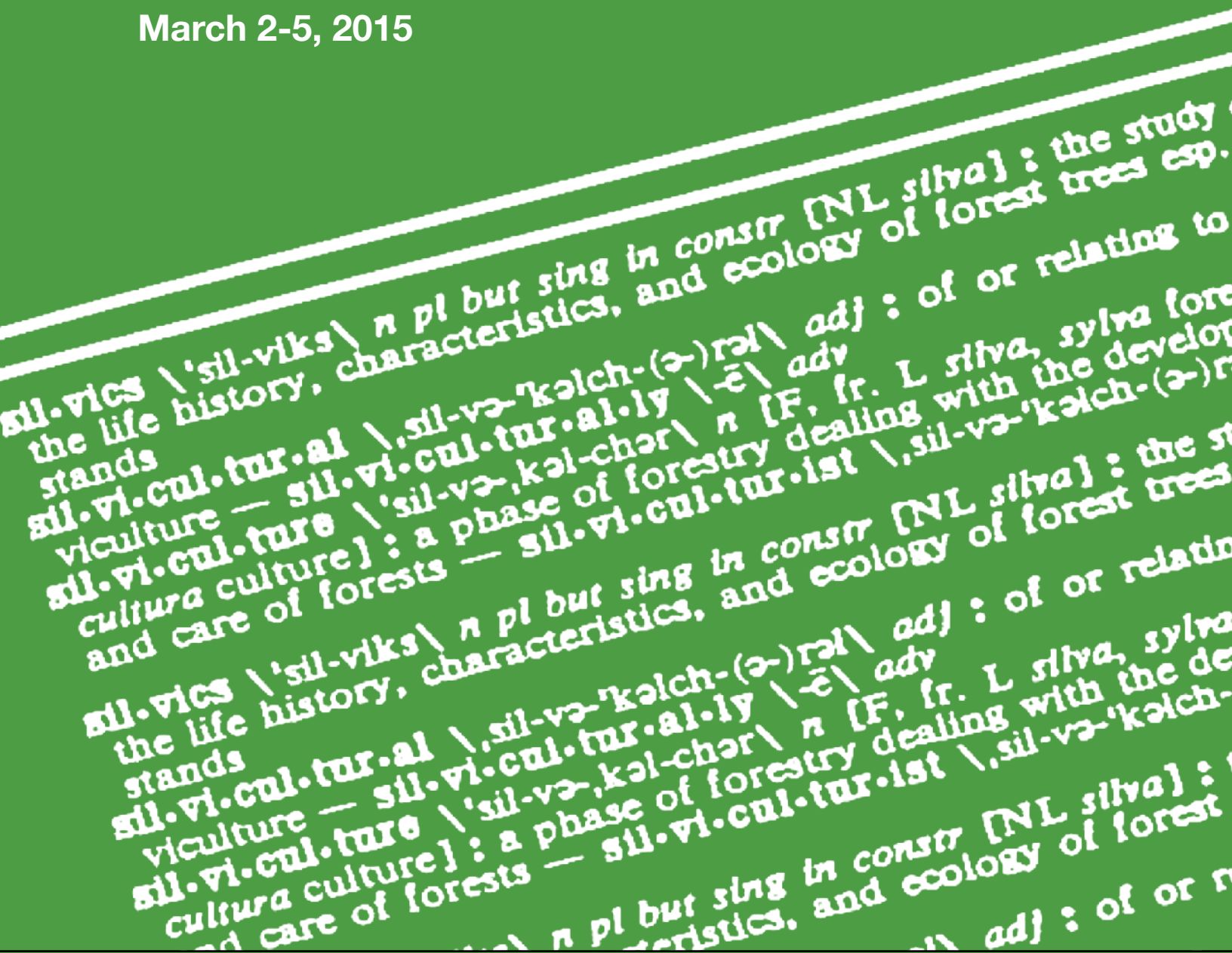
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Proceedings of the 18th Biennial Southern Silvicultural Research Conference

Knoxville, TN

March 2-5, 2015



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EDITORS' NOTE

Papers published in these proceedings were submitted by authors in electronic media. Some editing was done to ensure a consistent format. Authors are responsible for content and accuracy of their individual papers and the quality of illustrative materials.

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Edited by

Callie Jo Schweitzer, Wayne K. Clatterbuck, and Christopher M. Oswalt

Hosted by

Forestry, Wildlife & Fisheries Department, University of Tennessee

USDA Forest Service, Southern Research Station

Preface

The 18th Biennial Southern Silvicultural Research Conference was held March 2-5, 2015, at the Hilton Hotel, Knoxville, TN. This conference provided a forum for silviculturists, researchers, and practitioners to exchange the latest research information. Subjects of concurrent sections included soil/site relationships, forest threats, forest nutrition, forest conservation, pine regeneration, role of fire, shortleaf pine silviculture, longleaf pine silviculture, biometrics, PineMap, hardwood regeneration, hardwood management, biomass, and growth and development. The sessions included 2 invited keynote speakers, 101 oral presentations, and 61 poster presentations. The field trips highlighted mixed species management and research at the Oak Ridge Forest Resources Research and Education Center and the University of Tennessee's Cumberland Forest.

Acknowledgments

That this conference has been ongoing for 36 years is a testament to all those passionate about silviculture in the South. The collaboration among the nine universities and the Forest Service, U.S. Department of Agriculture Southern Research Station to ensure the success of the 18th Conference is commendable. We gratefully acknowledge the University of Tennessee, Forestry, Wildlife & Fisheries Department faculty, students, and staff, for handling local arrangements and logistics. A special thanks to the University of Tennessee's Student Chapter of the Society of American Foresters for all the on-site work they performed. We also offer recognition to Gordon Holley for Web site administration, to Chris Oswalt for poster presentation organization, and to Nancy Bastin and Ryan Sisk for publication communication and formatting assistance. As always, those who volunteered their time to moderate sessions, assist with student poster and presentation judging, and guide field tours have our sincere thanks.

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University of Tennessee Cumberland Forest: Martin Schubert (Forest manager), Wayne Clatterbuck

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March 3, 2015

Session 1:

Soil and Site Relationships, Kimberly Bohn, University of Florida

Pine Regeneration, Tom Fox, Virginia Tech University

Session 2:

Threats, Sonja Oswalt, USDA Forest Service, Southern Research Station

Threats, Callie Schweitzer, USDA Forest Service, Southern Research Station

Session 3:

Conservation, Joshua Adams, Louisiana Tech University

Session 4:

Nutrition, Jeremy Stovall, Stephen F. Austin State University

Fire, Brian Oswald, Stephen F. Austin State University

March 4, 2015

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Biometrics, Andrew Nelson, University of Arkansas at Monticello

Growth and Development, Gordon Holley, Louisiana Tech University

Session 2:

Longleaf 1, Geoff Wang, Clemson University

PineMap, Zhaofei Fan, Auburn University

PineMap, Andy Ezell, Mississippi State University

Longleaf 2, Ben Knapp, University of Missouri

Session 3:

Hardwood Management, Chris Oswalt, USDA Forest Service, Southern Research Station

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Kimberly Bohn
University of Florida

STATUS OF STATE FORESTRY BEST MANAGEMENT PRACTICES FOR THE SOUTHEASTERN UNITED STATES

R. Cristan, W.M. Aust, M.C. Bolding, S.M. Barrett, and J.F. Munsell[1]

Abstract—Forestry Best Management Practices (BMPs) are important measures for protecting the waters of the U.S., but few studies have compared monitoring strategies and implementation success of forestry BMPs across states. In order to assess the status of state forestry BMPs, a survey was sent to the state forestry agency in each U.S. state regarding their forestry BMP program. The survey included questions pertaining to agency involvement in developing BMP guidelines, rates of BMP implementation, monitoring methods, and the nature of state BMP guidelines (whether non-regulatory, quasi-regulatory, or regulatory). Surveys were completed by all 50 states and results allowed evaluation of the status and implementation of forestry BMPs by state and region. This paper will focus on survey responses from the thirteen southern states represented by the Southern Group of State Foresters. All thirteen southeastern states have conducted BMP monitoring and have future monitoring of BMPs planned. Eleven states have conducted or are currently conducting BMP effectiveness studies. All the southeastern states have conducted BMP implementation studies and the mean implementation rate is 92 percent which is above the mean national implementation rate of 91 percent. Seven states have non-regulatory BMP guidelines, five states have quasi-regulatory guidelines, and one state has regulatory guidelines. This study indicated that some states reported BMP deficiencies for some individual BMP categories, yet these states' average BMP implementation levels appear to be satisfactory.

INTRODUCTION

Forestry best management practices (BMPs) were formulated from the passage of the Federal Water Pollution Control Act (FWPCA) of 1972 and have evolved overtime as the states develop new BMP guidelines and revising existing guidelines (Edwards and Stuart 2002, Archey 2004, Phillips and Blinn 2004). The EPA required states to develop either regulatory or non-regulatory BMP programs in 1977 (Ice and others 2004). Section 319 of the reauthorization of the FWPCA of 1987 required states to report to the U.S. Congress on the status and impacts of non-point source pollution (Novotny 2003).

Forestry BMP regulations in the southeastern U.S. are generally non-regulatory; however, some states have quasi-regulatory BMPs (Aust and Blinn 2004). Quasi-regulatory BMPs normally consist of non-regulatory BMP guidelines, but fines may be associated with water quality issues from forest operations. Forestry BMP monitoring and implementation may vary by region or state. However, in the southeastern U.S., the Southern Group of State Foresters (SGSF) developed an approach for monitoring and implementation (Ice and others 2010). The SGSFs main goal for their forestry BMP approach in the south is to provide leadership and support to state forestry agencies to protect water quality (Southern Group of State Foresters 2012).

There are few studies evaluating forestry BMP implementation on a national or regional level. The SGSF publishes reports on BMP implementation of their 13 states with their last report in 2012. The overall BMP implementation rate was 87 percent in 2008 and 92 percent in 2012 (Southern Group of State Foresters 2012). The National Association of State Foresters (NASF) also conducts BMP surveys that evaluate the status of state developed and implemented forestry BMPs on a national scale. The NASF conducted five BMP surveys from 1992 to 2004 (Edwards and Stuart 2002, Archey 2004). Their surveys evaluated implementation rates, monitoring, policies, agency involvement, regulations, legislation, effectiveness studies, and other topics. The 2004 report conducted by Archey (2004) found that the mean implementation rate for twenty-seven states that reported data was 91 percent. The objective of this project is to redo the 2004 NASF survey to get an updated perspective on the status of state forestry BMPs nationwide. The survey results from the thirteen southeastern states are reported in this paper.

MATERIALS AND METHODS

An online survey was used to obtain the status and implementation of state forestry BMPs in the United States. The initial survey was developed in March 2012 by the National Association of State Foresters (NASF)

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and was based on a previous survey conducted by the NASF in 2004 (Archey 2004). The developed survey was altered so that it could be converted to an online survey using Survey Monkey in February 2013. The survey had to be approved by the Virginia Tech Institutional Review Board (VT IRB) before sending out the survey. The VT IRB also had to approve pre-notice and follow-up e-mails. The survey was approved in March 2013 and pre-notice letters were e-mailed on April 15th, 2013 to the lead state forester in each state. The pre-notice letters described the upcoming survey and included directions to pass the survey to the appropriate forest water quality personnel in their state. The survey was then e-mailed to the lead state forester on April 17th, 2013. The survey was closed out in December of 2013. Responses to the survey were downloaded into Microsoft Excel and Microsoft Access for formatting and presenting the results.

RESULTS AND DISCUSSION

Forestry BMP regulation in the southeastern states consisted of seven non-regulatory states, five quasi-regulatory states, and one regulatory state (table 1). All 13 states reported that they have BMP manuals and that they conduct forestry BMP implementation studies. Most recent BMP implementation study years ranged from 2010 to 2012 and the overall state BMP implementation rates ranged from 84 percent to 99 percent (table 1). Although state BMP implementation rates were high, some states reported lower (below state BMP implementation rate range) BMP implementation rates for individual BMP categories such as skid trails, stream crossings, wetlands, mechanical site preparation, and prescribed burning (table 2). Average implementation rates for forest roads, skid trails, stream crossings, and prescribed burning were all below the overall southeastern average of 92 percent. These categories of BMPs are of particular concern because they are forest operations that have been found to have relatively high potential erosion rates (roads, skid trails, site preparation, and firelines) or because of their close proximity to bodies of water (stream crossings, wetlands). Although the BMP implementation scores are higher than in previous years, these finding indicate the continuous need for attention to BMP implementation, particularly are these critical areas.

State forestry agencies are the lead agency in monitoring forest operations for BMP implementation in the southeastern U.S. (table 3). States reported that most recent BMP monitoring year ranged from 2008 to 2013 and the next planned monitoring year range was from 2013 to 2015. All of the southeastern states monitor sites post-forest operation; however, some states also reported that they monitor sites pre-forest operation, and during-forest operations. The post operation visits have the advantage of allowing the inspection personnel to monitor the sites after BMPs have been implemented. However, if the inspector detects some issue that needs modification it becomes more difficult to task logging contractors with return site

Table 1—Southeastern survey results for forestry BMP regulation, BMP manual year, implementation rate (%), and implementation year

State	State regulation	BMP manual	Implementation rate (%)	Implementation year
Alabama	Quasi-regulatory	2007	97	2010
Arkansas	Non-regulatory	2002	87	2011
Florida	Quasi-regulatory	2008	99	2011
Georgia	Non-regulatory	2009	97	2011
Kentucky	Regulatory	2008	94	2012
Louisiana	Non-regulatory	2000	96	2012
Mississippi	Non-regulatory	2008	91	2010
North Carolina	Quasi-regulatory	2006	85	2011
Oklahoma	Non-regulatory	1991	95	2010
South Carolina	Quasi-regulatory	2012	91	2012
Tennessee	Non-regulatory	2003	84	2010
Texas	Non-regulatory	2010	95	2011
Virginia	Quasi-regulatory	2011	90	2012

Table 2—Forestry BMP implementation rate results by individual BMP categories. Minimum, maximum, average, and number of states that reported data for that specific BMP category

BMP category	Minimum (%)	Maximum (%)	Average (%)	Number of states
Timber harvest	88	99	95.0	8
Forest roads	84	99	91.3	13
Skid trails	75	100	89.7	10
Log landings	92	100	95.8	9
Stream crossings	72	98	89.2	13
SMZs	86	98	93.2	13
Wetlands	70	100	94.1	9
Reforestation	95	100	97.6	7
Mechanical site preparation	74	99	91.6	9
Chemical site preparation	93	100	98.6	8
Pesticide	98	100	99.6	5
Fertilizer	100	100	100.0	2
Prescribed burning	60	100	87.4	8
Wildfire suppression	100	100	100.0	2
Wildfire rehabilitation	100	100	100.0	1
Public lands	94	100	97.8	5

Table 3—States that reported when they monitor BMPs, next planned monitoring, agencies that are involved in monitoring, and phase of forest operations that sites are monitored

State	Most recent year	Next planned year	Agencies involved	When sites monitored
Alabama	2012	2014	Forestry	PR, D, PO
Arkansas	2011	2015	Forestry	PO
Florida	2011	2013	Forestry	D, PO
Georgia	2013	2013	Forestry	D, PO
Kentucky	2012	2013	Forestry	D, PO
Louisiana	2012	2015	Forestry	PO
Mississippi	2010	2014	Forestry	PO
North Carolina	2008	2014	Forestry	D
Oklahoma	2010	2014	Forestry	PO
South Carolina	2012	2015	Forestry	PO
Tennessee	2010	2015	Forestry	PO
Texas	2011	2014	Forestry	PO
Virginia	2012	2013	Forestry	PO

PR, pre-forest operation; D, during-forest operation; PO, post-forest operation

visits, therefore there are some advantages to both pre and post closure site visits.

Eleven of the southeastern states reported that they have conducted BMP effectiveness studies (table 4). Majority of the eleven states completed these effectiveness studies between 2012 and 2013. Five states reported that they have ongoing BMP effectiveness studies and four states have effectiveness studies planned for the future. The effectiveness studies are conducted mostly by the state forestry agencies and academia. These finding indicate that research personnel are clearly involved in BMP research and that it could be advantageous for additional coordination between state forestry personnel, academics, and loggers. BMP workshops have been used commonly for logger and forester continuing education programs and we recommend that state forestry agencies continue to involve academic researchers in these programs.

CONCLUSION

The mean BMP implementation rate is 92 percent for the southeastern states and the mean national implementation rate is 91 percent. The implementation rates for the southeastern states are similar to what the SGSFs reported in their 2012 report which was also 92 percent (Southern Group of State Foresters 2012). However, the national and southeastern survey results indicate that there may be potential deficiencies in some individual BMP categories for some state BMP programs. Mean state implementation rate is an average over all the individual guidelines evaluated by each state and potential deficiencies are not noticeable until the implementation results are broken down by individual BMP categories. Some of the individual BMP categories with low BMP implementation rates would be forest roads, skid trails, wetlands, stream crossings, mechanical site preparation, and prescribed burning. Overall, state forestry BMP programs in the southeastern U.S. are effectively protecting forest water quality by having high implementation rates, updated BMP guidelines and manuals, monitoring BMPs every couple of years, and conducting BMP effectiveness studies.

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LITERATURE CITED

Archey, W. 2004. The National Association of State Foresters: 2004 progress report—state water resources programs for silviculture. Washington, DC: The National Association of State Foresters. Available online at www.stateforesters.org/node/811. [Date accessed: December 9, 2009].

Aust, W.M.; Blinn, C.R. 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: an overview of water quality and productivity research during the past 20 years (1982–2002). Water, Air and Soil Pollution: Focus. 4(1): 5-36.

Table 4—States that reported that they have conducted BMP effectiveness studies in the past, the year of the most recent completed effectiveness study, if the state has any current effectiveness studies, the next planned effectiveness study, and agencies that are or have conducted effectiveness studies

State	Previous study	Most recent study	Current study	Next planned study	Agency
Alabama	Yes	2012	No	NR	NR
Arkansas	Yes	2006	No	NR	A
Florida	Yes	2013	Yes	2013	F,A
Georgia	Yes	2012	Yes	NR	A, NGO
Kentucky	Yes	2013	No	NR	A,O
Mississippi	Yes	2013	No	2014	F
North Carolina	Yes	2012	Yes	2013	F
Oklahoma	Yes	2012	Yes	NR	A
South Carolina	Yes	NR	No	NR	NR
Texas	Yes	2007	No	2015	F
Virginia	Yes	2012	Yes	NR	F, A

NR, none reported; A, academia; F, forestry; E, environmental; O, other; NGO, non-government organization

Edwards, P.J.; Stuart, G.W. 2002. State survey of silviculture nonpoint source programs: A comparison of the 2000 northeastern and national results. Northern Journal of Applied Forestry. 19(3): 122-127.

Ice, G.; Dent L.; Robben J [and others]. 2004. Programs assessing implementation and effectiveness of state forest practice rules and BMPs in the West. Water, Air and Soil Pollution: Focus. 4(1): 143-169.

Ice, G.; Schilling, E.; Vowell, J. 2010. Trends for forestry best management practices implementation. Journal of Forestry. 108(6): 267-273.

Novotny, V. 2003. Water quality: Diffuse pollution and watershed management. New York: John Wiley & Sons. 864 p.

Phillips, M.J.; Blinn, C.R. 2004. Best management practices compliance monitoring approaches for forestry in the eastern United States. Water, Air and Soil Pollution: Focus. 4(1): 263-274.

Southern Group of State Foresters. 2012. Implementation of forestry best management practices: 2012 Southern region report. Southern Group of State Foresters, Water Resources Committee. 18 p. Available online at: www.southerforests.org/publications/Regional%20BMP%20report%202008.pdf/view. [Date accessed: unknown].

EVALUATION OF EROSION CONTROL BMPS ON DITCHED HAUL ROAD STREAM CROSSING APPROACHES FOLLOWING RECONSTRUCTION

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Abstract—Ditched forest roads leading to stream crossings and used for log transportation have recently been a topic of water quality concern and legal controversy. Best management practices (BMPs) can reduce potential water quality issues, yet few research studies have quantified BMP costs and reductions in sediment from implementing specific ditch BMPs. Researchers reconstructed fifty 15.2 m sections of ditches at stream crossings or cross drains and applied one of five treatments. Treatments were replicated 10 times and split by two slope classes in a completely randomized design. Silt fence traps were placed at the end of each ditch section prior to stream crossings or culvert inlets. Sediment pins were installed adjacent to the silt fence and sediment deposit depths were measured 42 days following ditch re-construction. Field based results show that median erosion rates were greatest for the Bare (48.0 Mg ha^{-1} yr^{-1}), followed by Seeded (25.0 Mg ha^{-1} yr^{-1}), Check Dams (22.2 Mg ha^{-1} yr^{-1}), Completely Rocked (2.6 Mg ha^{-1} yr^{-1}), and Seeded with Erosion Mat (2.1 Mg ha^{-1} yr^{-1}). Results from the Kruskal-Wallis statistical test showed that the Bare treatments had significantly greater erosion rates from Completely Rocked and Seeded with Erosion Mat treatments. Cost of BMP treatment was cheapest for Seeded ($9), followed by Seeded with Erosion Mat ($24), Check Dams ($45.40), and Completely Rocked ($119.50). Study results suggest that re-constructed ditches should drain short sections of road and should contain some erosion control BMP in order to reduce erosion and sediment delivery.

INTRODUCTION

Forest roads are an essential, yet expensive component of forest management. They are inherently prone to surface runoff and erosion issues as they often lack surface roughness, expose compacted bare mineral soils, and concentrate overland flow via ditches or rutted road surfaces (Grace 2005). Runoff from road and ditch networks can degrade water quality if not properly dispersed and controlled (Brown and others 2013). High sediment levels have been shown to be detrimental to many aquatic wildlife species (Henley and others 2000; Gibson and others 2005) and can lead to increased water treatment costs (Bridges and others 2010). Road and ditch networks sloping towards stream crossings (approaches) pose one of the greatest risks of sediment delivery from forest operations because of their juxtaposition to waterways (Aust and Blinn 2004). These areas are especially vulnerable to erosion immediately following construction or maintenance (Croke and others 2001). Subsequently, forestry best management practices (BMPs) have been developed for these areas to comply with the Federal Water Pollution Control Act of 1972 (Clean Water Act). Forest managers are continually challenged to implement cost-effective BMPs to ensure maximum financial benefits and minimize environment degradation. Therefore research that quantifies cost-efficacy of BMPs is highly desirable, especially for sensitive areas such as stream crossing approaches.

Many ditched forest road stream crossings are exempt from point source permitting under the Clean Water Act (Section 404) when they are considered normal ongoing silviculture operations that follow federal and state BMPs, do not alter hydrology, and do not introduce toxins into water sources. Environmental concerns regarding this exemption for ditched forest roads have sparked numerous debates and lawsuits (Boston and Thompson 2009). Of the most notable was the Ninth Circuit Court and later US Supreme Court ruling initiated by *National Environmental Defense Center v. Brown*. The Ninth Circuit Court ruled that runoff collected by ditches and conveyed into the nations waters constituted as an industrial point source pollutant and required a National Pollution Discharge Elimination System (NPDES) permit (Boston 2012). In response to the ruling, the Environmental Protection Agency (EPA) elicited and reviewed comments from professionals and the general public regarding the exemption (EPA 2012). The US Supreme Court later reversed the Ninth Circuit Court ruling in March of 2013 retaining that the EPA was given authority through

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legislation to determine what activities may or may not be exempt from NPDES permitting. The EPA announced that they would not change how the exemption would be enforced in July of 2014 (EPA 2014), yet further litigation is likely to ensue (MacCurdy and Timmons 2013). This highly controversial matter only further highlights the need for research to quantify forest road and ditch sediment rates. The objective of this paper was to quantify erosion rates following re-construction of ditches leading to stream crossings implementing four operational ditch BMP treatments. We also discuss the comparison of erosion rates to associated treatment costs.

MATERIALS AND METHODS

Study Sites

Three primarily insloped forest roads that would serve as class II haul roads for logging (Walbridge, 1990) were selected for this study. All road segments are located near Blacksburg, Virginia within the Ridge and Valley physiographic region. Blacksburg receives a mean annual rainfall total of 1041 mm, which is commonly well distributed throughout the year (NOAA 2015). Two of the road segments are managed by Virginia Tech and the third road segment by US Forest Service. Mean road grades are below 12 percent, with maximum slopes of 16 percent. All road sections are gated and restrict public vehicle use. However, the US Forest Service road provides firewood cutting permits, which allow occasional light truck use. Virginia Tech school forest roads serve access to a municipal water tower and cell phone tower. These roads also received light vehicle use during this study.

Study Design and Treatments

Fifty ditched stream crossing approaches from the three road segments were selected in June 2014 for ditch re-construction. A John Deere 450 bulldozer was used to remove soils and organic debris away from streams (upslope) within the ditch to improve ditch function, road drainage, and trafficability. Ditches were reconstructed according to site specific situations, but all were cleared to a minimum of approximately 15.2 m past each stream crossing. Where applicable, a rolling dip was installed at the end of treatments to minimize drainage area. A New Holland TN650 Farm tractor was used to smooth the road to ditch transition using a rhino blade. Road surfaces were not re-graveled following re-construction. Silt fence catchment areas were placed within each ditch section prior to the stream crossings (Robichaud and Brown 2002). A network of sediment pins were installed adjacent to the silt fence to allow for periodic measurement of sediment deposit depths. Treatments were applied to the nearest 15.2 m to the sediment catchment areas. Seed treatments were applied by hand spreading a Forest Service recommended seed and lime mixture of orchard grass (65 pls lbs ac^{-1}), annual rye (25 pls lbs ac^{-1}), white clover (2 pls lbs ac^{-1}), and pelletized lime. Seed and erosion mat treatments received the same seed mixture and were covered with an erosion mat interlaced with straw. Erosion mats were secured within the ditch by landscape staples. Number 1 surge (8.9 – 10.2 cm) was used for check dam and rock treatments. The number 1 surge was purchased from a local quarry and delivered on site. Rock treatments were applied using the frontend loading bucket of the farm tractor. Two rock check dams (~1.5 m length) were applied at ¼ and ¾ distances from catchment area in check dam treatments. Randomization of treatments was split by ditch slope (table 1). One of five treatments (bare, seeded, seeded and erosion mat, check dams, and completely rocked) was randomly assigned to each ditch section. Each treatment was replicated 10 times for a total of 50 experimental units, but was not balanced within slope classes (table 1). This study was analyzed as a completely randomized design. Data were analyzed for statistical significance using JMP statistical software. Due to unequal variance and non-normally distributed data, the Kruskal-Wallis test was used to detect treatment differences in median sediment delivery rates.

Table 1—Number of randomly assigned BMP treatments by slope class

	Number of Replications by Slope Class		
Treatments	0 – 6.99 percent	7.00 – 16 percent	Total
Bare	5	5	10
Seed	4	6	10
Seed and Erosion Mat	6	4	10
Check Dam	3	7	10
Rock	5	5	10
Total	23	27	50

Measurements

Sediment pin heights were measured 42 days following construction using a standard 1/16th inch increment measuring tape. Bulk densities were collected from each catchment area containing sediment deposits following the core method (Blake and Hartge 1986). Sediment volumes (m^{-3}) were calculated by multiplying depositional area (m^{-2}) by elevation gain (m). Sediment volumes were converted to a sediment load (Mg) by multiplying bulk density (Mg m^{-3}) of the trapped sediment by sediment volumes (m^{-3}) (Brown and others, 2013). Sediment loads were then divided by the drainage area and time (42 day) in order to obtain an annual erosion rate (Mg ha^{-1} yr^{-1}). Daily rainfall data were obtained from Virginia Tech airport weather station (software, VWS V14.00), which is located less than 11 km from all study areas. Ditch and road dimensions were measured using a total station (Sokkia total station model SET-520, Tokyo, Japan). Additional road characteristics measured included: surface roughness (Saleh 1993), soil particle size (Gee and Or 2002), road shape, percent canopy cover, and percent bare soil (Brown and others 2013).

RESULTS AND DISCUSSION

Field based results show that median erosion rates were greatest for the Bare (48.0 Mg ha^{-1} yr^{-1}), followed by Seeded (25.0 Mg ha^{-1} yr^{-1}), Check Dams (22.2 Mg ha^{-1} yr^{-1}), Completely Rocked (2.6 Mg ha^{-1} yr^{-1}), and Seeded with Erosion Mat (2.1 Mg ha^{-1} yr^{-1}). Results from the Kruskal-Wallis test showed the rank in which treatments performed (table 2). Higher score mean values indicate greater erosion rates. Bare treatments were significantly different from Completely Rocked and Seeded with Erosion Mat ($p = 0.0957$). Cost of BMP treatments was cheapest for Seeded ($9), followed by Seeded with Erosion Mat ($24), Check Dams ($45.40), and Completely Rocked ($119.50) treatments. Cost estimates were based on material, labor, and machine time per 15.2 m ditch section (table 3). Manual labor and machine time pay rates, typically of the area, were $20 and $60 hr^{-1}, respectively. Treatments that implemented

Table 2—Results of the Kruskal-Wallis statistical test. The score mean values show the rank in which treatments performed. Higher score means indicate greater erosion rates. Significant differences between treatments exist at α = 0.1

Treatment	Score Mean	P-value	Chi Square
Bare	34.8a		
Seeded Only	28.4ab		
Check Dams	25.2ab	0.0957	7.89
Completely Rocked	20.5b		
Seeded with Erosion Mat	18.6b		

Table 3—Cost of BMP treatment per 15.2 m ditch section

Treatment	Materials	Cost	Time	Cost	Total
Bare	n/a	n/a	n/a	n/a	$0
Seeded Only	Seed	$3	Manual labor (0.25 hrs)	$5	$9
	Lime	$1			
Seeded Mat	Seed	$3	Manual labor (0.5 hrs)	$10	$24
	Lime	$1			
	Straw mat	$10			
Check dams	#1 surge stone	$5.40	Manual labor (0.5 hrs)	$10	$45.40
			Tractor machine time (0.5 hrs)	$30	
Rocked	#1 surge stone	$44.50	Manual labor (0.75 hrs)	$15	$119.50
			Tractor machine time (1 hrs)	$60	

rock were most expensive, yet did not yield the lowest mean erosion estimates. Seeded treatments were less costly, but seeded only provided less initial soil cover (mean coverage of 50 percent) than Seeded with Erosion Mat (mean coverage of 75 percent) 42 days after ditch re-construction. The least costly and most effective sediment reductive BMP was Seeded with Erosion Mat.

Twenty-four rain events occurred over the 42 day period, totaling 12.2 cm of rainfall. The two largest rain events were approximately 1.5 cm, while most other events were less than 0.33 cm. Measured erosion rates were highly variable within treatments (table 4). All treatments had at least one replication where measured soil erosion rates within the ditch were equal to or less than a undisturbed mixed forest (0.72 Mg ha^{-1} yr^{-1}) (Yoho 1980). However, three bare ditch sections had similar erosion rates to gully type erosion (188-895 Mg ha^{-1} yr^{-1}) (Yoho 1980). Each of these three bare ditch sections had unique circumstances, which may have led to excessive erosion. First, the greatest measured erosion rate (3,279 Mg ha^{-1} yr^{-1}) was located on a 10 percent slope with 65 percent bare soil within the ditch (loose bedrock fragments provided cover) and a road drainage area ~560 m^2. This ditch section lacked adequate water control due to depth to bedrock. At the time of sediment measurement, an obvious concentrated flow path (150 m) starting from the travel surface and leading to the ditch was observed. The second greatest measured erosion rate (900 Mg ha^{-1} yr^{-1}) was located on a 6.6 percent slope with 95 percent bare soil within the ditch. In this circumstance, re-construction of the ditch revealed a seep stemming from a 90 percent bare soil cutslope. Increased moisture conditions and exposed soils may have led to excessive erosion. The third greatest erosion rate (389 Mg ha^{-1} yr^{-1}) had similar characteristic to the greatest measure. This ditch section has a 12 percent slope with 45 percent bare soil within the ditch (loose bedrock fragments provided cover). Additionally, the crown slope (slope from road edge to bottom of the ditch) was 33 percent. These finding demonstrate the highly variable nature of soil erosion, but also exemplify the importance of ditch BMPs and water control to minimize sediment delivery at stream crossings.

CONCLUSIONS

Ditched roads leading to stream crossings have the potential to deliver large quantities of sediment to streams under unique circumstances such as large bare soil contributing areas with steep slopes, bare cutslopes with active seeps, and rutted road surfaces that allow runoff to become concentrated. Findings from this study suggest Completely Rock and Seeded with Erosion Mat treatments were the best at reducing erosion following construction. However, the authors suspect to see greater treatment differences with time. Therefore our findings suggest that re-constructed ditches should drain short sections of road and contain some erosion control BMP in order to reduce erosion and sediment delivery. Some site conditions may merit additional BMP efforts and should be planned for.

LITERATURE CITED

Aust, W.M.; Blinn, C.R. 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: an overview of water quality and productivity research during the past 20 years (1982–2002). Water, Air and Soil Pollution. 4(1): 5-36.

Blake, G.R.; Hartge, K.H. 1986. Bulk density. In: Klute, A., ed. Methods of soil analysis, Part 1, physical and mineralogical methods, Madison, WI: American Society of Agronomy, Inc. and Soil Science Society of America, Inc.: 363-367.

Table 4—Measured erosion rates (Mg ha-1 yr-1) for each ditch section by treatment 42 day after ditch re-construction

Bare	Seeded	Check	Rocked	Mat
0.3	0.1	0.0	0.0	0.0
5.1	0.1	0.0	0.1	0.0
5.5	0.9	0.1	0.2	0.2
11.1	1.2	6.6	0.4	0.4
18.5	5.8	11.9	1.1	1.3
77.5	44.2	32.6	4.1	2.8
173.9	51.5	33.2	10.9	6.3
389.2	91.1	37.5	11.7	7.6
900.4	104.0	77.2	24.4	9.7
3279.5	114.0	101.5	72.4	32.6

Boston, K. 2012. Impact of the Ninth Circuit Court ruling (Northwest Environmental Defense Center v. Brown) regarding forest roads and the Clean Water Act. Journal of Forestry. 110(6): 344-346.

Boston, K.; Thompson, M. 2009. An argument for placing logging roads under the NPDES program. Ecology Law Currents. 36(4): 169-176.

Bridges, T.S.; Gustavson, K.E.; Schroeder, P. [and others]. 2010. Dredging processes and remedy effectiveness: relationship to the 4R's of environmental dredging. Integrated Environmental Assessment and Management. 6(4): 619-630.

Brown, K.R.; Aust, W.M.; McGuire, K.M. 2013. Sediment delivery from bare and graveled forest road stream crossing approaches in Virginia Piedmont. Forest Ecology and Management. 310: 836-846.

Croke, J.; Hairsine, P.; Fogarty, P. 2001. Soil recovery from track construction and harvesting changes in surface infiltration, erosion, and delivery rates with time. Forest Ecology and Management. 143: 3-12.

Environmental Protection Agency. 2012. Notice of intent to revise stormwater regulations to specify that an NPDES permit is not required for stormwater discharges from logging roads and to seek comment on approaches for addressing water quality impacts from forest road discharges. Federal Register. 77(100): 30473-30481.

Environmental Protection Agency. 2014. Stormwater discharges from roads. http://water.epa.gov/polwaste/npdes/stormwater/Stormwater-Discharges-From-Forest-Roads.cfm. [Date accessed: unknown.]

Gee, G.W.; Or, D. 2002. Particle-size analysis. In methods of soil analysis. Part 4, physical methods: 278-284.

Gibson, R.J.; Haedrich, R.L.; Wernerheirn, C.M. 2005. Loss of fish habitat as a consequence of inappropriately constructed stream crossings. Fisheries. 30(1): 10-17.

Grace, J.M. 2005. Forest operations and water quality in the south. Transactions of the American Society of Agricultural Engineers. 48(2): 871-880.

Henley, W.F.; Patterson, M.A.; Neves, R.J.; Lemly, A.D. 2000. Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. Reviews in Fisheries Science. 8(2): 125-139.

MacCurdy, M.G.; Timmons, D.L. 2013. Questions remain for the timber industry after Supreme Court's decision in Decker v. Northwest Environmental Defense Center. Environmental Law. 43: 827-846.

NOAA. 2015. Eastern region station digests. http://www.erh.noaa.gov/ersd/rnk.htm. [Date accessed: unknown.]

Robichaud, P.R.; Brown, R.E. 2002. Silt fences: An economical technique for measuring hillslope soil erosion. Gen. Tech. Rep. RMRS-GTR-94. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 24p.

Saleh, A. 1993. Soil roughness measurement: chain method. Soil and Water Conservation 48(6): 527-529.

Walbridge, T.A., Jr. 1990. The direct location of forest roads. Blacksburg, VA: Walbridge and Associates. 70 p.

Yoho, N.S. 1980. Forest management and sediment production in the south-a review. Southern Journal of Applied Forestry. 4(1): 27-36.

SEDIMENT ACCRETION RATES FOR NATURAL LEVEE AND BACKSWAMP RIPARIAN FORESTS IN THE MOBILE-TENSAW BOTTOMLANDS, ALABAMA

Kathryn R. Kidd, Carolyn A. Copenheaver, and W. Michael Aust[1]

Abstract—Several methods to quantify sediment deposition patterns in riparian forested wetlands have been used during recent decades. In this study, we used a dendrogeomorphic technique with green ash (*Fraxinus pennsylvanica*) to estimate sediment accretion rates for two time periods (1881 to 2012 and 1987 to 2012) along a natural levee (35 m from river) and water tupelo-baldcypress (*Nyssa aquatica-Taxodium distichum*) backswamp (75 m from river) adjacent to the Tensaw River in southwestern Alabama. Sediment accretion rates were significantly higher for the 1987 to 2012 time period along the natural levee ($p = 0.009$; 1.6 cm yr^{-1}) and backswamp ($p = 0.032$; 1.2 cm yr^{-1}) than for the 1881 to 2012 period (0.4 and 0.5 cm yr^{-1}). We further compared dendrogeomorphic sediment accretion rate estimates along the natural levee and backswamp to rates previously obtained using sediment pin and elevation survey methods at increased distances from the river (160 to 330 m) for the 1987 to 2012 time period. Regardless of method used we identified a negative trend in sediment accretion rates as distance from river increased and elevation decreased. Overall, this study demonstrates effective use of dendrogeomorphic techniques while minimizing site visits, compared to the other methods, to estimate sediment accretion rates across temporal and spatial scales.

INTRODUCTION

Development of methods to estimate the amount of sediment trapped by riparian forested wetlands has followed a recognized need for empirical data on sediment accretion in bottomland systems (Boto and Patrick 1979). Such methods include the use of ^{137}Cesium, feldspar clay marker horizons, sediment pins, elevation surveys, and dendrogeomorphic techniques. Each of these methods has advantages and disadvantages (USACOE 1993). Previous studies that have compared these methods have deemed each method valid for estimating sediment accretion rates in riparian forested wetlands (Cahoon and others 2000, Heimann and Roell, 2000, Kleiss 1996, USACOE 1993).

The dendrogeomorphic technique uses the total vertical depth of sediment accreted and tree age of an immediately adjacent tree to provide sediment accretion rates over the life span of trees used in the estimation technique. This method was validated by Hupp and Morris (1990) and has been used in several studies to provide sediment accretion rate estimates for time periods spanning from 10 to over 100 years across varying topographic positions in bottomland forests (Heimann and Roell 2000, Hupp and Bazemore 1993, Kleiss 1996, Phillips 2001). One major benefit of the dendrogeomorphic technique is the estimates are derived following a single site visit in which measurements are taken.

Sediment accretion rates were previously estimated using sediment pin and elevation survey methods in a water tupelo-baldcypress (*Nyssa aquatica-Taxodium distichum*) backswamp at distances of 160 to 330 m from the Tensaw River in southwestern Alabama (Aust and others 2012, McKee and others 2012). However, sediment pin and elevation based sediment accretion rates only represented a 24-year period (1986 to 2010) and these estimates were derived for only backswamp locations. Therefore, the objectives of this study were to: 1) use the dendrogeomorphic technique to estimate sediment accretion rates along a natural levee (35 m from river) and backswamp (75 m from river) adjacent to the Tensaw River for two time periods (1881 to 2012 and 1987 to 2012), and 2) compare dendrogeomorphic rates with previous sediment pin and elevation survey estimates to determine the influence of distance from river on sediment accretion rates.

MATERIALS AND METHODS

Study Area

This study was conducted along the western bank of the Tensaw River within the Mobile-Tensaw River Delta. The site was located in Baldwin County, Alabama

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approximately 50 km north of the Mobile Bay. Climate in this region is subtropical. On average, 1750 mm of precipitation is recorded annually with the greatest precipitation recorded during July (NOAA, 2011). The mean daily temperature is 19.2° C with the highest temperatures observed during June, July, and August.

The study site consisted of a natural levee and water tupelo-baldcypress backswamp (30°57' N, 87°53' W). The natural levee was located parallel to the riverbank and extended approximately 50 m inland until transitioning into the backswamp. The overstory and midstory on the natural levee were characterized by green ash (*Fraxinus pennsylvanica*), overcup oak (*Quercus lyrata*), and water oak (*Q. nigra*) with fewer occurrences of swamp tupelo (*Nyssa sylvatica* var. *bicolor*), black willow (*Salix nigra*), American elm (*Ulmus americana*), and hornbeam (*Carpinus caroliniana*) (Aust and Lea 1991). The backswamp was lower in elevation than the natural levee, which allows overbank floodwaters to pond and persist above the surface into summer months. Change in elevation within the backswamp was initially less than 15 cm (Aust and Lea 1991). Very poorly drained soils of the Levy (fine, mixed, superactive, acid, thermic Typic Hydraquents) series characterized the backswamp (Aust and others 2012). Thus, the backswamp was primarily composed of naturally regenerated flood-tolerant water tupelo and baldcypress, but was also characterized by smaller components of Carolina ash (*Fraxinus caroliniana*), pumpkin ash (*F. profunda*), black willow, red maple (*Acer rubrum*), and water elm (*Planera aquatica*) (McKee and others 2012). Average basal area in the backswamp was initially 75 m^2 ha^{-1} with approximately 85 percent comprised by water tupelo (Aust and Lea 1991). Buttonbush (*Cephalanthus occidentalis*) and dwarf palmetto (*Sabal minor*) were common shrubs in the understory for the backswamp and natural levee, respectively.

Field Methods

Vertical depth of accreted sediment was measured immediately adjacent to 20 green ash trees along transects parallel to the river. Ten co-dominant trees were sampled on the natural levee and ten along the front edge of the backswamp. Using a dendrogeomorphic technique previously validated by Hupp and Morris (1990), the original lateral roots were located on two sides of each ash tree. Once roots were located, the vertical distance between the lateral roots and current ground line was measured to the nearest 0.25 cm. Depths were measured 0.5 m away from the tree base to avoid interference of the base on deposition totals. To determine the time period in which the vertical sediment accumulated, tree cores were extracted below ground level and at DBH (diameter at breast height) for each tree using an increment borer.

Laboratory Methods

Tree cores were air dried and glued to wooden mounts. Cores were sanded with progressively finer sand paper until cellular structures became visible in the cross-sectional view under magnification. A tree-ring chronology was developed using cores extracted at DBH and ground level for the twenty green ash trees. 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 to the nearest 0.01 mm using the LinTab™ 5 RINNTECH® measuring system and TSAP-Win™ software (v. 4.69). 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

The two vertical depths measured at each sampled tree were averaged. Average total sediment deposition values were divided by the respective tree age to estimate sediment accretion rates for the time period in which the tree was living. Average sediment accretion rates were calculated for 1881 to 2012 and 1987 to 2012 time periods along the natural levee and backswamp. Accretion rates associated with green ash trees established from 1881 to 1976 were used to calculate the 1881 to 2012 estimates, while 1987 to 2012 sediment accretion rates were calculated using trees established from 1987 to 1994. Dendrogeomorphic sediment accretion rates were compared between the two time periods along the natural levee and backswamp. Comparisons were made using Wilcoxon-Mann-Whitney tests within the NPAR1WAY procedure in SAS 9.3 (SAS 2012).

Periodic sediment accretion data were available from previous periodic sediment pin (70 to 81 pins) measurements and repeated elevation surveys between 1986 and 2010 at distances of 160, 250, and 330 m from the river (Aust and others 1991, Aust and others 1997, Aust and others 2012, Gellerstedt and Aust 2004, McKee and others 2012, Warren 2001). Sediment pin measurements and elevation surveys were conducted across the backswamp on 20 x 20 m grids. To identify the influence of distance from river on sediment accretion rates, differences among dendrogeomorphic rate estimates at 35 and 75 m and sediment pin measurements at 160, 250, and 330 m were evaluated using a Kruskal-Wallis test. If significant, pairwise Wilcoxon-Mann-Whitney tests were conducted between the distances. Differences among all five distances using the dendrogeomorphic and elevation survey data were evaluated in an identical method. Exact methods

were specified for all nonparametric tests. All statistical analyses were performed at a significance-level $\alpha = 0.05$.

RESULTS AND DISCUSSION

Dendrogeomorphic Sediment Accretion Rates

Mean sediment accretion rates on the natural levee were significantly higher (124 percent, $p = 0.009$) for the 1987 to 2012 time period (1.6 ± 0.2 cm yr^{-1}) than the 1881 to 2012 time period (0.4 ± 0.0 cm yr^{-1}) (fig. 1). Similarly, sediment accretion rates were significantly higher ($p = 0.032$, 72 percent) along the backswamp for the 1987 to 2012 time period (1.2 ± 0.2 cm yr^{-1}) than the 1881 to 2012 time period (0.5 ± 0.1 cm yr^{-1}) (fig. 1). Differences in sediment accretion rates between the two time periods were attributed to altered sediment trapping efficiency due to changes in the dynamics of ground flora and understory vegetation following an adjacent forest harvest disturbance in 1986. This disturbance likely increased levels of light penetration and thus, initially resulted in increased ground flora and understory vegetation densities (Aust and others 1997). An initial increase in low, dense vegetation is likely the main reason for higher sediment accretion rates during the 1987 to 2012 time period. Similar increases in sediment trapping efficiency with increased vegetation have been noted following altered vegetation dynamics following harvesting disturbances (Aust and others 2006, Perison and others 1997). Other studies that have identified temporal patterns in sediment accretion have attributed differences to changes in upstream land use and channelization (Heimann and Roell 2000, Hupp and Morris 1990, Kleiss 1996). While upstream land use could have potentially impacted sediment accretion rates in this study, the localized impacts of the adjacent harvesting disturbance are most likely the reason for the differences. Another potential influence on the longer term (1881 to 2012) sediment accretion rates is erosion. Differences between gross and net sediment accretion rates have been reported in wetlands adjacent to the Olentangy River in Ohio (Mitsch and others 2014). In our study, sampled areas were in direct contact with overbank floodwaters which could have potentially eroded exposed portions of the natural levee and backswamp through time.

Sediment Accretion Rates With Increased Distance

Mean sediment accretion rates decreased as the distance from the river increased from 35 m to 330 m for the 1987 to 2012 time period (fig. 2). Dendrogeomorphic sediment accretion rate estimates for the natural levee and backswamp were higher than in the water tupelo-baldcypress backswamp stand at distances of 160 to 330 m from the river. A Kruskal-Wallis test indicated sediment accretion rates estimated from dendrogeomorphic and sediment pins were significantly different ($p < 0.001$) for at least one distance (35, 75, 160, 250, and 330 m from the river). Pairwise Wilcoxon-Mann-Whitney tests confirmed dendrogeomorphic estimates at 35 m (1.6 ± 0.2 cm yr^{-1}; $p = 0.034$) and

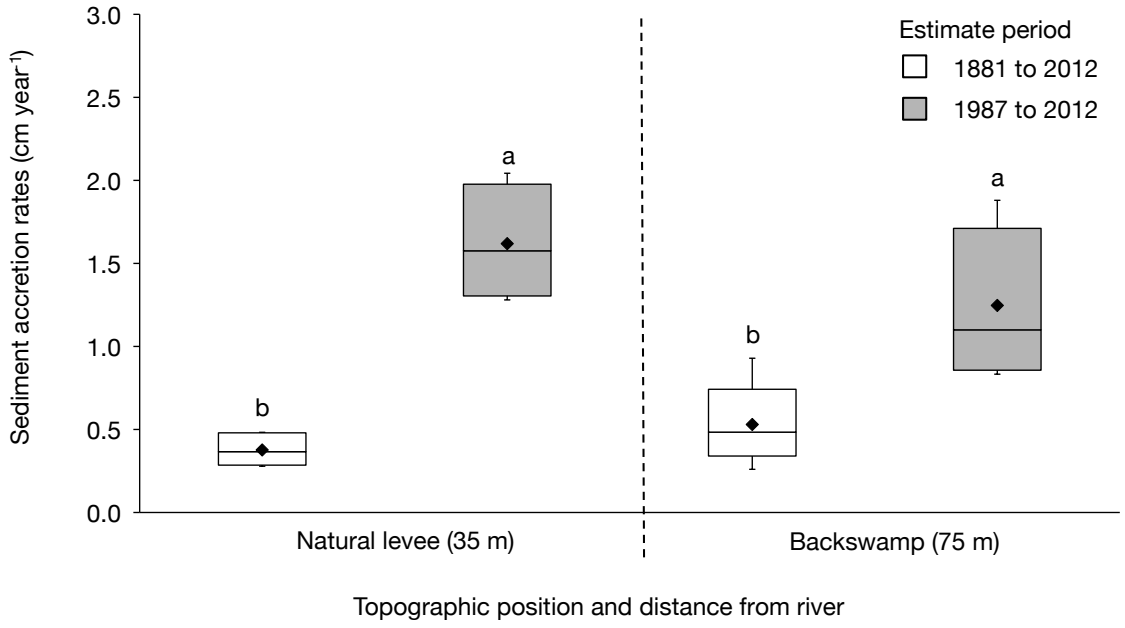


Figure 1—Sediment accretion rates (cm yr^{-1}) estimated using a dendrogeomorphic technique on green ash along a natural levee (35 m from river) and backswamp (75 m from river) adjacent to the Tensaw River. Estimates were derived for different time periods: 1881 to 2012 (tree ages ranged from 60 to 131 years) and 1987 to 2012 (tree ages ranged from 20 to 25 years). Mean estimates are indicated by closed markers. Different letters indicate estimates are significantly at $\alpha = 0.05$ based on Wilcoxon-Mann-Whitney tests.

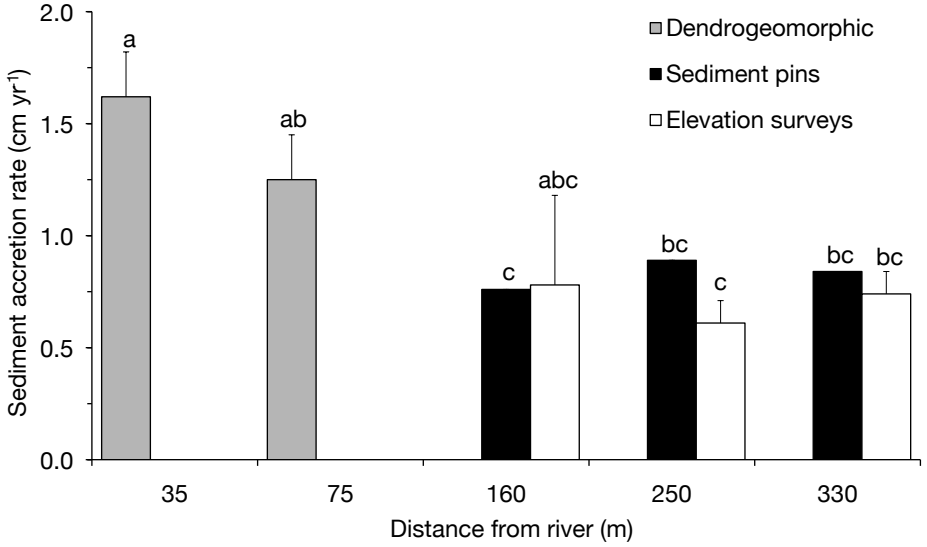


Figure 2—Mean sediment accretion rates (cm yr^{-1}) for the 1987 to 2012 time period as distance from the Tensaw River increased from the natural levee (35 m) across the mature tupelo-cypress backswamp (75 to 330 m) based on dendrogeomorphic estimates, previous sediment pin measurements, and elevation surveys. Different letters indicate estimates are significantly at α = 0.05 based on Wilcoxon-Mann-Whitney tests.

75 m (1.2 ± 0.2 cm yr^{-1}; p = 0.036) were significantly higher than sediment pin accretion rate estimates at 160 m (0.8 ± 0.0 cm yr^{-1}). Dendrogeomorphic sediment accretion rates at 35 m were significantly different from sediment pin estimates at 250 m (0.9 ± 0.0 cm yr^{-1}; p = 0.034) and 330 m (0.8 ± 0.0 cm yr^{-1}; p = 0.047). Analysis of the dendrogeomorphic and elevation survey estimates among the same five distances indicated values for at least one distance were significantly different (p = 0.043). Additional pairwise comparisons revealed dendrogeomorphic estimates at 35 m were significantly different from elevation survey estimates at 250 (0.6 ± 0.1 cm yr^{-1}; p = 0.047) and 330 m (0.7 ± 0.1 cm yr^{-1}; p = 0.047). Sediment accretion rates estimated from the dendrogeomorphic technique at 35 and 75 m were higher than the elevation survey estimates at 160 (0.8 ± 0.4 cm yr^{-1}), but were not statistically different (p = 0.250 for both distances) due to an elevated value at the 160 m distance (fig. 2). As expected, dendrogeomorphic sediment accretion rate estimates at 75 m were significantly higher than elevation survey based estimates at 250 m (p = 0.036) but not at 330 m (p = 0.140).

In our study, as distance from the Tensaw River increased, sediment accretion rates for the 1987 to 2012 time period decreased. At our study site, elevation also decreased with increased distance from the river. Trends in sediment deposition with changes in distance from river and elevation both agree (Asselman and Middlekoop 1995, Johnston and others 1984, Kesel and others 1974) and disagree (Hupp and Bazemore 1993, Hupp and Morris 1990, Kleiss 1996) with our results due to differences among studies in site characteristics and connectivity to sediment-laden waters (Hupp and others 2015). Decreased sediment accretion rates at increased distance in our study may have been influenced by the size of particles deposited. Larger particles generally fall out of suspension first, as overbank floodwater velocity decreases (Boto and Patrick 1979). Therefore, particle size could have influenced total vertical deposition values with change in distance from the river. Along the Mississippi River, the percentage of sand (0.063 to 2 mm) particles that were deposited on the natural levee (68 percent) and the levee backslope (47 percent) were greater than in the backswamp (3 percent) (Kesel and others 1974). Smaller silt (0.002 to < 0.063 mm) and clay (< 0.002 mm) particles made up particles in the backswamp than along the natural levee and levee backslope (Kesel and others 1974). Similar findings in the distribution of particles by size have been reported along transects extending from natural levees into backswamps in the Atchafalaya Basin in Louisiana (Hupp and others 2008) and in floodplains along Long Brank Creek in Missouri (Heimann and Roell 2000).

CONCLUSION

Riparian forests have the capacity to trap and store significant amounts of sediment from adjacent waterways. This study used a dendrogeomorphic technique, which minimized visits, to illustrate temporal patterns in sediment deposition. Further, this study identified spatial patterns of sediment deposition; specifically, decreased sediment accretion rates with increased distance from river and decreased elevation. Trends in repeated sediment deposition have numerous

implications on successional pathways in riparian forested wetlands. Overall, the dendrogeomorphic technique was effective in supplementing sediment accretion rate estimates derived from sediment pin and elevation survey methods to quantify the amount of sediment being trapped through time and space.

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LITERATURE CITED

Asselman, N.E.; Middelkoop, H. 1995. Floodplain sedimentation: quantities, patterns and processes. Earth Surface Processes and Landforms. 20(6): 481-499.

Aust, W.M.; Fristoe, T.C.; Gellerstedt, P.A. [and others]. 2006. Long-term effects of helicopter and ground-based skidding on site properties and stand growth in a tupelo-cypress wetland. Forest Ecology and Management. 226(1-3): 72-79.

Aust, W.M.; Lea, R. 1991. Soil temperature and organic matter in a disturbed forested wetland. Soil Science Society of America Journal. 55(6): 1741-1746.

Aust, W.M.; Lea, R.; Gregory, J.D. 1991. Removal of floodwater sediments by a clearcut tupelo-cypress wetland. Water Resources Bulletin. 27(1): 111-116.

Aust, W.M; McKee, S.E.; Seiler, J.R.; Strahm, B.D. 2012. Long-term sediment accretion in bottomland hardwoods following timber harvest disturbances in the Mobile-Tensaw River Delta, Alabama, USA. Wetlands. 32(5): 871-884.

Aust, W.M.; Schoenholtz, S.H.; Zaebst, T.W.; Szabo, B.A. 1997. Recovery status of tupelo-cypress wetland seven years after disturbance: silvicultural implications. Forest Ecology and Management. 90(2-3): 161-169.

Boto, K.G.; Patrick, W.H. 1979. Role of wetlands in the removal of suspended sediments. In: Greeson, P.E.; Clark, J.R.; Clark, J.E., eds. Wetland functions and values: the state of our understanding. In: Proceedings of national symposium on wetlands, Lake Buena Vista, FL. TPS 79-2. Minneapolis, MN: American Water Resources Association: 479-489.

Cahoon, D.R.; Marin, P.E.; Black, B.K.; Lynch, J.C. 2000. A method for measuring vertical accretion, elevation, and compaction of soft shallow-water sediments. Journal of Sedimentary Research. 70(5): 1250-1253.

Gellerstedt, P.A.; Aust, W.M. 2004. Timber harvesting effects after 16 years in a tupelo-cypress swamp. In: Connor, K.F., ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 4 p.

Heimann, D.C.; Roell, M.J. 2000. Sediment loads and accumulation in a small riparian wetland system in northern Missouri. Wetlands. 20(2): 219-231.

Holmes, R.L. 1983. Computer-assisted quality control in tree-ring dating and measurement. Tree-Ring Bulletin. 43: 69-78.

Hupp, C.R.; Bazemore, D.E. 1993. Temporal and spatial patterns of wetland sedimentation, West Tennessee. Journal of Hydrology. 141(1-4): 179-196.

Hupp, C.R.; Demas, C.R.; Kroes, D.E. [and others]. 2008. Recent sedimentation patterns within the central Atchafalaya Basin, Louisiana. Wetlands. 28(1): 125-140.

Hupp, C.R.; Morris, E.E. 1990. A dendrogeomorphic approach to measurement of sedimentation in a forested wetland, Black Swamp, Arkansas. Wetlands. 10(1): 107-124.

Hupp, C.R.; Schenk, E.R.; Kroes, D.E. [and others]. 2015. Patterns of floodplain sediment deposition along the regulated lower Roanoke River, North Carolina: annual, decadal, centennial scales. Geomorphology. 228: 666-680

Johnston, C.A.; Bubenzer, G.D.; Lee, G.B. [and others]. 1984. Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. Journal of Environmental Quality. 13(2): 283-290.

Kesel, R.H.; Dunne, K.C.; McDonald, R.C. [and others]. 1974. Lateral erosion and overbank deposition on the Mississippi River in Louisiana caused by 1973 flooding. Geology. 2(9): 461-464.

Kleiss, B.A. 1996. Sediment retention in a bottomland hardwood wetland in eastern Arkansas. Wetlands. 16(3): 321-333.

McKee, S.E.; Aust, W.M.; Seiler, J.R. [and others]. 2012. Long-term site productivity of a tupelo-cypress swamp 24 years after harvesting disturbances. Forest Ecology and Management. 265: 172-180.

Mitsch, W.J.; Nedrich, S.M.; Harter, S.K. [and others]. 2014. Sedimentation in created freshwater riverine wetlands: 15 years of succession and contrasts of methods. Ecological Engineering. 72: 25-34.

NOAA [National Oceanic and Atmospheric Administration]. 2011. Summary of Monthly Normals, 1981-2010, National Climatic Data Center. http://www.ncdc.noaa.gov/data-access/land-based-station-data/land-based-datasets/climate-normals/1981-2010-normals-data [Date accessed: February 22, 2015].

Perison, D.; Phelps, J.; Pavel, C.; Kellison, R. 1997. The effects of timber harvest in a South Carolina blackwater bottomland. Forest Ecology and Management. 90(2-3): 171-185.

Phillips, J.D. 2001. Sedimentation in bottomland hardwoods downstream of an east Texas dam. Environmental Geology. 40: 860-868.

SAS. 2012. SAS/STAT 9.3 User's Guide: The NPAR1WAY Procedure. Cary, NC: SAS Institute, Inc.

USACOE [U.S. Army Corps of Engineers]. 1993.Methods of measuring sedimentation rates in bottomland hardwoods. WRP Technical Note SD-CP-4.1. Vicksburg, MS: U.S. Army Corps of Engineers Waterways Experiment Station. 7p.

Warren, S.E. 2001. Sedimentation in a tupelo-baldcypress forested wetland 12 years following harvest disturbance. Blacksburg, VA: Virginia Polytechnic and State University. 72 p. M.S. thesis.

Yamaguchi, D.K. 1991. A simple method for cross-dating increment cores from living trees. Canadian Journal of Forest Research. 21(3): 414-416.

A BRIEF OVERVIEW OF THE 25-YEAR-OLD LONG-TERM SOIL PRODUCTIVITY STUDY IN THE SOUTH

D. Andrew Scott[1]

Abstract—The international Long-Term Soil Productivity experiment began in 1989 in response to the need for Forest Service, U.S. Department of Agriculture managers to understand and monitor the impacts of forest management on site productivity given the expected increase in timber harvesting at the time. It grew to include many other cooperators across the U.S. and Canada and today represents the largest coordinated study of forest management and soil productivity in the world. Twenty-five years after its inception, the Gulf Coastal Plain locations provide many important findings and lessons for management. Overall, soil compaction did not reduce early loblolly pine (*Pinus taeda* L.) survival or growth. In fact, pine volume was increased due to reduced competing vegetation in compacted plots. Intensive organic matter removal (whole-tree harvesting and complete organic matter removal), however, reduced stand volume growth, but only on nutrient-deficient sites. These findings raise questions about current guidelines related to compaction and intensive harvesting. Continued monitoring will help determine how resilient the soils and forests are to these one-time disturbances.

INTRODUCTION

The Long-Term Soil Productivity (LTSP) study began in 1989 as a joint effort between the Forest Service, U.S. Department of Agriculture's National Forest System (NFS) and Research and Development (R&D) branches, but quickly expanded to include cooperators with forest industry, the Canadian Forest Service, and the provincial forestry agencies in Ontario and British Columbia. The study was initiated to determine the fundamental impact of forest management on inherent site productivity and support standards for maintaining soil productivity in managed forests. The research and standards were critically needed because many anecdotal or short-term studies at the time indicated that intensive forestry was potentially detrimental to inherent site productivity, but managers had neither solid research nor defensible standards upon which to act.

The 1980s were a time of major transitions in forest management across all land ownerships, but especially on NFS lands. Timber harvesting on NFS lands had greatly increased from less than 3 billion board feet (bbf) in 1946 to 10-12 bbf from 1960 to 1980 (U.S. Dept. Agriculture Forest Service 2015), and harvesting on NFS lands was projected to reach 20 bbf by 2030 (Thomas 2011). At the same time, however, the rise of the environmental movement of the 1960s and 1970s and new policies, e.g., the National Forest Management Act of 1976 (NFMA), resulted in a much greater concern for the impact of harvesting and forest management in general on other resources such as soil, water, wildlife, and aesthetics. Other forestry and environmental stressors, such as elevated ozone, acid rain, and the fledgling bioenergy movement initiated in the wake of the 1973 and 1979 energy crises were also influencing forest science related to sustainability. Finally, concerns were increasing regarding the sustainability of multiple forest rotations, especially for stands managed intensively for timber production.

Thus, in the late 1980s, a group of scientists and managers from Forest Service R&D and NFS came together to discuss how forest management might impair the inherent productivity of the site (as required by the NFMA), what evidence was available at that time, and finally, how to study it and provide better guidelines to managers. This work resulted in the LTSP's seminal publication at the 7th North American Forest Soils Conference and marked the beginning of the LTSP network. The review concluded that of all the possible mechanisms whereby forest management might alter the inherent productivity of the land, those with the greatest potential were reductions in site organic matter or losses of soil porosity (Powers and others 1990).

The following year, Allan Tiarks of the then-Southern Forest Experiment Station installed the first set of treatment plots on the Palustris Experimental Forest within the Kisatchie National Forest in Louisiana (Tiarks

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and others 1990). Over the next 8 years, twelve more sets of treatment plots were installed from North Carolina to Texas. These plots all had the same basic core design, as did another ca. 50 locations throughout the U.S. and an additional ca. 40 in Canada. They were designed to test the effect of fundamental factors involved in forest management, not any one particular or current treatment. This approach was able to provide experimental bounds beyond those actually employed by management, but it also received criticism because it was not directly relevant to the current practices of the time.

The objectives of this manuscript are to provide a summary of lessons learned over the past 25 years with respect primarily to loblolly pine (*Pinus taeda* L.) responses to organic matter removal and soil compaction and to provide inference to current management questions.

MATERIALS AND METHODS

Site Descriptions

The methods used to install the LTSP study in Louisiana, Mississippi, and Texas have been published previously (Scott and others 2004, Scott and Dean 2006). Briefly, two main treatments, organic matter removal and soil compaction, with three levels of each were installed in a factorial design at 10 separate locations across these three states. Three additional southern sites are located in North Carolina, but this analysis will focus on the Gulf Coastal Plain study sites. The locations were chosen specifically to represent productive sites suitable for intensive pine timber production. Moreover, they were placed on a gradient of potential water deficit: the North Carolina sites had essentially no modeled water deficit (difference between potential evapotranspiration and precipitation), while the Texas sites had the highest deficit in the loblolly pine range. All the sites were dominated by southern pines prior to harvesting. The locations were grouped by state as individual randomized complete block experiments, but similarities among treatments allow results to be grouped across all blocks.

Experimental Design

The three levels of organic matter removal were bole-only harvest (OM0), whole-tree harvest (OM1), and complete organic matter removal, which included all forest floor to the bare mineral soil (OM2). Stumps were not removed from any treatment. The compaction treatments were no machine traffic (C0), moderate compaction (C1), and severe compaction (C2). The treatment plots were 0.4 ha in size, and vegetation composition and biomass and soil characteristics were characterized preharvest. The stands were clearcut harvested and the treatments were imposed. On most sites, the compaction was imposed preharvest because it was more efficient to conduct when the wobble-wheel compactor was not running over unseen stumps. Each main plot was split, with one half receiving multiple herbicide applications (H1) and the other receiving no herbicide (H0). Herbicides varied by site and understory vegetation present but included glyphosate and triclopyr. Volunteer loblolly pine trees were controlled manually for the first few years on all plots except in Texas, where planted pine survival was so poor that volunteer pines were allowed to restore stocking levels. For this analysis, only whole-plot data are reported.

A separate study was conducted on one additional plot on the MS2, LA1, and LA3 sites. These three individual plots were whole-tree harvested (OM1) with operational compaction, i.e., standard feller buncher and skidder operations. These plots were then split in half, with one side receiving no additional treatment (F0). The other side received 243 kg ha^{-1} of diammonium phosphate (DAP, 18-46-0) providing 44 kg ha^{-1} N and 56 kg ha^{-1} P, respectively, at age 4.

Following initial harvesting and treatment, containerized loblolly pine from 10 known half-sib families were planted at a 2.5- by 2.5-m spacing (1600 trees ha^{-1}). The measurement subplots for most measurements consisted of 20 rows of 8 trees on each half-plot (320 trees per whole plot). Every 5 years (and some additional time periods), vegetation was measured for height (10 percent sample for ages 15 and 20) and diameter. Volume (or biomass) was calculated using similar methods as Scott and others (2014). Soil bulk density was sampled every 5 years using the same equipment for all measurements (Grossman and Reinsch 2002).

Design and Analysis

Each treatment was replicated within a state. Mississippi and Texas locations each had three replications located within close proximity and on the same respective soil series. Louisiana had four replications, but the replicate blocks were not all in close proximity to each other and were each on a different soil series. Thus, for the 10 replicate blocks, 6 soil series were represented (table 1). The data were analyzed by a mixed-model analysis of variance where the overall affect was assessed (Federer and King 2007) and effects were considered significant at $\alpha=0.10$.

RESULTS AND DISCUSSION

Compaction

Many misconceptions abound with respect to terms describing soil disturbance caused by forest management. Frequently, foresters (and others) consider soil “compaction” to be a general term encompassing all soil damage associated with any ground-based harvest operation. Compaction is one

Table 1—Site characteristics of the 10 locations of the Long-Term Soil Productivity study in the Gulf Coastal Plain

Site name	Location	Soil series	Soil texture (surface/subsurface)
LA1	Palustris Experimental Forest	Malbis	sandy loam/loam
LA2	Catahoula RD, Kisatchie NF	Glenmora	sandy loam/clay loam
LA3	Catahoula RD, Kisatchie NF	Metcalf	silt loam/silty clay
LA4	Catahoula RD, Kisatchie NF	Mayhew	silt loam/silty clay loam
MS1-3	Chickasawhay RD, DeSoto NF	Freest	loam/loam
TX1-3	Davy Crockett NF	Kurth	loamy fine sand/sandy clay loam

particular type of soil disturbance and consists of a reduction in porosity. It can be present without vertical mixing or soil shear, in which case many preferential flow paths for water may not be drastically disturbed. Rutting usually includes a compaction component, but also incorporates soil shearing and vertical mixing of the soil as the soil flows under pressure. Churning may or may not result in porosity loss because fully saturated soil cannot be compacted, but flow paths and pore sizes are drastically affected and vertical mixing is maximized. These different disturbances affect both dynamic and static soil properties and processes quite differently. This study focused on the impacts of compaction only, not rutting or churning.

Soil compaction has two primary impacts on properties that affect plant roots: increasing soil strength and reducing aeration. Strength (the resistance a plant root receives as it penetrates soil) is increased because friction is increased due to the closer soil particles. Aeration is reduced due to fewer and smaller pores. Root growth can be limited by both of these, and obviously both are dynamically related to water content (Siegel-Issem and others 2002, Scott and Burger 2013). This study was designed with an earlier conceptual model of compaction and root growth in mind, however. It was called the root-growth-limiting bulk density (RGLBD), and it incorporated two metrics: soil texture and bulk density (Daddow and Warrington 1983). Bulk density is a static measure that is closely related to strength at a given water content and is relatively easy to measure. The goal of the study design was to achieve 80 percent of the root-growth-limiting bulk density with the severe compaction treatment and a geometric mean between the uncompacted and severe treatments with the moderate treatment. However, based on the preharvest conditions, the six soils tested ranged from 68 to 93 percent of the RGLBD (mean of 82 percent) without any compaction in the surface 10 cm. Moderate compaction (C1) increased the mean to 89 percent, while severe compaction (C2) reached a mean of 90 percent RGLBD (data not shown). The sandier soils (Kurth, Glenmora) generally achieved a greater absolute amount of compaction than the heavier textured soils (Metcalf, Mayhew). Actual posttreatment bulk density averaged 1.29 Mg m^{-3} across all 10 locations in the C0 plots, and 1.37 and 1.39 Mg m^{-3} in the C1 and C2 plots, respectively (fig. 1).

This compaction was hypothesized to reduce early survival and growth of planted pines through increased soil strength when the soils were dry (growing season) and reduced aeration when wet (dormant season). However, soil compaction had no effect at any site on early survival of planted pines (data not shown). First year survival averaged 94 percent and second year survival averaged 80 percent across the 10 blocks (values included year 3 data from Mississippi, where year 2 data were not available). Survival was relatively stable after the second growing season, at which point normal stand development mortality occurred.

Age 5 volume growth of planted pines was not affected overall by compaction, but volume growth was about 16 percent higher on compacted plots at age 10 (fig. 2), and this increase was sustained through age 15 (and age 20 on the LA and MS sites). This finding was unexpected. One possibility is that the compaction increased the soil water-holding capacity by reducing the size of very large pores. This effect has been seen in coarse volcanic-origin soils in the West, where bulk densities may be relatively low (Gomez and others 2002). These soils simply have too many large pores to hold much water. Given the relatively highly compacted nature of the soils before treatment in this study, it is unlikely that soil water-holding capacity was improved. A second hypothesis is that bulk density recovered very quickly so that growth was unaffected. The data show that while surface soil (0-10 cm) bulk density of compacted plots declined precipitously after initial treatment, reaching pretreatment levels by age 5, the uncompacted soil bulk density was declining at a similar rate (fig. 1). Thus, relative differences were essentially maintained. A third hypothesis is that the compaction had a relatively greater impact on the rooting and growth of competing vegetation. At age 5, competing vegetation biomass on the compacted plots was only 60 percent of that on the uncompacted plots (fig. 3).

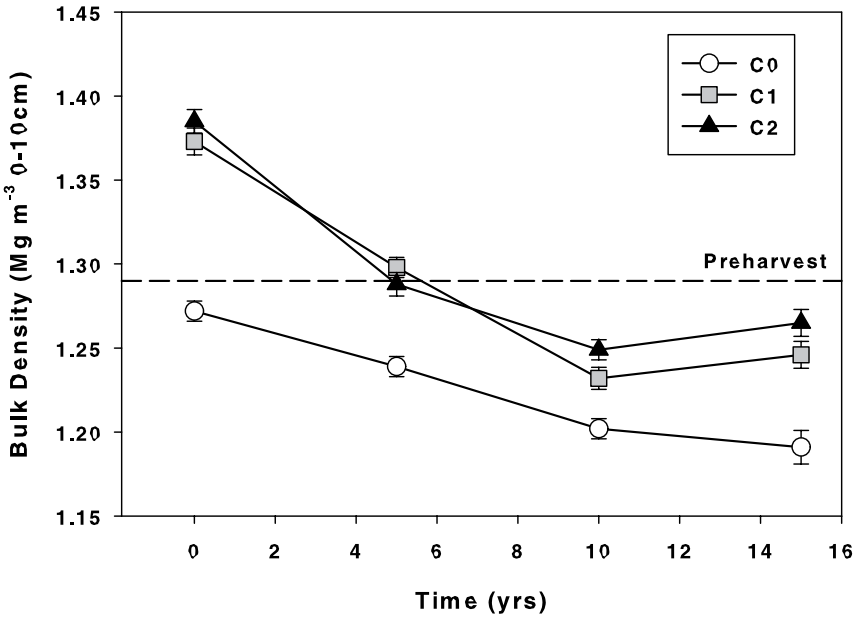


Figure 1—Soil bulk density (0-10 cm) before and after harvest and three levels of applied compaction [no machine traffic (C0), moderate compaction (C1), and severe compaction (C2)] on seven soil series (10 total blocks) across Mississippi, Louisiana, and Texas. Time=0 represents the year of harvest and treatment.

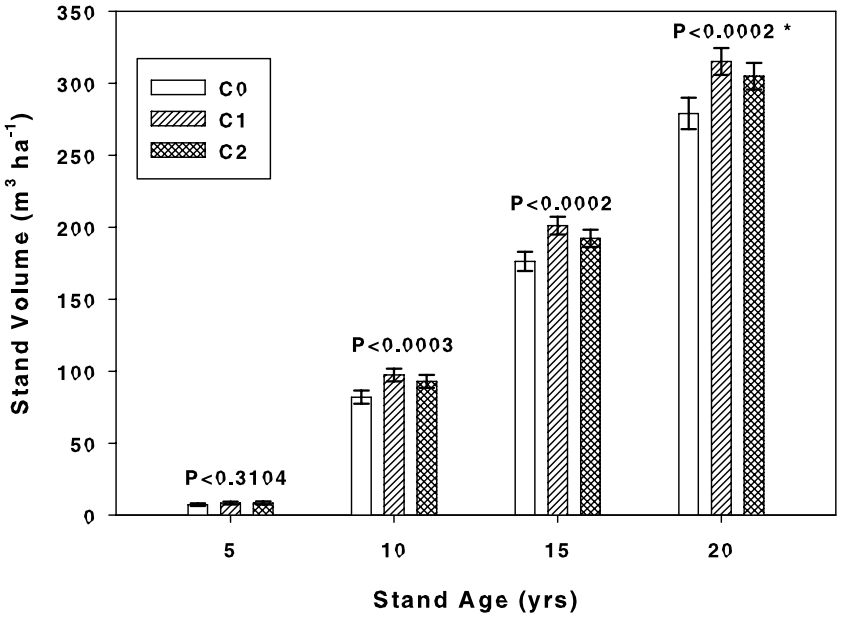


Figure 2—Loblolly pine volume following three levels of compaction [no machine traffic (C0), moderate compaction (C1), and severe compaction (C2)] across 10 blocks of the southern LTSP study in Mississippi, Louisiana, and Texas. *Indicates n=7 because Texas blocks had not reached 20 years old.

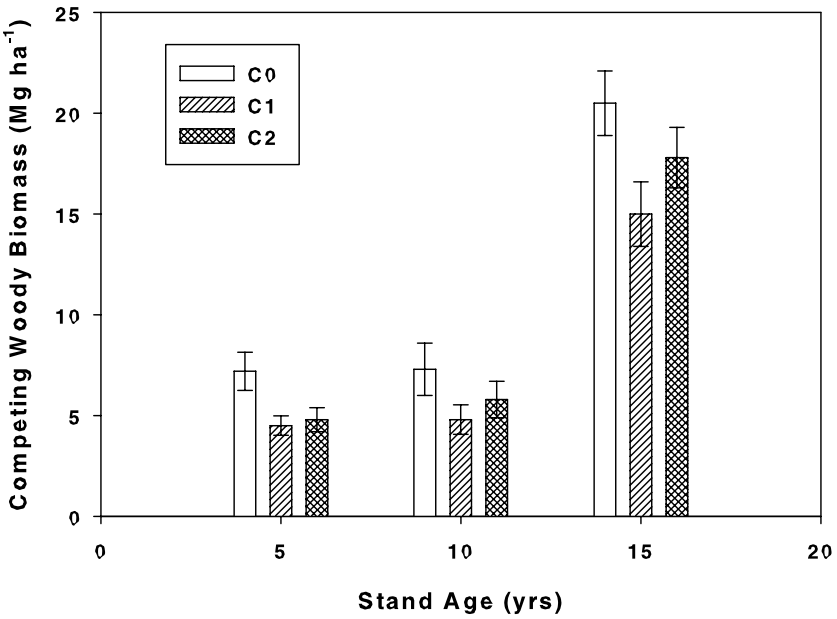


Figure 3—Biomass of competing woody vegetation following three levels of compaction [no machine traffic (C0), moderate compaction (C1), and severe compaction (C2)] across 10 blocks of the southern LTSP study in Mississippi, Louisiana, and Texas.

Apparently the compacting effect reduced the ability of naturally regenerated vegetation to seed, sprout, or achieve root growth in the first few years, giving the planted pines a competitive advantage.

These results, which indicate simple compaction reduced naturally regenerated hardwoods and shrubs but not planted pines, challenge the prevailing management paradigm of restricting traffic to skid trails to avoid the total area disturbed in forests. This paradigm is based on the generally accepted finding that the majority of compaction occurs following the first few equipment passes (Steinbrenner 1955, Hatchell and others 1970, Sidle and Drlica 1981, Greene and Stuart 1985, Han and others 2006). Therefore, it has been widely accepted that reducing compaction is best achieved 1) by avoiding traffic except on dry soils, 2) by using slash to distribute loads, or 3) by minimizing the area compacted. Rutting and churning on moderately well or somewhat poorly drained sites has a much greater potential for reducing site productivity (Aust and others 1995). This raises the question—if we encourage less concentrated traffic, might we both reduce the potential negative effects of more severe disturbance related to multiple passes and disrupted soil structure as well as potentially improve tree productivity through controlling interspecific competition? The Farm Forestry Forties at Crossett Experimental Forest, Arkansas (Guldin 2002), the Hope Demonstration Forest, Arkansas (Farrar Jr. and others 1984), and the Escambia Experimental Forest in Alabama (Barlow and others 2011) provide support for such an approach. The Crossett Farm Forties are naturally regenerated, uneven-aged loblolly and shortleaf pine (*Pinus echinata* Mill.) stands that were harvested annually for >30 years and then on a 5-year cycle for the past 40 years. The Hope Demonstration Forest is also an uneven-aged loblolly and shortleaf pine stand and was harvested six times from 1966 to 1981. The Escambia Farm Forty is a >60-year-old uneven-aged longleaf pine (*Pinus palustris* L.) stand, harvested six times since 1948. Skid trails were never designated, and the uneven-aged, naturally regenerated character necessitated distributed traffic, yet productivity is still high (Guldin 2002, Barlow and others 2011). In fact, the scarification provided by harvesting equipment is useful for preparing seedbeds for naturally regenerated southern pine (Guldin 2004). For landowners with limited resources, this approach might be more useful than designated skid trails that may require costly ripping or bedding site preparation to ameliorate.

Organic Matter Removal

The treatments used in this experiment were not intended to replicate any specific harvesting intensity exactly as conducted within the operational conditions present in 1990; they were intended to provide a clear gradient of organic matter and nutrient removal. Because harvesting technology often changes faster than a long-term study can be conducted, this approach enables the results to be applicable regardless of current technology. These treatments

resulted in an average relative organic matter removal increase of 16 and 54 percent in the OM1 and OM2 treatments, respectively (table 2). Aboveground nitrogen (N) removals were 67 and 134 percent greater in the OM1 and OM2 treatments, respectively, than the OM0 treatment. Phosphorus (P) removals were 78 and 199 percent greater in the OM1 and OM2 treatments, respectively.

When the study was established, it was hypothesized that the primary impacts to site productivity would be observed after the stand reached canopy closure, when overall nutrient demand is highest and supply declines. Survival was not expected to be affected by organic matter removal, as nutrient availability is generally elevated following harvesting. In general, early survival was not greatly affected (data not shown). However, second year survival on the Texas sites was drastically reduced in the OM2 plots (Scott and Stagg 2013). The TX sites not only were at the western (driest) edge of loblolly pine natural range, but the soils in TX were sandy loams (nearly loamy fine sand), and the TX sites experienced a drought between 1 and 2 years postplanting. Apparently, the forest floor provided an insulating effect on soil water content and temperature on these sites.

The organic matter removal effect was significant, however, at every age measured (fig. 4) on average, but the OM2 treatment had no additional effect on volume relative to the OM1 treatment. However, treatment effects were very site specific; volume response has been correlated with site-level productivity and the initial nutrient availability of the site (Scott and others 2004, 2007, 2014, Scott and Dean 2006). These results present a few important implications.

First, and most importantly, removing more than tree boles can reduce site productivity, and our data so far do not indicate a convergence of stand volume through age 20. It is possible that natural processes may restore productivity over longer times, but nothing indicates these processes are having a positive effect yet. The degree of organic matter removal was not important in determining site productivity impact. Removing the hardwoods, understory, and forest floor (OM2) removed over 40 percent more N and 68 percent more P than removing pine boles and tops (OM1) (table 2), yet the relative impact on site productivity was not significant compared to removing whole pine trees. Secondly, productivity lost by intensive organic matter removal was restored with limited rates of fertilizer (fig. 5). On the LA1 and MS3 locations, pine volume on the fertilized halves of whole-tree harvested plots was 56 percent greater than on the unfertilized plot. In LA3, pine volume did not respond to fertilizer application. The overall volume response (37 percent) was thus not significant ($p<0.11$). However, the plot on LA3 only had 73 percent of the mean volume for the other nine plots, and productivity was not reduced by intensive organic matter removal on this site. LA1 and MS3 were both sites in which pine volume was reduced by whole-tree harvest.

Thus, whole-tree harvesting by itself has the potential to reduce site productivity on infertile pine sites in the South, but several soils showed no reductions in productivity with complete organic matter removal. This finding is especially important given the increase in concern for biomass harvesting guidelines, especially for limited-resource landowners or those who choose not to use nutrient amendments (NFS lands, many State lands, naturally regenerated forests, etc.). Most biomass harvest guidelines focus on determining the percentage of initial slash (generally, noncommercial trees and tops) to be retained on site. From a site productivity perspective, this approach is precisely the opposite of the best approach. On very highly productive sites, where slash and organic matter is likely very high, our results did not indicate that a one-time removal of all aboveground organic matter affected productivity negatively, but on infertile soils productivity was affected. Slash and surface organic matter serve purposes other than simply maintaining productivity, however, and these functions should be considered when designing guidelines. But the use of a single percentage based on existing stand volume or biomass may fail to protect productivity on infertile sites and unnecessarily restrict harvest on highly fertile sites. These data also show that conservative rates of

Table 2—Quantities of biomass, nitrogen, and phosphorus removed in three organic matter removal treatments [bole-only harvest (OM0), whole-tree harvest (OM1), and complete organic matter removal (OM2)] from 10 blocks of the southern LTSP study across Mississippi, Louisiana, and Texas

Treatment	Biomass (Mg ha^{-1})	Nitrogen (kg ha^{-1})	Phosphorus (kg ha^{-1})
OM0	127.5 (23.5)	130.1 (51.6)	8.6 (4.72)
OM1	148.0 (24.9)	216.8 (61.7)	15.3 (5.57)
OM2	196.6 (54.3)	304.1 (117)	25.7 (13.4)

Note: values are means with standard deviations in parentheses.

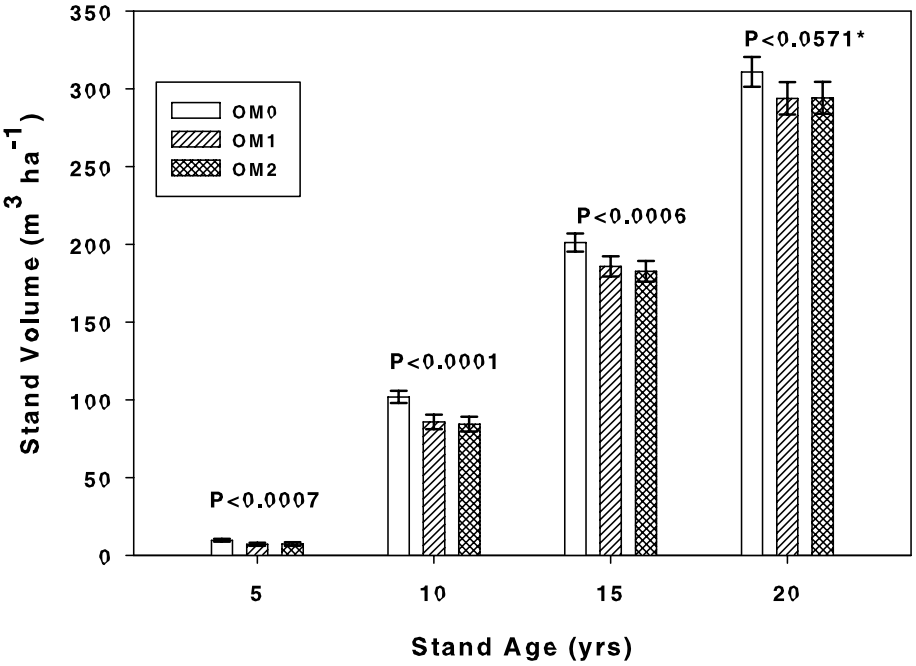


Figure 4—Pine volume following three levels of organic matter removal [bole-only harvest (OM0), whole-tree harvest (OM1), and complete organic matter removal (OM2)] across 10 blocks of the southern LTSP study in Mississippi, Louisiana, and Texas. *Indicates n=7 because Texas blocks had not reached 20 years old.

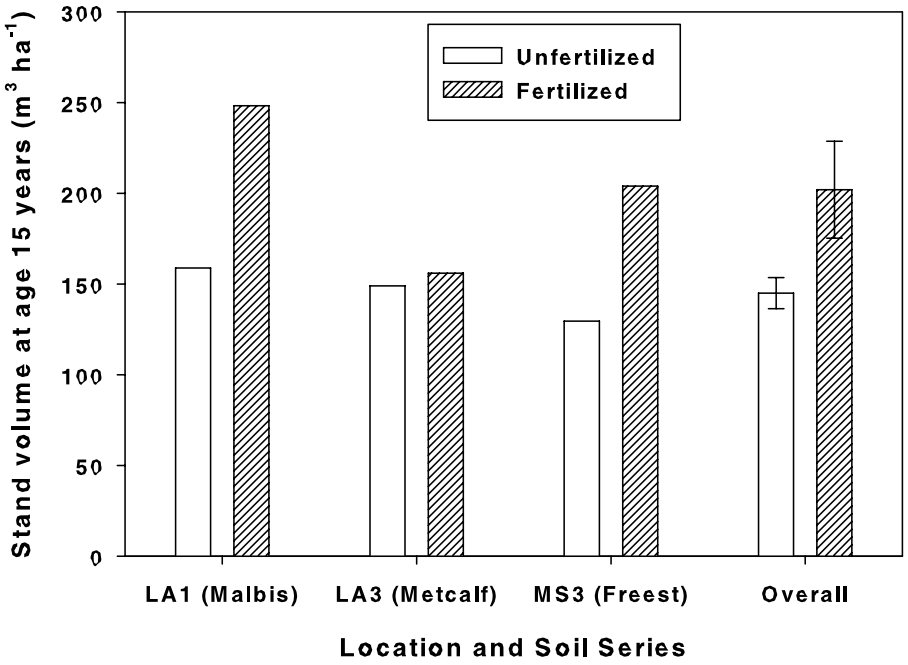


Figure 5—Pine volume response to fertilizer (56 and 44 kg ha^{-1} of phosphorus and nitrogen, respectively, applied at age 4) on three whole-tree harvested plots with operational compaction at age 15 (see table 1 for location information).

traditional fertilizers restore lost productivity, at least on these site types. Concerns have been raised about potential productivity loss from calcium depletion in the South (Huntington 2000), which could be exacerbated by biomass harvesting, but these data provided no indication of a calcium depletion.

CONCLUSIONS

Forest harvesting has the potential to affect long-term site productivity primarily through two mechanisms: removing organic matter and nutrients from the site and altering soil porosity and the air-water balance. Planted loblolly pine productivity through 15-20 years was not negatively affected by soil compaction across six soil series, including two soils with heavy clay subsoils. Soils were not rutted or churned, but the absence of any negative impact was both surprising and important for understanding the resilience of forest soils to impacts. Conversely, removing more organic matter than just the boles reduced long-term pine growth on infertile soils, including loamy textured soils. Removing all aboveground organic matter did not reduce growth on the more fertile soils, illustrating that assessing site capacity for intensive harvesting is likely more important than harvest intensity for maintaining long-term site productivity. The southern LTSP study has been a vital component of the international LTSP network, and continued monitoring and process-level research will continue to assess how resilient these forests are to harvesting.

ACKNOWLEDGMENTS

The international LTSP network currently comprises well over two dozen scientists and a similar number of forest managers, and the continued progress on any one study location is intimately related to the much greater value of the overall network than of any one region. For that, I wish to thank all current and past participants for their collaborations. A special recognition goes to Allan Tiarks, the scientist who installed all 10 locations covered in this manuscript and completed the early measurements. Similarly, Rick Stagg deserves extraordinary praise as the forester in charge of actual ground operations for both installation and all measurements since 1991. Jerry Ragus and Emanuel Hudson, soil scientists from Region 8, and many employees of the individual National Forests were essential for establishing and maintaining the study. Finally, I offer a special gratitude to Felix Ponder and Bob Powers. Felix installed LTSP plots in Missouri, and his collaborations with Purdue University led to my first job as an undergraduate research technician and my initial interest in the LTSP program. Bob was the guiding force behind the entire program from its inception until his retirement in 2008, and specifically mentored and encouraged me greatly from my first day with the Forest Service until his untimely passing.

LITERATURE CITED

Aust, W.M.; Tippett, M.D.; Burger, J.A.; McKee Jr., W.H. 1995. Compaction and rutting during harvesting affect better drained soils more than poorly drained soils on wet pine flats. Southern Journal of Applied Forestry. 19(2): 72-77.

Barlow, R.; Kush, J.S.; Boyer, W.D. 2011. Sixty years of management on a small longleaf pine forest. Southern Journal of Applied Forestry. 35(1): 1-4.

Daddow, R.L.; Warrington, G.E. 1983. Growth-limiting soil bulk densities as influenced by soil texture. WSDG-TN-00005. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Watershed Systems Development Group. 17 p.

Farrar Jr., R.M.; Murphy, P.A.; Colvin, R. 1984. Hope farm woodland: 33-Year Production in an uneven-aged loblolly-shortleaf pine stand. Journal of Forestry. 82: 476-479.

Federer, W.T.; King, F. 2007. Combining split plot or split block designed experiments over sites. In: Variations on split plot and split block experimental designs. New York: John Wiley & Sons: 213-237.

Gomez, A.G.; Powers, R.F.; Singer, M.J.; Horwath, W.R. 2002. Soil compaction effects on growth of young Ponderosa pine following litter removal in California's Sierra Nevada. Soil Science Society of America Journal. 66(4): 1334-1343.

Greene, W.D.; Stuart, W.B. 1985. Skidder and tire size effects on soil compaction. Southern Journal of Applied Forestry. 9: 154-157.

Grossman, R.B.; Reinsch, T.G. 2002. Bulk density and linear extensibility. In: Dane, J.H.; Topp, G.C. eds. Methods of soil analysis, part 4--physical methods. Madison, WI: Soil Science Society of America: 201-228.

Guldin, J.M. 2002. Continuous cover forestry in the United States--experience with southern pines. In: von Gadow, K.; Nagel, J.; Saborowski, J. eds. Continuous cover forestry: assessment, analysis, scenarios. Dordrecht, Netherlands: Kluwer Academic Publishers: 295-307.

Guldin, J.M. 2004. Reproduction cutting methods for naturally regenerated pine stands in the South. In: Rauscher, H.M.; Johnsen, K. eds. Southern forest science: past, present, and future. Gen. Tech. Rep. SRS-75. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 83-95.

Han, H.-S.; Page-Dumroese, D.S.; Han, S.-K. 2006. Effects of slash, machine passes, and soil moisture on penetration resistance in a cut-to-length harvesting. International Journal of Forest Engineering. 15: 11-24.

Hatchell, G.E.; Ralston, C.W.; Foil, R,R. 1970. Soil Disturbances In Logging: Effects on soil characteristics and growth of loblolly pine in the Atlantic coastal plain. Journal of Forestry. 68: 772-775.

Huntington, T.G. 2000. The potential for calcium depletion in forest ecosystems of southeastern United States - review and analysis. Global Biogeochem Cycles. 14(2): 623-638.

Powers, R.F.; Alban, D.H.; Miller, R.E. [and others]. 1990. Sustaining site productivity in North American forests: problems and prospects. In: Gessell, S.P.; Lacate, D.S.; Weetman, G.F.; Powers, R.F. eds. Sustained productivity of forest soils. Vancouver, BC: University of British Columbia, Faculty of Forestry: 49-79.

Scott, D.A.; Burger, J.A. 2013. Longleaf and loblolly pine seedlings respond differently to soil compaction, water content, and fertilization. Plant Soil. 375(1-2): 255-265.

Scott, D.A.; Dean, T.J. 2006. Energy trade-offs between intensive biomass utilization, site productivity loss, and ameliorative treatments in loblolly pine plantations. Biomass and Bioenergy. 30(12): 1001-1010.

Scott, D.A.; Eaton, R.J.; Foote, J.A. [and others]. 2014. Soil ecosystem services in loblolly pine plantations 15 years after harvest, compaction, and vegetation control. Soil Science Society of America Journal. 78(6): 2032-2040.

Scott, D.A.; Novosad, J.; Goldsmith, G. 2007. Ten-year results from the North American long-term soil productivity study in the western Gulf coastal plain. In: Furniss, M.; Clifton, C.; Ronnenberg, K. eds. Advancing the fundamental sciences: proceedings of the Forest Service national Earth sciences conference. Gen. Tech. Rep. PNW-689. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station: 331-340.

Scott, D.A.; Stagg, R.H. 2013. Plant community responses to soil disturbance and herbicide treatments over 10 years on the Texas LTSP study. In: Guldin, J.M. ed. Proceedings of the 15th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-175. e-GTR. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 21-27.

Scott, D.A.; Tiarks, A.E.; Sanchez, F.G. [and others]. 2004. Forest soil productivity on the southern long-term soil productivity sites at age 5. In: Connor, K.F. ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 372-377.

Sidle, D.M.; Drlica, R.C. 1981. Soil compaction from logging with a low-ground pressure skidder in the Oregon Coast Ranges. Soil Science Society of America Journal. 45: 1219-1224.

Siegel-Issem, C.M.; Burger, J.A.; Powers, R.F. [and others]. 2002. Seedling root growth as a function of soil density and water content. Soil Science Society of America Journal. 69: 215-226.

Steinbrenner, E.C. 1955. The effect of repeated tractor trips on the physical properties of forest soils. Northwest Science. 29(4): 155-159.

Thomas, J.W. 2011. The future of the national forests : who will answer an uncertain trumpet? Fair Chase. 26: 16-23.

Tiarks, A.E.; Kimble, M.S.;Elliott-Smith, M.L. 1990. The first location of a national, long-term forest soil productivity study: methods of compaction and residue removal. In: Coleman, S.S.; Neary, D.G. eds. Proceedings of the 6th biennial southern silvicultural research conference. Gen. Tech. Rep. SE-70. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 431-442.

U.S. Department of Agriculture Forest Service. Cut and Sold Reports. [Online]. Available at http://www.fs.fed.us/forestmanagement/products/sold-harvest/cut-sold.shtml. [Date accessed: August 25, 2015].

SOIL CO_2 EFFLUX AND WATER USE EFFICIENCY ACROSS DIVERSE COVER TYPES IN SOUTHERN APPALACHIAN HARDWOOD FORESTS

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Abstract—We are investigating biogeochemical cycling in a mixed hardwood forest in the Ridge and Valley physiographic province in Montgomery County, Virginia. The broad aim of the study is to understand how carbon, water and nutrient cycles vary among diverse stand types in a relatively small spatial area. The specific objectives here are to determine patterns in soil CO_2 efflux or respiration (R_s) and water use efficiency among cover types. Four 0.02 ha sample plots, replicated four times, were established in four cover types – white oak (WO, *Quercus alba*), Scarlet oak (SO, *Q, coccinea*), chestnut oak (CO, *Q. montana*) and mixed pine – oak (PO, *Pinus* spp., *Quercus* spp.). In each plot, diameter at breast height was measured on all trees greater than 5.1 cm. R_s was measured monthly. Foliage from two dominant or co-dominant trees in each plot was sampled for water use efficiency using $\delta^{13}C$ discrimination analysis. Soil temperature alone explained 93 percent of the variation in R_s and the variance due to cover type was not significant. Water use efficiency was greatest in SO where $\delta^{13}C$ was -26.6 per- mille which was significant different than CO (-28.3) and WO (-28.5) values.

INTRODUCTION

Natural hardwood stands constitute a considerable proportion of US forests, yet much is unknown about factors controlling carbon and nutrient cycling in this ecosystem. Largely because of changes in water availability, as influenced by upslope subsidies, productivity and species composition in these forests change over very short distances from ridge top to cove landscape positions. Currently these forests are experiencing pressure from land use and environmental changes which influence their functioning and the services derived from them (Turner and others 2003). Most models that predict influences of land use and global change on the fluxes of carbon, water, and nutrients do not include considerations at the finer scale of cover types in these very diverse temperate forest ecosystems. Studies have shown the responses of carbon, water and nutrient cycling differed from one species to another. Raich and Tufekcioglu (2000) found that soil respiration rates differ among plant biomes and this was because vegetation has influence upon soil microclimate, the quantity of detritus and litter fall. At a global scale, Raich and Tufekcioglu (2000) found that soil respiration (Rs) increases with litter fall in relatively mature ecosystem, suggesting that soil respiration increased in sites with greater rates of detritus production, plant detritus provide the energy that drives the respiration. This trend at a local scale has been suggested to be poorly correlated (Reiners 1968, Ellis 1969) due to local factors such as soil type, inter-site habitats, species composition, or land use history may obscure correlations that are obvious at broader scales. The associations of temperature and moisture content to respiration have been reported differently. Davidson and others (1998) found the spatial and temporal variation in soil respiration in temperate forests, ranging from 5.3-8.5 Mg ha^{-1} yr^{-1} across soil drainage classes. In this system, an exponential function predicting soil CO_2 efflux using soil temperature accounted for about 80 percent of seasonal variation in efflux across the sites (Davidson and others 1998).

Water use efficiency (WUE) is a ratio of net CO_2 assimilation and transpiration and it generally increases during drought conditions. WUE can be indirectly estimated at the leaf level via carbon isotope discrimination which favors lighter isotopes to the heavier ones (Monclus and others 2005). At a particular site, ranking of leaves of C_3 species according to their discriminative values should give a ranking of their WUE at the leaf level (Monclus and others 2005).

This study is undertaken to examine R_s and WUE as measured with ^{13}C isotope discrimination in four cover types (stand types), white oak (WO, *Quercus alba* L.), scarlet oak (SO, *Quercus coccinea* Muench.), chestnut oak (CO, *Quercus montana* Willd.) and mixed pine-oak (PO, *Pinus* spp. and *Quercus* spp.) in the ridge and valley physiographic province in southwest Virginia. Our results will improve our understanding of how carbon and water use differ across the dominant cover types in natural southern Appalachian hardwoods and how

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external forces might impact future ecosystem function. Our specific hypotheses are 1) R_s will be largely driven by soil temperature and moisture, 2) this R_s relationship will differ among the cover types, and 3) WUE of species will be largely driven by the soil moisture gradient.

SITE LOCATION

The study site is located in the ridge and valley physiographic province in Montgomery County, Virginia. The geology is composed primarily of limestone and sandstone. Specifically, the soils are a mixture of Berks (Typic Dystrudepts) and Weikert (Lithic Dystrudepts), which are acidic and well drained soils with low available water holding capacity and low nutrient status (Copenheaver and others 2006). The elevation in this region varies from 152m at the valley to more than 732m (Fenneman 1938). The specific study area is a predominately southern facing slope. The mean annual temperature is 10.8 ˚C, (January mean is -0.6 ˚C and the July mean is 21.7 ˚C). The average annual precipitation is 1020 mm with maximum being in April and September and annual frost free days are 162 (Rhoades 1995).

EXPERIMENTAL DESIGN

We examined four common cover types: white oak, scarlet oak, chestnut oak and mixed pine-oak. Within each of these cover types, four circular 1/50 ha plots were established. The experimental design used is a randomized complete block with four replications (fig. 1).

MATERIALS AND METHODS

Aboveground Biomass

In each plot, all trees with >5.1 cm diameter at breast height (DBH; at 1.37m above the ground), were measured for diameter. The precise point at which the diameter measurement was taken was tagged for the subsequent remeasurements. Algometric equations developed by Jenkins and others (2003) and Ter-Mikaelian and Korukhin (1997) were used to calculate aboveground dry mass from DBH.

Water-Use Efficiency

Water use efficiency was estimated using a stable carbon isotope ($\delta^{13}C$) discrimination technique (Sands and Mulligan, 1990). In mid-July, growing season foliage was sampled from the upper third of the crown of dominant and co-dominates tree species in each plot using a shotgun. Two dominant or co-dominant trees representing the cover type designation in each plot were chosen, with the exception of the mixed oak-pine plots where four trees were used, two each from Table Mountain pine (*Pinus pungens* Lamb.*)* and chestnut oak. Leaves were oven dried at 65˚C, ground to a fine powder, and analyzed for $\delta^{13}C$ on an Isoprime-100 Isotope Ratio Mass Spectrometer (Elementar Americas Inc, Mt. Laurel, NJ 08054-3409).

Soil CO_2 Efflux

Soil CO_2 efflux was measured monthly at three sub-samples in every plot. A modified Li-Cor 6200 CO_2 gas

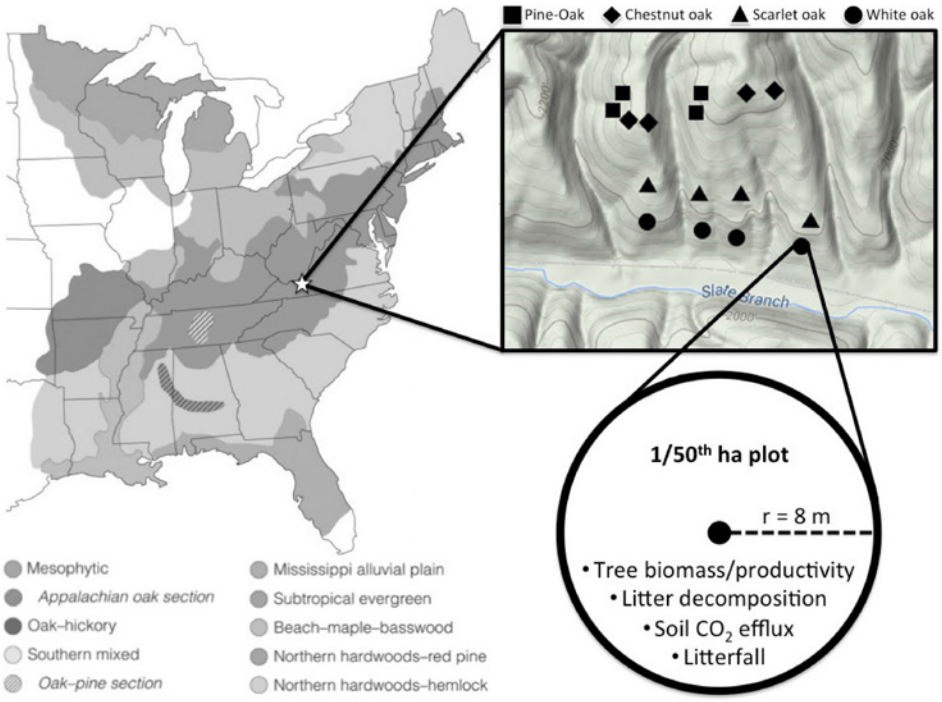


Figure 1—Study site location, Montgomery County, Virginia and the distribution of the sample plot within the larger mesophytic forest ecoregion (adapted from Smith and Smith 2012).

analyzer with a 24.5 cm diameter chamber was used (Tyree and others 2014). Soil temperature and moisture content were measured at 12 cm and 0-10 cm depth, respectively, at each sub-sample point.

STATISTICAL ANALYSES

One-way analysis of variance (AVOVA) was used to test the significances of the above ground biomass across the canopy types. Tukey's HSD was used to separate the means at a statistical significance of 5 percent. R_s data were analyzed with multiple linear regressions to determine relationships with soil temperature, moisture content and stand characteristics. Significant differences in means were determined by ANOVA and HSD at 5 percent significant difference.

Foliar $\delta^{13}C$ was analyzed on an Isoprime-100 Isotope Ratio Mass Spectrometer (Elementar Americas Inc, Mt. Laurel, NJ 08054-3409). One-way ANOVA determined the significances of water use efficiencies of the dominant trees across stand types, LSD determined the statistical mean differences at significant level of 5 percent.

RESULTS AND DISCUSSION

Cover types were accurately delineated (fig. 2). The above ground biomass of each cover type is largely dominated (over 75 percent in all cases) by their respective species names (fig. 2). In the PO stand, 69 percent of the stand biomass is pine and 27 percent oak. The cover type distribution in respect to their stem number and aboveground biomass is associated with availability of moisture content along the gradient. Whittaker (1956) showed moisture controls the distribution along the gradient in Appalachian hardwood forest, with the areas on lower slopes having higher stems/ha than upper slopes. In this study the number of stems/ha increased with elevation, WO (280), SO (395), PO (495) and CO (530). The decrease in stem density at lower elevation may be associated with changes in past stand use such as influences from nearby mining, and harvesting (Copenheaver and others 2006); stands at the lower slope position are more accessible due to a nearby railway and road.

WUE is the amount of carbon fixed per unit water transpired (Sands and Mulligan 1990), which indirectly can be estimated via carbon isotope discrimination. Plants which show less ^{13}C discrimination have greater WUE (table 1). SO had significantly higher $\delta^{13}C$ discrimination indicating it has higher WUE than other species (table 1). However, soil moisture content was not the lowest in SO (12.31 percent) and was nearly as high as WO (12.67 percent). This suggests that soil moisture contents alone do not directly explain water use efficiency. Some studies have shown WUE vary from xeric to mesic environments. Plants growing under xeric condition, where moisture content is low,

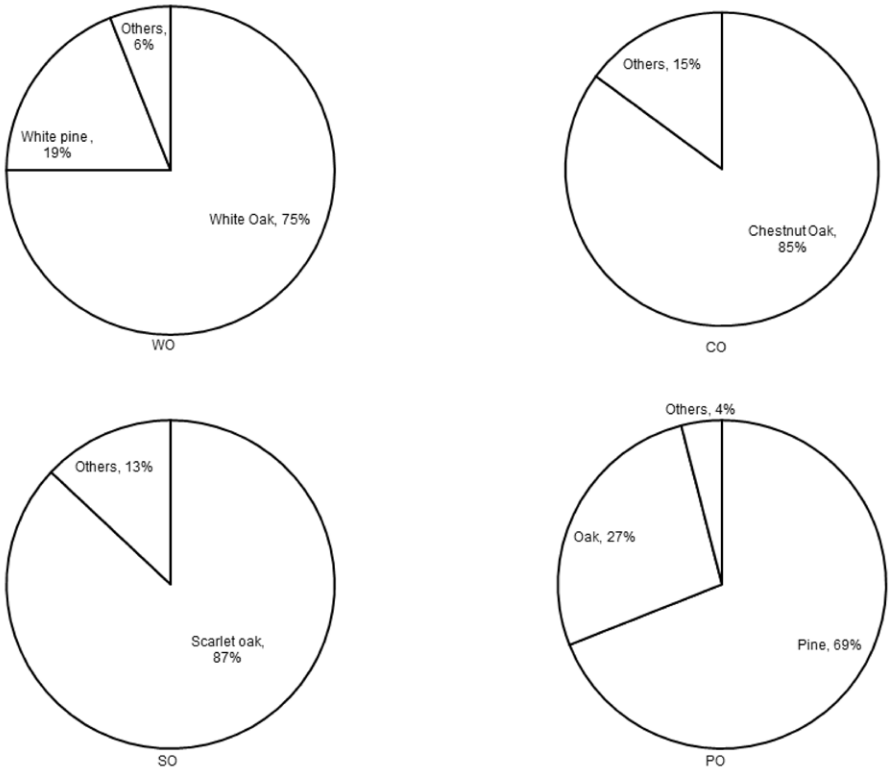


Figure 2—Aboveground biomass percentage for four ridge and valley cover types, white oak (WO), pine-oak (PO), scarlet oak (SO) and chestnut oak (CO) examined in the study.

Table 1—Mean $\delta^{13}C$ discrimination and soil moisture for four ridge and valley tree cover types

Cover Type	δ13C discrimination	Mean Soil Moisture (July-October)
Scarlet oak	-26.60 A	12.31
Pine-Oak:		
Table mountain pine	-27.81 AB	10.29
Chestnut oak	-28.51 B	
Chestnut oak	-28.28 B	9.35
White oak	-28.45 B	12.67

experience higher WUE than plants growing under mesic conditions as in the case of radiata pines (Sands and Nambiar 1984). Principally stomatal closure during drought reduces transpiration rate more than the photosynthesis rate and WUE increases. Many factors influence WUE such as vapor pressure deficit (Sandford and Jarvis 1986), wind speed (Smith 1980) and vegetation height increases (Kriedemann and Barrs 1983). Further, our measure of soil moisture content may not be adequately estimating water availability on these rocky upland sites.

R_s varied greatly over the seven months of measurements with few differences between cover types (fig. 3a). In July and August, efflux rates were greatest in CO and WO. WO R_s fell sharply in September; Rs fell sharply in all cover types in October and then remained low through the winter months.

Soil temperature followed the same pattern for all cover types (fig. 3b) while soil moisture (fig. 3c) between the cover types varied more than temperature. WO and SO generally had higher soil moisture and CO and P-O generally lower moisture (fig. 3c, table 1).

R_s tracked soil temperature closely (fig. 3a and 3b) and when modeled, soil temperature alone explained 93 percent of the variance in R_s (fig. 4). Despite soil moisture varying among covers types (fig. 3c), moisture did not explain a significant amount of variation in R_s. Cover type also did not influence the R_s and temperature relationship. This is likely because soil temperature was very consistent across cover types (fig. 3b). Similar results have been shown by others studies, Davidson and others (2000) reported on exponential function of CO_2 efflux to soil temperature accounting 80 percent of seasonal variation in efflux across sites and a decline in R_s was associated with a decrease in soil matric potential. Boone and others (1998) reported water content was unrelated to R_s (R^2=0.86, temperature alone and R^2= 0.90, temperature and moisture content) but there was a strong correlation between temperature and R_s. Likewise, Templeton and others (2015) found soil temperature to be the most important driver and soil moisture explaining a very small amount of variation in R_s in loblolly pine stands. Some studies reported differently, Martin and Bolstad (2005) found that in deciduous trees, R_s was correlated to soil temperature, moisture and site conditions and that Rs varied with severity, timing and duration of drought. Witkamp (1966) and Kramer and Boyer (1995) reported the complex interdependency that existed between soil temperature, water content, soil properties and structures in influencing the rate of soil respiration, and Wildung and others (1975) reported soil temperature and moisture being the principle factors in influencing the plant root decomposition and the rate of CO_2 evolution in the soils. The results of the present study indicate no interaction between soil temperature and soil moisture content on R_s. Our study was for seven months covering late summer, fall and winter. In a much longer time frame with a greater range in soil moisture an influence of soil moisture on R_s may develop.

CONCLUSIONS

- R_s tracked closely with soil temperature. Soil temperature alone explained 93 percent of the variance in Rs.
- Cover type did not influence Rs. One model fit all cover types.
- Water use efficiency as indicated by $\delta^{13}C$ discrimination was greatest in scarlet oak. Elevation-driven soil moisture gradients alone were not sufficient in explaining water use efficiencies of the canopy dominant species.

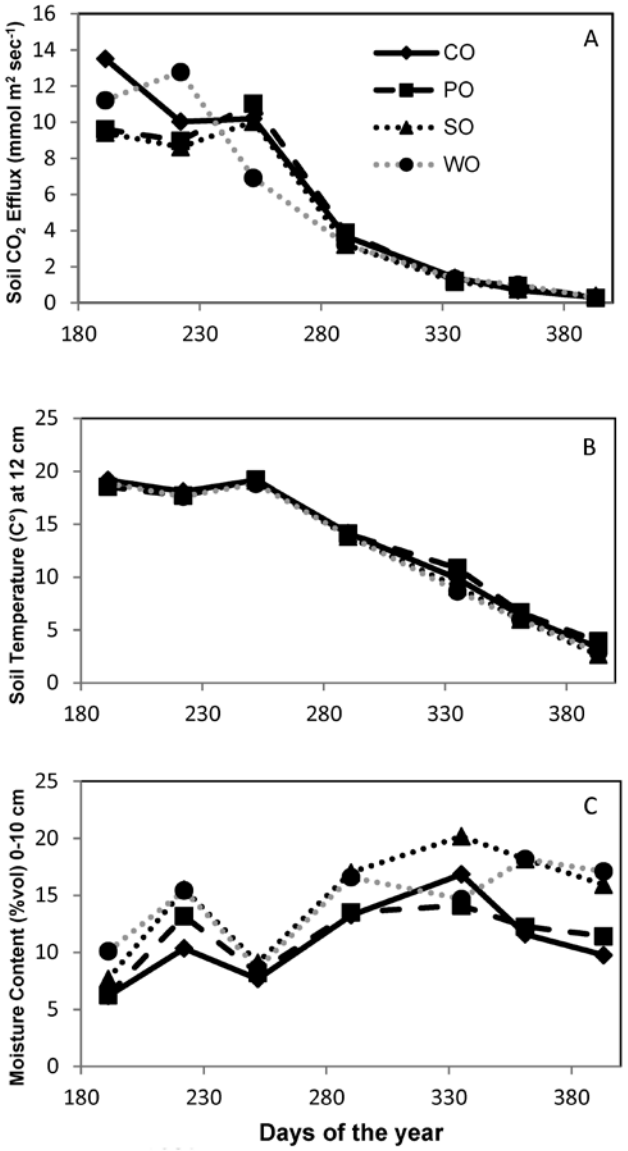


Figure 3—The relationship between soil CO_2 efflux (a), soil temperature (b) and soil moisture content (c) in four common Appalachian cover types; White oak (WO), Scarlet Oak (SO), Chestnut Oak (CO) and Pine Oak (PO; mixture of pine and oak).

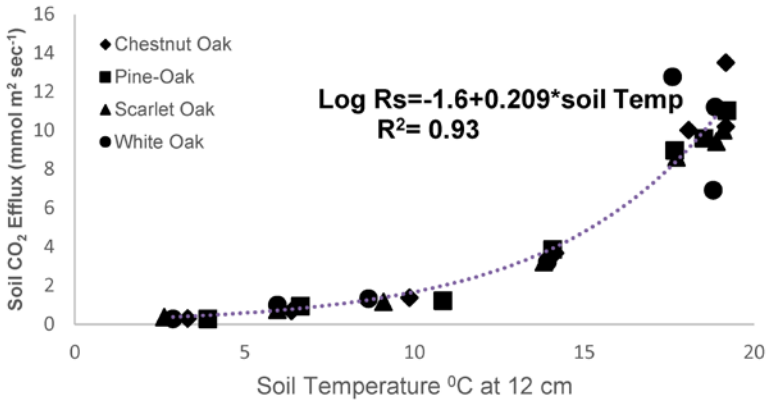


Figure 4—Relationship between soil temperature and soil CO_2 efflux (Rs) in four common Appalachian cover types.

LITERATURE CITED

Boone, R.D.; Nadelhoffer, K.J; Canary, J.D; Kaye, J.P. 1998. Roots exert a strong influence on the temperature sensitivity of soil respiration. Nature. 396: 570-572.

Copenheaver, C.A.; Matthews, J.M; Showalter, J.M.; Auch, W.E. 2006. Forest stands development patterns in the Southern Appalachians. Northeastern Naturalist. 13: 447-494.

Davidson, E.A.; Belk, E.; Boone, R.D. 1998. Soil water and temperature as independent or confounded factors controlling soil respiration in temperate mixed hardwood forests. Global Change Biology. 4: 217-227.

Davidson, E.A.; Verchot, L.V.; Cattanio, J.H. [and others]. 2000. Effects of soil water content on soil respiration in forests and cattle pastures of eastern Amazonia. Biogeochemistry. 48: 53-69.

Ellis, R.C. 1969. The respiration of the soil beneath some Eucalyptus forest stands as related to the productivity of the stands. Australian Journal of Soil Resource. 7: 349-357.

Fenneman, N.M. 1938. Physiography of eastern United States. New York: McCraw Hill. 714 p.

Jenkins, C.J.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A., 2003. National scale biomass estimators for United States tree species. Forest Science. 49: 12-35.

Kramer, P.J.; Boyer, J.S. 1995. Water Relations of Plants and Soil. New York: Academic Press. 495 p.

Kriedemann, P.E.; Barrs, H.D. 1983. Photosynthetic adaptation to water stress implications for drought resistance. In: Raper and Kramer, eds. Water and temperature stresses in humid temperate climates. Boulder, Colorado: Westview Press. 374 p.

Martin, J.G.; Bolstad, P.V. 2005. Annual soil respiration in broadleaf forests of northern Wisconsin: influence of moisture and site biological, chemical and physical characteristics. Biogeochemistry. 73: 149-182.

Monclus, R.; Dreyer, E.; Delmotte, F.M. [and others]. 2005. Productivity, leaf traits and carbon isotopic discrimination in 29 *Populus deltoids* X *P. nigra* clones. New Phytology. 167: 53-62.

Raich. J.W.; Tufekcioglu, A. 2000. Vegetation and soil respiration: correlations and controls. Biogeochemistry. 48: 71-90.

Reiners, W.A. 1968. Carbon dioxide evolution from the floor of three Minnesota forests. Ecology. 49(3): 471-483.

Rhoades, R.W. 1995. Succession in mature oak forest in southwest Virginia. Castanea. 60: 98-106.

Sandford, A. P.; Jarvis,P.G. 1986. Stomatal responses to humidity in selected conifers. Tree Physiology. 2: 293-297.

Sands, R.; Mulligan, D.R. 1990. Water and nutrient dynamics and tree growth. Forest Ecology and Management. 30: 91-111.

Sands, R.; Nambiar, E.K.S. 1984. Water Relation of radiate pine in competition with weeds. Canadian Journal of Forest Resources. 14: 233-237.

Smith, W.K. 1980. Importance of aerodynamic resistance to water use efficiency in three conifers under field conditions. Plant Physiology. 65: 132-135.

Smith, T.M.; Smith, R.L. 2012. Element of Ecology. Pearson l education. San Francisco, CA: Benjamin Cummings. 615 p.

Templeton, B.S.; Seiler, J.R.; Peterson, J.A.; Tyree, M.C. 2015. Environmental and stand Management influences on soil CO_2 efflux across the range of loblolly pine. Forest Ecology and Management. 355: 15-23.

Ter-Mikaelian, M.T.; Korzukhin, M.D. 1997. Biomass equations for sixty five North American tree species. Forest Ecology and Management. 97: 1-24.

Turner, M.G.; Pearson, S.M.; P. Bolstad, P.; Wear, D.N. 2003. Effects of land-cover change on spatial pattern of forest communities in the Southern Appalachian Mountains (USA). Landscape Ecology. 18: 449-464.

Tyree, M.; Seiler J. R.; Maier. C. 2014. Contrasting genotypes, soil amendments, and their interactive effects on short-term total soil CO_2 efflux in a 3-year-old *Pinus taeda* L. plantation. Soil Biology and Biochemistry. 69: 93-100.

Whittaker, H.R. 1956. Vegetation of the Great Smoky Mountains. Ecological Monographs. 26: 1- 80.

Wildung, R.E.; Garland, T.R.; Buschbom, R.L. 1975. The interdependent effects of soil temperature and water content on soil respiration rate and plant root decomposition in arid grassland soils. Soil Biology and Biochemistry. 7: 373-378.

Witkamp, M. 1966. Decomposition of litter in relation to the environment, microflora and microbial respiration. Ecology. 49: 194 -201.

Pine Regeneration

Moderator:

Tom Fox
Virginia Tech University

GROWTH AND PERFORMANCE OF LOBLOLLY PINE GENETIC PLANTING STOCK THROUGH EIGHT YEARS

Randall J. Rousseau, Scott D. Roberts, and Billy L. Herrin[1]

Abstract—Currently, the need in the pine market is to develop higher sawtimber quality trees. The pine biomass and pulpwood market supports the low end of the product chain. However, we must improve on the quality of the southern pine for construction lumber if the southern region is expected to capture the shortfall of the sawtimber market expected in the future. Various pine genetic types ranging from open-pollinated to varietal seedlings must be closely evaluated for quality characteristics as to determine the best genetic material for regeneration. In 2007, a test of mass control-pollinated (MCP) and 2nd-generation open-pollinated seedlings (OP) was planted on a retired pasture site in north Mississippi. Measurements were taken annually from age one to age six and then again at age eight. The performance and quality of the MCP and OP seedlings will be discussed as to the recommendations for landowners in north Mississippi. In addition, the varietal portion of the study, which is a mix of different varietals, will be compared to both the performance of the MCP and OP seedlings.

INTRODUCTION

The landscape of southern pine markets and especially loblolly pine (*Pinus taeda* L.) has been shifting away from the pulp and paper industry and gradually more towards sawtimber (e.g., McKeand and others 2008). This change began in the mid to late 1990s as offshore pulp and paper mills in Brazil and other countries began production in a significant fashion capitalizing on the ability to use a single species, lower growing costs, and lower mill labor costs. The result of this shift has resulted in closure of a number of mills in southern states. The pulpwood market, mainly along the southern coastal areas, continues to be strong with the more recent increases in the European pellet market. This market takes advantage of the need for early pine thinnings that were typically needed for reducing the number of stems to allow better growth. These first thinnings in pine plantations are designed not only to reduce the number of stems per acre but also remove those stems that exhibited some obvious defect such as forking, disease, and sinuosity.

Non-Industrial Private Forest (NIPF) landowners in the South have willingly embraced pine plantation culture and have come to expect good financial returns from these plantations (e.g., McKeand and others 2003). However, the management of these plantations for the most part lacks the intensity typically practiced by the timber industry. While the NIPF landowners are willing to do site preparation work and some first-year herbaceous competition control, they generally lack knowledge in what type of seedlings they should be planting. In general, they seem to think that 2nd-generation open-pollinated planting stock is just as good as any of the alternatives that are available. One reason for this is the cheaper cost of the open-pollinated seedlings, which currently ranges from $60 to $80 per thousand and the continual practice of planting greater than 600-700 seedlings per acre. Under this strategy, the landowner would expect to pay a total of $87 to $110 per acre, with $42 to $56 per acre for seedlings and contract planting of somewhere around $45 to $53 per acre. McKeand and others 2006 indicated that landowners would definitely benefit from purchasing the best genetic seedlings available even it meant paying higher prices.

There is no doubt that the selected individuals within second-generation populations have demonstrated considerable gain over those of the first-generation. But today there are a number of different genetic seedling stock types, including both control-pollinated (mass control-pollinated or MCP) and varietals (i.e. clones). These genetic types have the specific advantage of exploiting more of the available genetic variation relative to open-pollinated seedlings. There is no doubt that more reliable information on productivity and quality is needed to allow NIPF landowners and consultants to make informed decisions on selection of available genetic planting stock (Burkhardt 2013).

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This particular study was designed with two specific goals; a comparison of a single select 2nd-generation open-pollinated family to that of a selected MCP family, and evaluation of numerous varietal seedlings as provided through ArborGen as a portion of the ArborGen Testing Service (ATS).

METHODS

The test site is located near Holly Springs, Mississippi on the North Mississippi Branch Experiment Station of Mississippi State University. The test site was previously in pasture, which consisted of primarily Bermuda grass (*Cynodon dactylon*). A Loring silt loam with a minor portion made up of a Cahaba-Providence silty-clay loam are the two soil types found on the test site. Prior to planting the site was sub-soiled to a depth of 14 inches on 12 foot centers and then treated with a 2 percent solution of glyphosate applied as a three-foot band directly over the sub-soil area. The test design is a randomized complete block consisting of six blocks, three genetic seedling types (all of which were provided by ArborGen), arranged in 100-tree block plots with the inner 64 trees forming the measurement plot. The test site was planted by hand at a spacing of 12 x 9 feet in April 2007. During the first growing season the test site was treated with the herbicide Select in an attempt to control the Bermuda grass problem and mowed later in the year. In 2008, a broadcast application of Oustar was completed in May and followed up with another application of Select in June.

The three different genetic seedling types included a single 2nd-generation open-pollinated family (OP), a mass control-pollinated (MCP) family and several varietals. Both the OP and MCP planting stock were bare-root, while the varietal was containerized planting stock. The difference between the varietal plots and the 2nd-generation open-pollinated and the mass control-pollinated seedling is that the interior 64 trees forming the varietal plot consists of 56 different varietals and eight control seedling types. All of the trees within the measurement plots were measured for height at ages one, two, three, four, six, and eight while DBH was measured at ages three, four, six, and eight. Various types of stem quality degrade, which included excessive stem sinuosity, forking, presence of fusiform rust (*Cronartium quercuum* f. sp. *fusiforme*), ramicorn branching, and crown dieback or breakage were also measured at age eight.

RESULTS

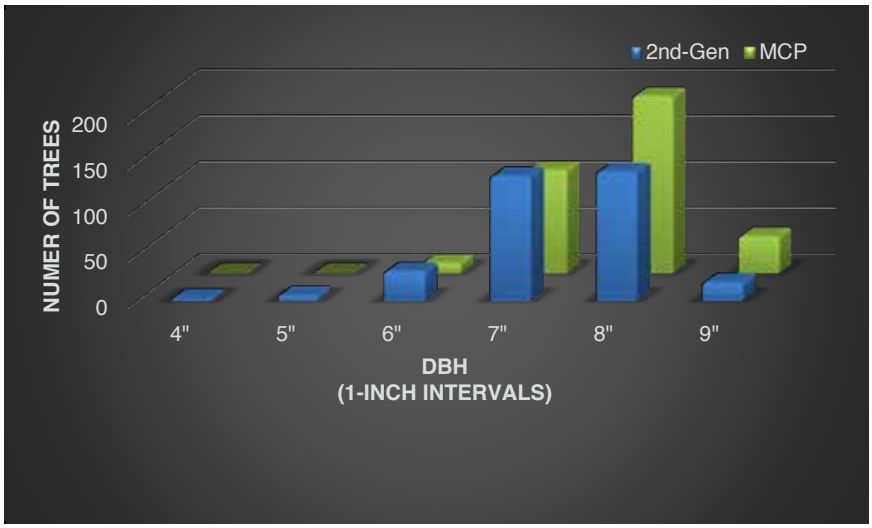
Comparison of mean survival, DBH, height, and volume at ages four, six, and eight between the 2nd-generation open-pollinated family and the mass control-pollinated family showed that in general the MCP family outperformed the OP family. Age-four survival of the OP family was higher at 95.3 percent than the MCP family at 93.2 percent. However, age-eight survival between the OP and MCP families was quite similar at 93.5 percent and 93.0 percent, respectively. Although, there were significant diameter differences between the OP and MCP families for all ages measured, the 0.03 inch difference shown at age four remained the same at age eight. The greatest difference between the OP and MCP families was exhibited in total height. However, this difference did not show up until age eight. At ages four and six, the height difference between the MCP and OP family was identical with the MCP family being 0.9 feet taller. At age eight this difference had expanded to 1.6 feet, with the MCP family exhibiting a mean height of 33.8 feet. Correspondingly, the volume performance between the MCP and the OP family revealed an increasing difference with the MCP volume increasing faster than the OP family as age increased.

Examination of the frequency of the individuals within the MCP and the OP families showed that both total height and diameter were more variable for the OP family (fig. 1). The OP family exhibited trees in the four, five, and six inch diameter class whereas the MCP family showed no individual trees below the six-inch class. In addition, the MCP had very few individuals in the six-inch diameter class. A greater percentage of individuals for both the MCP family and the OP family fell in the 31-35 foot class but the MCP family had a larger number of individuals in this class as well as in the 36-40 foot height class. The OP family had a greater number of individual trees falling into the lower height classes as compared to the MCP family (fig. 1).

Interesting differences were evident among planting blocks. In blocks one and two the OP family outperformed or was almost identical to the MCP family. However, for blocks three through six the MCP family exhibited a significant greater volume production over the OP family (fig. 2).

The variation among the 56 varietals included in the varietal plots was highly significant for height, diameter, and individual tree volume. The composite mean of all 56 varietals for all traits at all ages was lower than that of both the MCP and OP families. However, the best three varietals exhibited age-eight survival, diameter, total height, and volume greater than both MCP and OP family means (table 1).

Every tree in the test was assessed for what was termed as log degrade. This included characteristics such as stem sinuosity, poor stem form, forking, excessive limb breakage, ramicorn branching and the presence of fusiform rust. Surprisingly, both the MCP and OP families and the top three volume performing varietals all expressed approximately similar amounts of degrade (i.e. approximately 14 percent). There was a single varietal identified as 288, which exhibited a very low



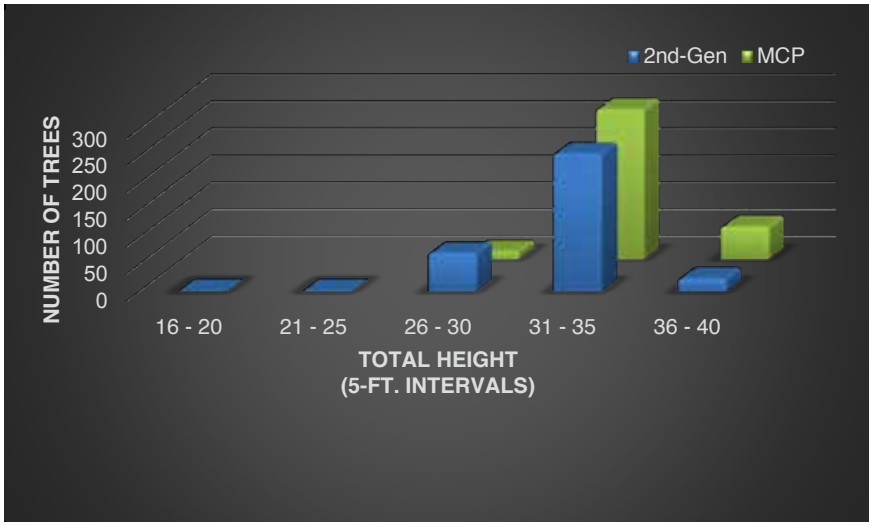


Figure 1—Distribution of trees by height and diameter class at eight years for both the MCP and OP families included in Genetic Comparison Test located near Holly Springs, MS.

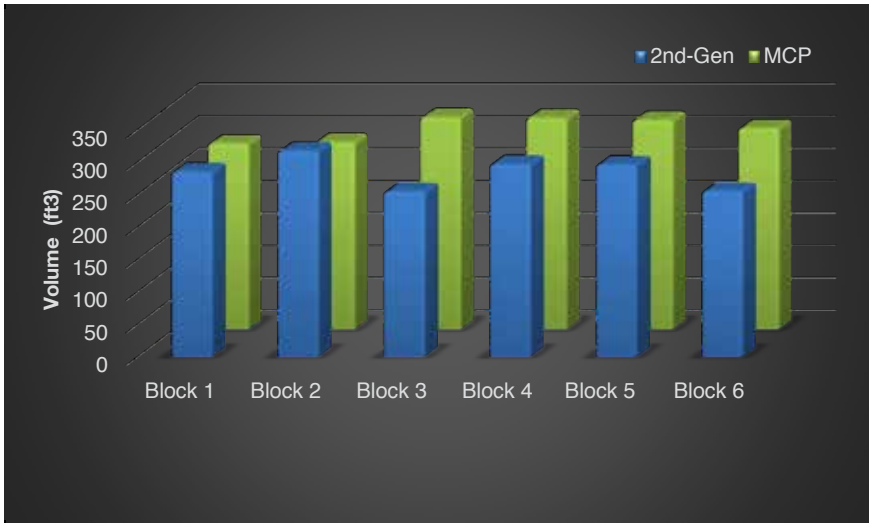


Figure 2—Age-eight total cubic foot volume differences by block for both the MCP (green) and OP (blue) in the Genetic Comparison Test located near Holly Springs, MS.

Table 1—Comparison of the three top performing varietals to the MCP and OP family means at eight years for the Genetic Comparison Test located near Holly Springs, MS

Genetic Stock Type	Survival (%)	DBH (in)	Height (ft)	Volume (ft^3)
Varietal 567	100	8.0	33.9	5.71
Varietal 586	100	8.1	33.3	5.66
Varietal 484	100	7.9	34.1	5.58
2^{nd}-Gen OP	93.5	7.5	32.2	4.78
MCP	93.0	7.8	33.8	5.31

percent degrade combined with good age-eight growth characteristics for diameter, height, and volume at 7.5 in., 33.6 ft. and 4.94 ft^3, respectively.

DISCUSSION

Interest has continued to grow among NIPF landowners to determine if higher quality genetic seedlings will actually lead to increased revenue. While this test is limited due to the inclusion of only a single family of both mass control-pollinated and open-pollinated 2^{nd}-generation family, there is no doubt that the MCP family performed better than the OP family. The MCP family exhibit greater uniformity than the OP family as shown in the diameter and height frequency distribution, where all of the MCP trees were grouped into the larger classes. This is to be expected, since full-sibs (MCP) possess a greater amount of additive genetic variation over open-pollinated (OP) individuals. While some non-additive genetic variance is also expected this has been suggested to be very low (Jansson and Li 2004). The rationale for this was that the selection of both parents was based on high general combining ability (GCA). Other studies have also noted that gains in productivity, fusiform resistance, and stem straightness have been remarkable when both male and female parents have been carefully selected (e.g., McKeand and others 2008). What was extremely interesting was the response that was observed between the MCP and OP families across the six blocks. Blocks one and two were extremely heavy to Bermuda grass and the soil more compacted from traffic. Subsequently the performance of all of the trees suffered, no matter the genetic type. In these two blocks the growth performance between the MCP and OP families was much more similar than in the other four blocks. This seems to indicate that the MCP family has the capability of expressing it's inherit growth potential better than the OP family as site index increases (e.g., McKeand and others 2006).

The performance of the varietal plot is understandable as this is was really designed to be a portion of a much larger varietal trial by ArborGen to determine possible selections for commercial release. When examined at this test site alone, selections can be made but stand performance is unknown since only six individuals per varietal were evaluated. In this test, three varietals demonstrated excellent survival and growth when compared to the mean of the OP family. However, the age-eight performance of these three varietals was very similar to that of the MCP family. The mean diameter of these three varietals was the biggest difference which also affected volume.

From this specific data, the question remains as to whether varietals are worth the extra cost compared to either OP or MCP seedlings. But, before that can be truly answered, there is a need to include quality characteristics that will provide added value to the landowner. As mentioned earlier both the MCP and OP families graded out very similar in relationship to quality. However, one specific varietal (i.e. 288) demonstrated excellent quality characteristics. This specific genotype was superior to the OP family in terms of growth and quality while also exhibiting quality characteristics superior to the MCP family. While overall volume performance is lower than the MCP family the quality of this varietal will override this aspect when the focus is more on producing quality (e.g., Cumbie and others 2013). The next step to examine is the performance of this varietal when it competes with itself when evaluated in stand conditions.

CONCLUSION

After eight years, the MCP family has outperformed the OP 2^{nd}-generation family in diameter, height, and volume. The MCP family was also shown to be more uniform than the OP family, with the majority of the MCP trees being found in the taller height classes as well as the greater diameter classes. In addition, the MCP family expressed greater growth potential in those blocks where herbaceous competition was controlled to a greater extent. Unfortunately, both families showed

very similar percentage of quality characteristics that would provide even greater value for the MCP family. Careful selection of an MCP family that expresses both excellent growth and quality characteristics would yield greater revenue over the OP family and most of the varietals evaluated within this test.

LITERATURE CITED

Burkhart, H.E. 2013. Modeling genetic improvement effects on growth and value of forest plantations. In: Cummingham, M., ed. Proceedings of the 32nd southern forest tree improvement conference. Clemson, SC: Clemson University. http://www.sftic.org. 113 p.

Cumbie, W.P.; De La Torre, R.; Cunningham, M; Pait, J. 2013. Will advanced loblolly pine genetics deliver added value to landowners in the Southeastern United States. In: Cummingham, M., ed. Proceedings of the 32nd southern forest tree improvement conference. Clemson, SC: Clemson University. 113 p. http://www.sftic.org.

Jansson, G.; B. Li. 2004. Genetic gains of full-sib families from disconnected diallels in loblolly pine. Silvae Genetica. 53(2): 60-64.

McKeand, S.E.; Mullin, T.J.; Byram, T.D.; White, T. 2003. Deployment of genetically improved loblolly and slash pine in the south. Southern Journal of Applied Forestry. 101(3): 32-37.

McKeand, S.E.; Abt, R.C.; Allen, H.L.; Li, B.; Catts, G.P. 2006. What are the best loblolly pine genotypes worth to landowners. Journal of Forestry. 104(7): 352-358.

McKeand, S.E.; Gerwig, D.M.; Cumbie, W.P.; Jett, J.B. 2008. Seed orchard management strategies for deployment of intensively selected loblolly pine families in the southern US. In: Lindgren, D., ed. Proceedings of a seed orchard conference. Umea, Sweden: 177-182.

SCREENING *PINUS TAEDA* (LOBLOLLY PINE) FAMILIES FOR PHYSICAL AND MECHANICAL PROPERTIES USING VIBRATIONAL SPECTROSCOPY

Gifty E. Acquah, Brian K. Via, Lori G. Eckhardt, and Oladiran O. Fasina[1]

Abstract—In a bid to control the loblolly pine decline complex, stakeholders are using the selection and deployment of genetically superior families that are disease tolerant. It is vital that we do not compromise other important properties while breeding for disease tolerance. In this preliminary study, near infrared spectroscopy was utilized in conjunction with data collected via conventional methods to develop partial least squares regression models that were used to rapidly predict the basic density, stiffness and ultimate strength of 14 genetically superior loblolly pine families that have been selectively bred to be disease tolerant. Calibration and independent validation data were from southern pines acquired from a commercial sawmill. Seven or eight latent variables were used in model development. The coefficients of determination of the predictive models were more promising for MOE (0.58) and MOR (0.42).

INTRODUCTION

Pinus taeda (loblolly pine) is the most economically important tree species in the USA. With 30 million acres in plantations in the southern US alone, it provides 110,000 jobs and contributes approximately $30 billion to the economy of this region. However, over the past 50 years, reduced growth, decline and eventual mortality have been associated with loblolly pine trees (Eckhardt and others 2010). In a bid to control this decline complex, stakeholders are using the selection and deployment of genetically superior families that are disease tolerant. It is vital that we do not compromise other important properties while breeding for disease tolerance, and the only way to determine this is to measure them. However, with the large number of trees involved in tree breeding programs, it will not be feasible to determine these properties with the conventional methods that require considerable sample preparation thus time consuming, expensive and mostly destructive. There is therefore the need for alternative analytical tools that have high throughput and are cost effective.

In this preliminary study, near infrared spectroscopy (NIRS) coupled with multivariate data analysis was used to characterize and then rapidly predict the basic density, modulus of elasticity (MOE) and modulus of rupture (MOR) of loblolly pine families that have been selectively bred to be disease tolerant. These properties are important because any changes in them will impact the yield and/or quality of final products – whether it be pulp, paper, lumber or engineered wood product. Furthermore, it is essential that strength is not compromised else mortality due to reasons other than forest disease such as wind failure could occur.

MATERIALS

Fourteen year old loblolly pine trees were harvested from two forest sites. Seven families were harvested from Yulee FL and seven from Waycross GA in February and March of 2014 respectively. The diameter at breast height (DBH) of these trees ranged from 12.4 cm to 21.6 cm. Trees were crosscut into 1.5 m bolts along the bole, then 50 cm 'disks' were taken from the mid portions for further processing into the final test specimen. Nominal 2 x 4-in southern pine boards were also acquired from West Fraser Inc., a commercial sawmill located in Opelika AL.

METHODS

Conventional Testing

For mechanical testing, the three-point bending test as specified in ASTM D143 was employed. Test specimens cut to final dimensions of 2.5 x 2.5 x 41 cm were conditioned to an average moisture content (MC) of about 9 percent in a control chamber (temp: 22 °C; relative humidity: 55 percent) before testing. Samples were loaded into a Zwick-Roell load frame equipped with 10KN load cell and a computer controlled screw-drive crosshead, and force applied at 1.3 mm/min on the tangential face. The MOE (i.e. Stiffness) was computed as the slope of the linear portion of the

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load-deflection curve. MOR (i.e. Ultimate strength) was calculated as

$$\frac{a \cdot Pmax}{(b \cdot h^3/12)}$$

where

a: Distance between loading point and nearest support
Pmax: Maximum load
b: Width
h: Height

Basic density was determined as the ratio of the mass of a test specimen to its volume.

Near Infrared Spectroscopy (NIRS)

The infrared region is the wavelength range of 780 nm – 1 mm (wave number range 12820 – 10 cm^{-1}) that lies between the visible and microwaves regions of the electromagnetic spectrum. NIRS measures the amount of near infrared light a sample absorbs, transmits or reflects based on its chemical composition. Spectra was collected from samples ground to pass an 80-mesh screen (Jiang and others 2014) with a PerkinElmer Spectrum Model 400 NIR spectrometer. A sample was scanned thirty-two times at a resolution of 4 cm^{-1} and averaged into one spectrum for analysis.

Multivariate Data Analysis – Model Calibration and Validation

PerkinElmer Spectrum Quant+ software was used to develop Partial Least Squares regression (PLS) models. The PLS1 algorithm, utilized the raw untreated NIR spectra (i.e. absorption of near infrared light at different wavelengths) as the independent (X) variables and regressed them on the results acquired from the conventional testing methods (i.e. dependent – Y variables). Thirty-one samples from the southern pines stock was used for calibration. An additional thirteen samples from this material that were not part of the calibration set were used in independent validation. Several statistics including standard error of calibration (SEC), standard error of prediction (SEP), coefficient of determination (R^2) and residual predictive deviation / ratio of performance to deviation (RPD) were used to evaluate the performance of our models. Models that had the least error values were selected and used to predict the basic density, MOE and MOR of the fourteen genetically superior loblolly pine families.

RESULTS AND DISCUSSION

NIR Spectra

Raw NIR spectra characteristic of loblolly pine used for model development is shown in figure 1. Spectra highlights some important peaks that have been assigned to specific chemical components of wood. For instance, the peak occurring at 4080 cm^{-1} is known to result from cellulose, while that at 6945 cm^{-1} has been attributed to the presence of lignin and extractives (Schwanninger and others 2011). These chemical constituents are the basis for the application of NIRS in the prediction of density and strength of wood – wood chemistry have been shown to have strong correlations with these non-chemical properties. Cellulose has a strong relationship with density, which in turn has an influence on the stiffness and ultimate strength of wood (Via and others 2003).

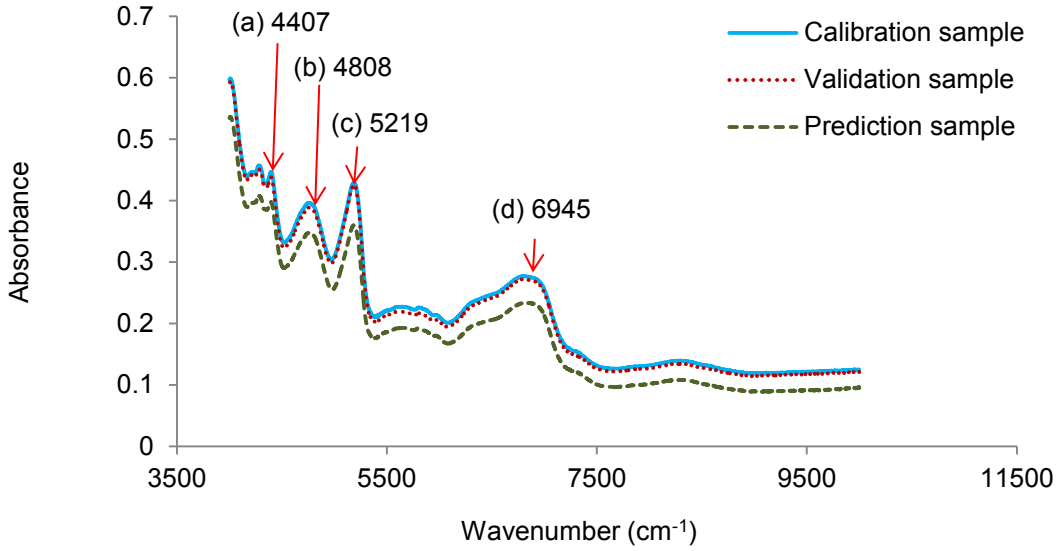


Figure 1—Characteristic NIR spectra of loblolly pine wood showing some important wavenumbers. (a) - cellulose and hemicelluloses; (b) - cellulose; (c) - water; (d) - lignin and extractives.

Table 1—Conventional lab results used for model development and independent prediction

	Basic Density (g/cm^3)	*MOE (MPa)*	*MOR (MPa)*
Calibration set (n = 31)			
Min	0.43	5800	46.2
Max	0.69	15100	125.0
Average	0.56	10100	83.4
SD	0.07	2500	17.3
Validation set (n = 13)			
Min	0.47	6200	41.2
Max	0.62	13900	110.0
Average	0.44	9400	77.6
SD	0.03	2100	17.8
Prediction set (n = 14)			
Min	0.51	6010	40.3
Max	0.65	15100	130.0
Average	0.56	10091	97.4
SD	0.04	2601	40.3

Conventional Lab Results

The density, MOE, and MOR of loblolly pine measured using the conventional methods are presented in table 1. Average density was 0.56 cm^{-3} for the calibration set and 0.51 cm^{-3} for the independent test set. Stiffness for all samples used in this study ranged from a low of 5800 MPa to a high of 15100 MPa. The calibration set was slightly stiffer than the validation set. For ultimate strength, the range of the calibration set was wider than the range of the validation set -- i.e. 78.8 MPa and 68.8 MPa respectively.

Although means of the calibration and validation sets had good overlap which help with model performance, the range of especially the mechanical properties were not as wide as seen in the literature. MOE have been reported to be as low as 2200 MPa, and as high as 26000 MPa (Kelley and others 2004, So and others 2002).

PLS Model Performance

Fit statistics showing how models performed are presented in table 2. Seven or eight latent variables (LVs) were used in developing PLS models. LVs are linear combinations of the x and y variables extracted in such a way that variation in both response and regressors are optimally explained. The proportion of variance accounted for by the first LV is the maximum, and this progressively decreases as successive LVs are extracted. SEC was used to evaluate how precisely the regression line fitted the data, while SEP measured the precision of a model's predicting ability during validation. The closer these errors are to zero, the better the model. A small difference between calibration and validation errors is an indication that the test sets were well predicted by calibration models. Ideally, the SEP of a model should not be more than 1.3 times its SEC (Acquah and others 2015). Only the models for predicting stiffness and ultimate strength passed this criteria. The SEC and SEP were respectively 1800 MPa and 1900 MPa for MOE, and 13.5 MPa and 17.4 MPa for MOR.

R^2 is a measure the total variance between measured and predicted that can be modeled linearly. Usually, R^2 keeps increasing as more LVs are added in the development of a model. To ensure models were not overfitted, we computed the adjusted R^2, a measure that penalizes if insignificant LVs are added to a model just so the R^2 get inflated. R2adj values ranged from a low of 0.51 for MOR, to a high of 0.64 for basic density. Furthermore, another statistic, the RPD was used to evaluate the SEP in terms of SD of the reference data (i.e. y-variables). In order for a model to predict a parameter with precision, the SEP should be considerably lower than SD. RPD values for all three models were less than 1.5. For a model to be

Table 2—Fit statistics of PLS regression models using raw NIR spectra

Property	LVs	SEC	SEP	SD	RPD	R^2	R^2_{adj}	R^2_{pred}
Basic Density (g/cm^3)	8	0.05	0.08	0.07	0.9	0.65	0.64	0.19
Modulus of Elasticity (MPa)	8	1800	1900	2500	1.3	0.63	0.62	0.58
Modulus of Rupture (MPa)	7	13.5	17.4	17.3	1.0	0.53	0.51	0.42

applicable for preliminary predictions in tree selection for improvement programs, this threshold should be exceeded (Hein and others 2009).

Although the model developed for basic density had the highest adjusted R^2, its RPD was less than 1. Thus as expected, it performed worst when it was applied in parameter prediction for the fourteen genetically superior loblolly pine families. Even though the models for stiffness and ultimate strength were not the best, their predictive ability was promising when applied to the loblolly pine families that were not included in model calibration, table 2.

CONCLUSIONS

In this preliminary study, PLS regression models were developed with NIR spectra of southern pines and reference data acquired through standard methods. These models were used to rapidly predict the basic density, Modulus of Elasticity (Stiffness) and Modulus of Rupture (Ultimate strength) of fourteen *Pinus taeda* (loblolly pine) families that have been selectively bred to be disease tolerant.

Although the number of LVs used in model development was relatively high, the similar R^2 and adjusted R^2 values was an indication that models were not overfitted. RPD values for models developed in this study were however low. Inspite of this, the linear relationship between measured and NIR predicted parameter were promising for the loblolly pine families that were not included in model calibration. (i.e. MOE = 0.58; MOR = 0.42).

Work is ongoing to increase the precision of the predictive ablity of these models. At the completion of this project, models should be able to rapidly predict not only the studied properties of these 14 families, but also other genetically superior loblolly pine families that are currently being deployed by stakeholders. The long term goal of this study is to make the right feedstocks available for the conventional forest industry, as well as to support the emerging bioeconomy.

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LITERATURE CITED

Acquah, G.E.; Via, B.K.; Fasina, O.O.; Eckhardt, L.G. [In press]. Nondestructive prediction of the properties of forest biomass for bio energy, fuel and chemical applications using near infrared spectroscopy (NIRS). Journal of Near Infrared Spectroscopy.

Eckhardt, L.; Sword Sayer M.A.; Imm, D. 2010. State of pine decline in the southeastern United States. Southern Journal of Applied Forestry. 34: 138-141.

Hein, P.R.G.; Campos, A.C.M.; Trugilho, P.F. [and others]. 2009. Near infrared spectroscopy for estimating wood basic density in *Eucalyptus uroplylla* and *Eucalyptus grandis*. Cerne, Lavras. 15: 133-141.

Jiang, W.; Han, G.; Via, B.K. [and others]. 2014. Rapid assessment of coniferous biomass lignin-carbohydrates with near-infrared spectroscopy. Journal of Wood Science and Technology. 48: 109-122.

Kelley, S.S.; Rials, T.G.; Snell, R. [and others]. 2004. Use of near infrared spectroscopy to measure the chemical and mechanical properties of solid wood. Journal of Wood Science and Technology. 38: 257-276.

Schwanninger, M.; Rogrigues, J.; Fackler, J. 2011. A review of band assignments in near infrared spectra of wood and wood components. Journal of Near Infrared Spectroscopy. 19: 287-308.

So C-L.; Groom, L.H.; Rials, T.G. [and others]. 2002. Rapid assessment of the fundamental property variation of wood. In: Outcalt, K.W., ed. Proceedings of the 11th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-48. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 176-180.

Via, B.K.; Shupe, T.; Groom, L. [and others]. 2003. Multivariate modeling of density, strength and stiffness from near infrared spectra for mature, juvenile and pith wood of longleaf pine (*Pinus palustris*). Journal of Near Infrared Spectroscopy. 11: 365-378.

EVALUATING FIRST-YEAR PINE SEEDLING SURVIVAL PLATEAU IN LOUISIANA

Puskar N. Khana, Thomas J. Dean, Scott. D. Roberts, and Donald L. Grebner[1]

Abstract—First-year seeding survival has been a continuing problem since the start of commercial pine plantation forestry in the 1950s. First-year survival of bare-root loblolly pine seedlings on intensively prepared sites in Louisiana has maintained a survival plateau between 79 to 89 percent with an average of about 82 percent. The specific objectives of this study were to identify seedling and microsite quality distributions in a current plantation, and evaluate a conceptual model explaining the plateau in first-year seedling survival. The study was approached with a conceptual model simulation using hypothetical data followed by model evaluation using field data. Simulation results indicated that consistent survival could result from random pairing of initial seedling and site quality distributions. Analysis of data collected from seedlings obtained from the Louisiana Department of Agriculture and Forestry (LDAF) indicated that 58 percent of seedlings were associated with the most frequent quality class that comprised seedlings with volume between 2.64 to 5.26 cm^3 and average caliper and stem height 4.22 mm and 25.75 cm, respectively. Similarly, assessment of microsites at Weyerhaeuser planting sites in Louisiana indicated that 51 percent of planted seedlings were associated with the most frequent microsite quality class which supported less than 10 cm first-year height increment. Model evaluation from seedling and microsite quality distributions in current operational practice indicated that using larger than 5 mm caliper size would increase first year survival to above 90 percent. This would, however, result in higher establishment costs, so the preference of this strategy would largely depend on management goals of the owner.

INTRODUCTION

First-year seedling survival is a very important stage in plantation forestry. First-year seedling survival of loblolly pine was a major issue until the 1970s, but improvements in nursery techniques and site preparation practices, as well as seedling genetics and physiology, resulted in first-year seedling survival in pine plantations generally exceeding 80 percent (Fox and others 2007). In research settings, however, the survival rate is commonly above 95 percent due in part to the controlled nature of the experiments. Interestingly, first-year survival has maintained a rate between 80- 90 percent in operational environment (South and others 2001). Descriptive statistics of first-year seedling survival data from Louisiana Department of Agriculture and Forestry (LDAF) between 1998 and 2007 indicated that survival success has a plateau between 80-87 percent under normal weather conditions. Although these survival rates are good, the inability to have these rates to be consistently in the 90the percentile is of concern. Why are survival rates not consistently higher in operational plantation? This study attempts to provide an explanation for the plateau in seedling survival based on an analysis of seedling and microsite qualities involved in operational plantations.

The specific objectives of this study are to identify seedling and microsite quality distributions in a current plantation, and evaluate a conceptual model explaining the current level of first-year seedling survival.

CONCEPTUAL MODEL

The conceptual model consists of two components. A joint distribution of seedling and microsite quality and a probability function of first-year survival that uses the joint distribution as input. Integration of the probability function gives the first-year survival rate. For a discrete distribution, the model can be explained by the following four steps. First, quality distributions for both the seedlings and the microsites are needed. Table 1 presents a hypothetical case with assumed seedling and microsite quality distributions using five quality classes. Quality class values for both seedlings and microsites range from one which indicates the highest quality to five which indicates the lowest quality. In table 2, we calculate a joint frequency distribution from the random pairing of seedlings and microsites given the quality distributions. The random reciprocal planting of each seedling and microsite results in a five by five joint frequency table. A probability rule that provides proportional survival of each pairing of

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Table 1—Seedling and Microsite quality distribution for conceptual model. Quality class 1 indicates lowest value while 5 indicates the best value. The values in seedling and microsite column are relative frequencies in respective quality class

Quality class	Seedling	Microsite
5	0.10	0.10
4	0.20	0.15
3	0.35	0.35
2	0.25	0.20
1	0.10	0.20

Table 2—Joint frequency distribution of seedling and microsite quality. Quality class 1 indicates lowest value while 5 indicates the best value. The cell values in each seedling and microsite quality pairing are relative frequencies

Seeding quality	Microsite quality				
	5	4	3	2	1
5	0.01	0.02	0.03	0.02	0.01
4	0.01	0.03	0.05	0.03	0.01
3	0.03	0.07	0.12	0.08	0.03
2	0.02	0.04	0.07	0.05	0.02
1	0.02	0.04	0.07	0.05	0.02

seedling and microsite is also needed and is placed in a joint frequency table. Our hypothetical probability rule is illustrated in table 3. The underlying presumption for the rule is that higher quality seedlings associated with higher quality microsites will result in higher survival as compared to lower quality pairings. Multiplication of the corresponding cells in tables 2 and 3 results in the predicted first-year survival for each seedling-microsite pair (table 4). The sum of values under each cell in table 4 provides an estimate of first-year survival for this plantation. For this example, the sum of values under each cell is 0.51 which indicates first-year survival of this plantation is 51 percent. Following these simulation steps, survival percentage could be computed for other seedling and microsite quality distributions.

Using this approach, we simulated first-years survival of 1600 plantings using random reciprocal pairing of 20 seedling quality distributions and 20 microsite quality distributions. The term plantings refer to individual seedlings planted across sites. Predicted first-year survival values varied widely between 0 to 100 percent depending on initial seedling and microsite quality distributions. Plantings associated with distributions containing higher proportions of seedlings and microsites in the better quality classes obviously resulted in higher survival compared to those with a higher proportion of seedlings and microsites in the lower quality classes. This provides initial support for the concept that our nursery and site preparation practices might be providing similar quality distributions of seedlings and microsites every year to create a plateau in first-year seedling survival.

FIELD TEST

Data

The field portion was an attempt to quantify actual quality distributions of seedlings and microsites and randomly combine these pairings to determine how sensitive survival may be to random combinations of the relative frequency of seedling and microsite qualities. Seedlings and microsites commonly used in operational plantations were used to estimate their quality distributions. Seedling data were obtained using a 1000-seedling bag of loblolly pine seedlings purchased from the Louisiana Department of Agriculture and Forestry (LDAF). These seedlings were measured for stem caliper and height. Similarly, microsite data were collected from measurements of microsite variables

Table 3—Expected survival assumption table. Quality class 1 indicates lowest value while 5 indicates the best value. Cell value 0 indicating no survival while 1indicating 100% survival in respective pairing of seedling and microsite qualities

Seedling quality	Microsite quality				
	5	4	3	2	1
5	0	0	0	0	0
4	0	0	0	0	0
3	0	0	1	1	1
2	0	0	1	1	1
1	0	0	1	1	1

Table 4—Predicted first-year survival table. Quality class 1 indicates lowest value while 5 indicates the best value. The cell values in each seedling and microsite quality pairing are relative frequencies. Sum of all cells gives predicted first-year survival

Seedling quality	Microsite quality				
	5	4	3	2	1
5	0	0	0	0	0
4	0	0	0	0	0
3	0	0	0.12	0.08	0.03
2	0	0	0.07	0.05	0.02
1	0	0	0.07	0.05	0.02

at 8 selected planting sites of Weyerhaeuser Inc. in Livingston, Louisiana. At each of those planting sites, we randomly identified 1600 seedlings and assessed their microsite environment. The selected seedlings whose microsites were assessed were measured in spring 2008 and 2009 to record their height increment. In addition, microsite variables such as bed height and soil compaction were recorded at each seedling location. In addition, 160 soil samples were collected to identify soil texture, moisture content, and mineral nitrogen content for the selected seedlings.

Seedling and Microsite Quality

To identify the quality distribution of LDAF nursery seedlings, we computed the volume of 907 measurable seedlings contained in the bag of nursery seedlings using a volume formula that used caliper and stem length measurements for the calculation. Volume was used as an index of seeding quality because it included both stem caliper and height to characterize seedling quality. The range of volumes was divided into 5 equal quality classes and the relative frequency in each class was determined. Abbreviation ‘S’ was used to indicate seedlings and quality class value of 1 indicated the highest quality while a 5 indicated the lowest quality. This provided an estimated quality distribution for nursery seedlings in current operational plantations in Louisiana in which LDAF seedlings are used. To characterize the microsite quality distribution, the predicted first-year height increment of seedlings was used as a quality index. The height increments were predicted from a reduced regression model with microsite variables (bed height, soil penetration, texture, and mineral nitrogen) as dependent variables against first-year height increment. Thus, microsite quality was not based on actual seedling height increment, but rather on the modeled relationship between observed height increment and a suite of microsite variables. Then, the range of predicted height increments were divided into 5 equal quality classes and the relative frequency in each class was computed to determine the microsite quality distribution. Abbreviation ‘M’ was

used to indicate microsites and quality class value of 1 indicated the highest quality while a 5 indicated the lowest quality.

Evaluation of Conceptual Model

In order to predict First-year survival from the simulated planting of LDAF nursery seedlings at a typical Weyerhaeuser site-prepared plantation, we followed the steps described in the conceptual model. In the first step, we computed the joint frequency distribution from seedling and microsite quality pairings. Since the outcome of planting seedlings of known quality at microsites of known quality could not be conducted in this study, first-year survival prediction required assumptions about proportional survival of each seedling and microsite quality combination. We then developed a proportional survival assumption that provided an average and a range of first-year survival as observed in Louisiana. Furthermore, strategies to increase average first-year survival by improving seedling and microsite quality distribution were evaluated.

RESULTS AND DISCUSSION

Seedling and microsite quality distribution results are presented in table 5. Nursery seedlings supplied from LDAF had stem calipers between 3.22 mm to 7.85 mm while stem height varied between 23.18 cm to 29.25 cm. The lowest seedling quality class, S5, had seedlings with less than 2.63cm^3 while the seedlings in highest quality class, S1, had volume greater than 10.53 cm^3.The most frequent quality class was S4 which comprised 58 percent of the seedlings with volume between 2.64 to 5.26 cm^3 and their average caliper and stem height were 4.22 mm and 25.75 cm, respectively. Similarly, M4 was the most frequent microsite quality class that supported upto10.37 cm height increment and it comprised 51 percent of the measured microsites (table 6). The highest microsite quality class M1 that supported greater than 31.11 cm of predicted first-year height growth contained only 2 percent of the measured microsites.

To predict first-year survival for the simulated planting of LDAF nursery seedlings at a typical Weyerhaeuser

Table 5—Seedling quality classes, relative frequencies and their associated average caliper and stem size (N=907). Seedling class S5 indicating lowest quality while S1 indicating best quality

Seedling quality class	Relative frequency	Volume (cm^3)	Average caliper (mm)	Average stem height (cm)
S5	0.31	< 2.63	3.22	23.18
S4	0.58	2.64 - 5.26	4.22	25.75
S3	0.09	5.27 - 7.89	5.35	27.29
S2	0.02	7.90 - 10.52	6.28	28.19
S1	0.01	> 10.53	7.85	29.25

Table 6—Microsite quality classes, relative frequencies and their associated predicted height increment (N=1600). Microsite class S5 indicating lowest quality while M1 indicating best quality

Microsite quality class	Relative frequency	Predicted Height increment range (cm)
M5	0.29	<= 0
M4	0.51	0 – 10.37
M3	0.09	10.37 – 20.74
M2	0.09	20.74 – 31.11
M1	0.02	> 31.11

Table 7—Expected survival assumption table. Quality class 1 indicates lowest value while 5 indicates the best value. Cell value 0.60 is indicating 60% survival while 0.99 is indicating 99% survival in respective pairing of seedling and microsite qualities

Seedling quality	Microsite quality				
	5	4	3	2	1
5	0.60	0.60	0.65	0.65	0.70
4	0.60	0.65	0.70	0.95	0.95
3	0.65	0.70	0.80	0.99	0.99
2	0.65	0.95	0.99	0.99	0.99
1	0.70	0.95	0.99	0.99	0.99

Table 8—Average, minimum and maximum first-year survival of plantings with given minimum caliper size seedlings involved during planting

Survival (%)	Caliper (mm)	
	>=4	>=5
Average	0.90	0.94
Minimum	0.79	0.85
Maximum	0.95	0.98

planting site, we estimated proportion of surviving seedlings in each seedling-microsite quality pairing. Table 7 shows the proportional survival probabilities that resulted in the average and range of first-year survival similar to observed operational plantings. The values in table 7 were estimated following repeated applications of the model using different proportional survival assumptions. This survival assumption resulted in an average survival range from 70-80 percent with an average of 82 percent. This analysis could be expanded to infer potential strategies of increasing the average first-year seedling survival. Improving the quality of seedling and microsites in current plantations would raise the overall average of first-year survival. However, locating higher quality microsites during planting may not be a practical option due to the extra time required to locate the best sites; however, the planter could discard smaller seedlings in favor of higher quality seedlings. Table 8 presents results of a simulation with seedlings at least 4-mm caliper to increase the seeding quality. The results indicate that average survival would be 90 percent with at least a 4-mm caliper and 94 percent with at least a 5-mm caliper. This means increasing caliper size could be a promising strategy to increase first-year survival in current operational environment. South and others (2001) also found similar results related to sensitivity of seedling caliper on first-year survival. Planting only larger caliper seedlings would increase establishment costs because some seedlings would have to be discarded, or can produce seedlings with a higher caliper in the nursery by using a lower seedling density in the planting beds, but this will also increases the production cost per seedling. Therefore, a decision to use higher caliper seedlings to realize higher survival percentage would require more economic analysis.

CONCLUSION

This study highlights the influence that initial seedling quality and microsite quality have on first-year seedling survival. The simulation study showed that First-year survival is responsive to initial quality distributions and that the observed survival plateau could be the result of repeated pairing of similar quality distributions. This could be the unintentional result of standardized nursery and site preparation practices providing similar distributions of seedling and microsite quality in operational planting environments. One of the practical

strategies to increase first-year seedling survival is to increase seedling caliper size. Larger caliper seedlings would increase survival rate and growth, and ultimately yield from plantations.

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LITERATURE REVIEW

Fox, T.R.; Jokela, E.J.; Allen, H.H. [and others]. 2007. Tree nutrition and forest fertilization of pine plantations in the Southern United States. Southern Journal of Applied Forestry. 31(1): 5-11.

South, D.B.; Rakestraw, J.L.; Lowerts, G.A. 2001. Early gains from planting larger-diameter seedlings and intensive management are additive for loblolly pine. New Forests. 22(1-2): 97-110.

Threats

Moderator:

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STATUS OF FUSIFORM RUST INCIDENCE IN SLASH AND LOBLOLLY PINE PLANTATIONS IN THE SOUTHEASTERN UNITED STATES

KaDonna C. Randolph[1]

Abstract—Southern pine tree improvement programs have been in operation in the southeastern United States since the 1950s. Their goal has been to improve volume growth, tree form, disease resistance, and wood quality in southern pines, particularly slash pine (*Pinus elliottii*) and loblolly pine (*P. taeda*). The disease of focus has been fusiform rust, which is caused by the fungus *Cronartium quercum* f. sp. *fusiforme*. Prior to the 1930s, fusiform rust existed at endemic levels across the region, but following changes in forest management practices during the 1950s and 1960s, the disease increased to epidemic proportions. Since the 1970s, rust-resistant planting stock has been developed and increasingly deployed throughout the southeastern United States. Analysis of data collected by the Forest Inventory and Analysis Program of the Forest Service, U.S. Department of Agriculture showed that current rust incidence is generally higher in slash pine plantations than loblolly pine plantations and that, during the past 30 to 40 years, declines in rust incidence were evident in young (i.e., 5 to 15 years old) planted loblolly pine stands but not in young planted slash pine stands.

INTRODUCTION

Timberland acreage of artificially regenerated[2] (hereafter referred to as "planted") pine stands has increased steadily in the southeastern United States (fig. 1) since 1952 (fig. 2) (Conner and Hartsell 2002). In 2012, there were over 40 million acres of planted pine (*Pinus* spp.) and other softwood timberland in the southeastern United States, 95 percent of which was in the slash pine (*P. elliottii*) and loblolly pine (*P. taeda*) forest types (Miles 2015). Growth rates in these forest types now meet or exceed 300 feet^3 acre^{-1} year^{-1}, more than double the rates of the 1950s (Fox and others 2007, McKeand and others 2003), making them some of the most productive forests in the world (McKeand and others 2003).

The significant increase in pine plantation acreage and productivity since the 1950s is largely due to improvements in silvicultural techniques (e.g., site preparation, fertilization, and weed control); nursery practices; and planting stock (Fox and others 2007). Research cooperatives in Florida, Georgia, North Carolina, and Texas, along with the Resistance Screening Center (RSC) of the Forest Service, U.S. Department of Agriculture in Asheville, NC (Cowling and Young 2013), have been instrumental in improving the volume growth, tree form, disease resistance, and wood quality of slash and loblolly pine planting stock (Allen and others 2005, Schmidt 2003). Li and others (1999) reported that seedlings from first-generation seed orchards produced volume gains of 7 to 12 percent over wild seed, and seedlings from second-generation seed orchards established in the 1980s were projected to produce an additional 14 to 23 percent gain in volume. Third-generation seedlings have been projected to increase productivity even more (Aspinwall and others 2012). The proportion of increased productivity due specifically to increased fusiform rust resistance is substantial. Gains in mean annual increment over a 25-year rotation due to increased resistance to fusiform rust have been estimated at 25 to 30 percent for slash pine and 5 to 7 percent for loblolly pine (Brawner and others 1999, Vergara and others 2007).

Fusiform Rust

Fusiform rust is an endemic disease caused by the pathogen *Cronartium quercum* f. sp. *fusiforme* that requires both pine and oak (*Quercus* spp.) species to complete its life cycle. Infections on pine trees typically result in spindle-shaped galls on the branches and stems (Phelps and Czabator 1978), although galls can be round, oval, or odd-shaped as well. Stem infections that occur before five years of age typically result in tree mortality, whereas infections on older trees create open cankers that continue to enlarge and degrade stem quality and often become points of breakage during storms (Anderson and others 1986). Although fusiform

[2]The Forest Inventory and Analysis (FIA) Program categorizes forest stands into two regeneration classes: natural and artificial. Artificially regenerated stands include stands with distinct evidence of planting or seeding. For simplicity, the term "planted" is used throughout the article to refer to the artificially regenerated stands.

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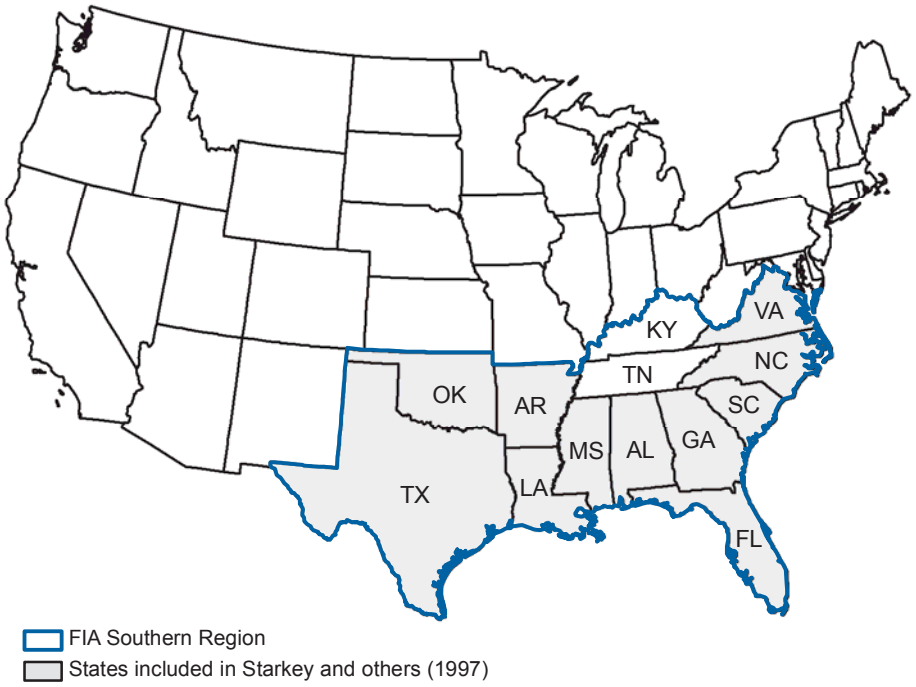


Figure 1—Fusiform rust incidence data from all 13 states of the Forest Inventory and Analysis (FIA) Program Southern Region were included in the report by Randolph and others (2015). The report by Starkey and others (1997) excluded Kentucky and Tennessee.

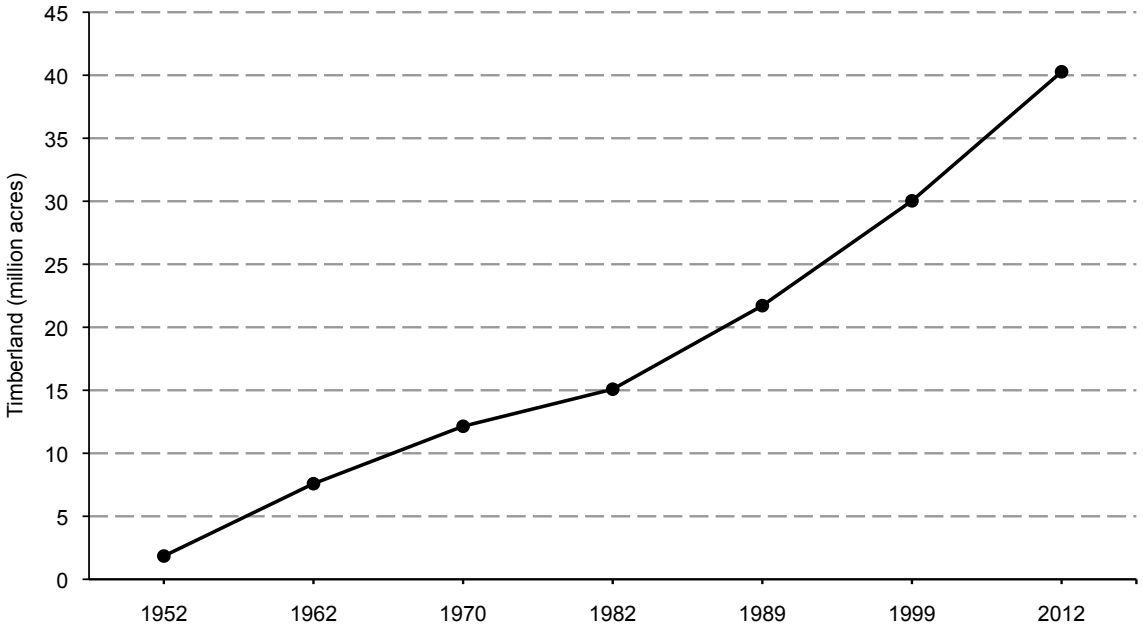


Figure 2—Timberland acreage of artificially regenerated pine (*Pinus* spp.) and other softwood forest types in the southeastern United States, excluding Kentucky, by year. Estimates for 2012 are from the FIA database (Miles 2015). All other estimates are from Table 16.8 in Conner and Hartsell (2002).

rust affects several southern pine species, slash pine and loblolly pine are the most susceptible (Phelps and Czabator 1978).

In the early 1970s, fusiform rust infection exceeded 50 percent in young pine plantations across much of the southeastern United States. Isogram charts of rust incidence in 8- to 12-year-old slash pine and loblolly pine plantations showed a ridge of high rust incidence extending from Louisiana to South Carolina for both species (Squillace 1976). Starkey and others (1997) estimated that fusiform rust infection was ≥50 percent on 8 percent of the 3.8 million slash and loblolly pine timberland acres inventoried by the Forest Service Forest Inventory and Analysis (FIA) Program in Mississippi, North Carolina, South Carolina, and Virginia in the late 1970s. By the early 1990s, ≥50 percent rust incidence was observed on only 1.2 percent of the 8.8 million slash and loblolly pine timberland acres in those same four states. Southwide (fig. 1), however, the proportion of slash pine timberland acreage with ≥10 percent rust incidence showed a slight upward trend over the same time period (Starkey and others 1997).

Objectives

As the fortieth anniversary of the establishment of the RSC approached, and 13 years after the publication by Starkey and others (1997), leaders of the university-industry tree improvement research cooperatives in Florida, North Carolina, and Texas, and the manager of the RSC, organized an information exchange working group meeting with the general theme "Integrating Fusiform Rust Research, Screening, and Breeding" (Cowling and Randolph 2013). During this meeting, several questions were raised concerning the current status of fusiform rust, including (Cowling and Randolph 2013):

1. What is the current status of the fusiform rust epidemic in slash pine and loblolly pine?
2. Has fusiform rust incidence decreased, increased, or remained the same throughout the South over the last 40 years?
3. Have the areas of high rust disease hazard changed over time?

Follow-up discussions led to the development of two papers (Cowling and Randolph 2013, Randolph and others (2015) that capitalized on four decades of FIA fusiform rust incidence data collected across the southeastern United States.

The first paper (Cowling and Randolph 2013) explored the benefits of conducting an updated analysis of the FIA data similar to the one performed by Starkey and others (1997) and included an initial analysis of fusiform rust incidence data collected by FIA between the years 2000 and 2011. During the course of the paper's writing, it was discovered that when the Enhanced FIA Program was introduced in 1999 (Bechtold and Patterson 2005), FIA data collection protocols were revised so that fusiform rust symptoms, i.e., rust galls directly on the main stem or on branches within 12 inches of the stem, would be noted only on trees ≥5 inches diameter at breast height (d.b.h.) rather than ≥1 inch d.b.h. as had been done previously. As a result, it was not possible to make current estimates of timberland area damaged by fusiform rust that would be directly comparable to those reported by Starkey and others (1997). Therefore, to address the three questions listed above, Randolph and others (2015) based their analyses on estimates of percent rust at either the state or the plot level. Results for planted slash and loblolly pine timberland were presented at the 18th Biennial Southern Silvicultural Research Conference (BSSRC) and are summarized in the sections that follow. Readers are referred to Randolph and others (2015) for similar results for natural slash and loblolly pine timberland and for a more detailed discussion overall.

METHODS

Plot-, condition-, and tree-level data from the late 1970s through 2012 were obtained from the FIA database (O'Connell and others 2010) for Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, South Carolina, Tennessee, Texas, and Virginia. Only data from timberland plots with the slash pine or loblolly pine forest type and live slash pine and loblolly pine trees ≥5 inches d.b.h. were kept in the data set. Timberland is defined as forest land capable of producing industrial wood in excess of 20 $feet^3$ $acre^{-1}$ $year^{-1}$ and not withdrawn from timber utilization (USDA Forest Service 2006).

Data from the 2010 (Texas), 2011 (Florida, Kentucky, and Tennessee) and 2012 (all other states) FIA inventories were used to estimate the current status of fusiform rust ($\hat{R}_{current}$) in each state by forest type. Likewise, data from the FIA inventories conducted in the late 1970s (Arkansas, Mississippi, North Carolina, South Carolina, and Virginia) or early 1980s (Alabama, Florida, Georgia, Louisiana, Tennessee, and Texas) were used to estimate the past status of fusiform rust ($\hat{R}_{oldest}$) in each state by forest type. The ratio of means estimator (Cochran 1977, Zarnoch and Bechtold 2000) was used to estimate $\hat{R}_{current}$ and $\hat{R}_{oldest}$ as the percentage of live trees (≥5 inches d.b.h.) with symptoms of fusiform rust. The change in rust incidence over time was calculated as the difference ($\hat{R}_{diff}$) between the oldest and most current inventory for each state and forest type. $\hat{R}_{current}$ and $\hat{R}_{oldest}$ were considered significantly different from one another (α=0.05) if the 95 percent confidence interval for $\hat{R}_{diff}$ did not include zero.

In order to map current rust hazard, plot-level percentages of rust incidence (p_i) were calculated for

each FIA plot inventoried in the 2010s. Following the methods of Starkey and others (1997), inverse distance weighted interpolation (ESRI® 2010) was applied to the p_i values to create a grid (raster) coverage of rust incidence percentages for each forest type. The resulting grid coverages were classified into categories of low hazard (0 to <10 percent infection), moderate hazard (10 to 30 percent infection), and high hazard (>30 percent infection).

RESULTS AND DISCUSSION

Current Fusiform Rust Incidence

Estimates of current fusiform rust incidence varied by state and forest type. For the slash pine forest type, the percentage of live trees (d.b.h. ≥5 inches) with symptoms of fusiform rust ranged from a low of 6.4 percent in Mississippi to a high of 21.2 percent in Georgia. For the loblolly pine forest type, the percentage of live trees (d.b.h. ≥5 inches) with symptoms of fusiform rust ranged from a low of <1 percent in Kentucky, Oklahoma, Tennessee, and Virginia to a high of 12.7 percent in Georgia. In states for which estimates were made for both forest types, rust incidence was generally higher for the slash pine forest type than for the loblolly pine forest type (fig. 3). This may be due, at least in part, to slash pine's greater susceptibility to fusiform rust infection (Zhao and Kane 2012).

Temporal Changes in Rust Incidence

Estimates of the change in fusiform rust incidence in young (5 to 15 years old) slash and loblolly pine plantations from the late 1970s or early 1980s to the 2010s showed that, in some states, rust incidence in young loblolly pine plantations is lower now than it was previously, but this was not the case for young slash pine plantations. Significant (α=0.05) declines in rust incidence in the loblolly pine forest type were observed in Alabama, Georgia, Louisiana, South Carolina, and Texas (fig. 4). In Virginia, a significant (α=0.05) increase in rust incidence in the loblolly pine forest type from 0.2 percent in 1977 to 2.0 percent in 2012 was observed. For young slash pine plantations, there was a significant (α=0.05) increase in fusiform rust incidence in Florida, whereas rust incidence remained statistically unchanged in Georgia (fig. 5). The reason or reasons for the increase in fusiform rust incidence in planted slash pine in Florida are unclear. Increased genetic gains in tree growth and improved management practices that allow rust-infected slash pine trees to survive longer now than in the past may be contributing factors (Randolph and others 2015).

The extent to which rust reductions in the young planted loblolly pine stands can be attributed directly to the planting of rust-resistant planting stock is not known with certainty because decreases in rust incidence also were observed in naturally regenerated loblolly pine stands (Randolph and others 2015). However, the decreases in fusiform rust incidence in the planted loblolly pine forest types were typically greater in the planted stands than in the natural stands (Randolph and others 2015). Thus, at least a portion of the reduction in rust incidence was likely due to the deployment of rust-resistant planting stock.

Current Rust Hazard

Despite some decreases in fusiform rust incidence over the last 30 to 40 years, rust hazard remains high throughout much of the southeastern United States, especially in areas where the historical range of slash pine and loblolly pine overlap (fig. 6). Rust hazard is highest for the slash pine forest type in Georgia, southeastern Alabama, northern Florida, and an area centered on the border of Texas and Louisiana. For the loblolly pine forest type, high rust hazard is currently concentrated in northern Florida and across the Upper Coastal Plain and Piedmont regions of Alabama, Georgia, and South Carolina. Rust hazard is lowest for loblolly pine along the northern portion of its historical geographical range (fig. 6).

SUMMARY

The current patterns of rust incidence and rust hazard generally reflect what was observed by Squillace (1976) and Starkey and others (1997). Given that *C. quercum* f. sp. *fusiforme* is endemic to the southeastern United States, such patterns are likely to persist. As slash pine and loblolly pine continue to expand through plantation forestry, areas where these species are not historically native and where one or both of them now exist should be monitored closely for evidence of fusiform rust.

Rust incidence is generally higher in slash pine plantations than loblolly pine plantations, and though decreases in fusiform rust incidence were evident over the last 30 to 40 years in young loblolly pine plantations, no declines were observed in young slash pine plantations. Thus, there appears to be an opportunity to improve fusiform rust management in slash pine plantations through either improved planting stock or silvicultural practices, or both.

ACKNOWLEDGMENTS

I would like to thank Ellis B. Cowling and Dale A. Starkey for their collaboration in the study on which I based my BSSRC presentation. Appreciation is also extended to the FIA field crews, past and present, for their diligent collection of fusiform rust incidence data.

LITERATURE CITED

Allen, H.L.; Fox, T.R.; Campbell, R.G. 2005. What is ahead for intensive pine plantation silviculture in the South? Southern Journal of Applied Forestry. 29(2): 62-69.

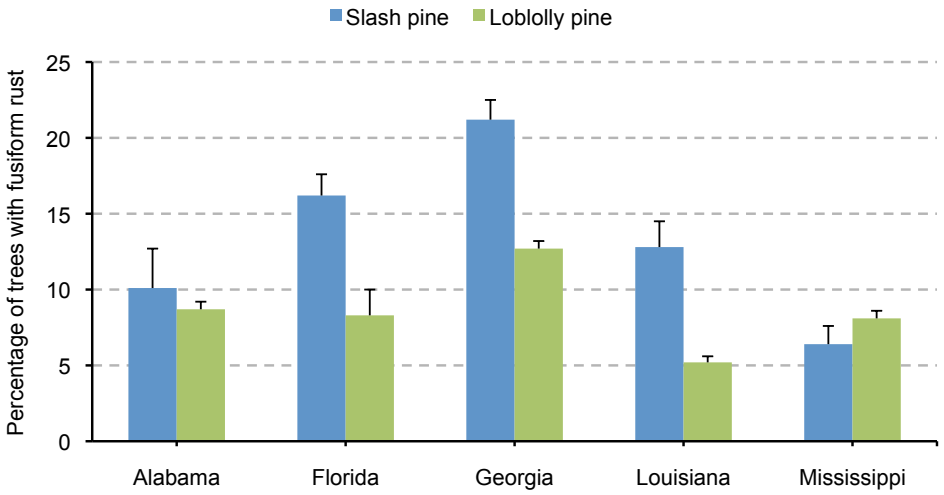


Figure 3—Estimated percentage of live trees (diameter at breast height ≥5 inches) with symptoms of fusiform rust, by state and forest type, for the years 2011 (Florida) and 2012 (all other states). Bars extending from the columns represent one standard error.

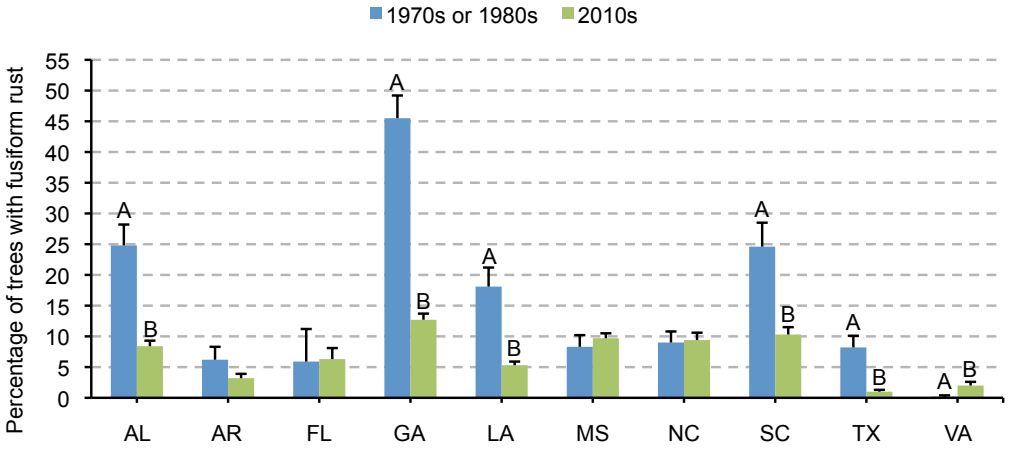


Figure 4—Estimated percentage of live trees (diameter at breast height ≥5 inches) with symptoms of fusiform rust in young (5 to 15 years old) loblolly pine plantations, by state and inventory decade. Bars extending from the columns represent one standard error. Within a state, columns with different letters are significantly different from one another (α=0.05).

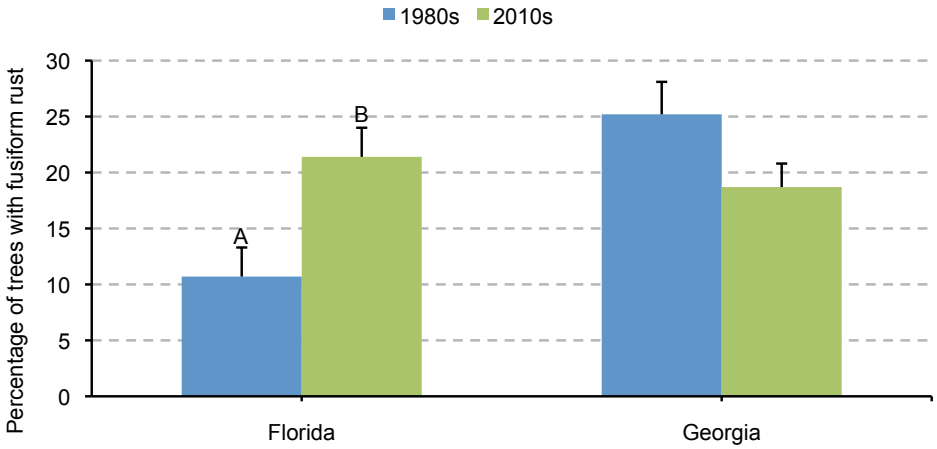


Figure 5—Estimated percentage of live trees (diameter at breast height ≥5 inches) with symptoms of fusiform rust in young (5 to 15 years old) slash pine plantations, by state and inventory decade. Bars extending from the columns represent one standard error. Within a state, columns with different letters are significantly different from one another (α=0.05).



Figure 6—Historical ranges of slash pine (*Pinus elliottii*) and loblolly pine (*P. taeda*) in the eastern United States (U.S. Geological Survey 1999).

Anderson, R.L.; McClure, J.P.; Cost, N.; Uhler, R.J. 1986. Estimating fusiform rust losses in five southeast states. Southern Journal of Applied Forestry. 10(4): 237-240.

Aspinwall, M.J.; McKeand, S.E.; King, J.S. 2012. Carbon sequestration from 40 years of planting genetically improved loblolly pine across the southeast United States. Forest Science. 58(5): 446-456.

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

Brawner, J.T.; Carter, D.R.; Huber, D.A.; White, T.L. 1999. Projected gains in rotation-age volume and value from fusiform rust resistant slash and loblolly pines. Canadian Journal of Forest Research. 29: 737-742.

Cochran, W.G. 1977. Sampling techniques. 3rd ed. New York: John Wiley & Sons, Inc. 428 p.

Conner, R.C.; Hartsell, A.J. 2002. Chapter 16: Forest area and conditions. In: Wear, D.N.; Greis, J.G., eds. 2002. The southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 357-402.

Cowling, E.; Randolph, K. 2013. Potentials for mutually beneficial collaboration between FIA specialists and IEG-40 pathologists and geneticists working on fusiform rust. Forests. 4: 1220-1231.

Cowling, E.; Young, C. 2013. Narrative history of the Resistance Screening Center: its origins, leadership and partial list of public benefits and scientific contributions. Forests. 4: 666-692.

ESRI®. 2010. ArcMAP™ 10.0. Redlands, California.

Fox, T.R.; Jokela, E.J.; Allen, H.L. 2007. The development of pine plantation silviculture in the southern United States. Journal of Forestry. 105(7): 337-347.

Li, B.; McKeand, S.; Weir, R. 1999. Tree improvement and sustainable forestry – impact of two cycles of loblolly pine breeding in the U.S.A. Forest Genetics. 6(4): 229-234.

McKeand, S.; Mullin, T.; Byram, T.; White, T. 2003. Deployment of genetically improved loblolly and slash pines in the South. Journal of Forestry. 101(3): 32-37.

Miles, P.D. 2015. Forest Inventory EVALIDator web-application, version 1.6.0.01. St. Paul, MN: U.S. Department of Agriculture Forest Service, Northern Research Station. http://apps.fs.fed.us/Evalidator/evalidator.jsp. [Date accessed: March 9, 2015].

O'Connell, B.M.; LaPoint, E.B.; Turner, J.A. [and others]. 2010. The Forest Inventory and Analysis database: Database description and users manual, version 5.1.4 for phase 2. Washington, DC: U.S. Department of Agriculture Forest Service. 537 p.

Phelps, W.R.; Czabator, F.L. 1978. Fusiform rust of southern pines. U.S. Department of Agriculture Forest Service Forest Insect and Disease Leaflet 26. http://www.na.fs.fed.us/spfo/pubs/fidls/fusiform/fidl-fusi.htm. [Date accessed: October 27, 2014].

Randolph, K.C.; Cowling, E.B.; Starkey, D.A. 2015. Long-term changes in fusiform rust incidence in the southeastern United States. Journal of Forestry. 113(4): 381-392.

Schmidt, R.A. 2003. Fusiform rust of southern pines: a major success for forest disease management. Phytopathology. 93: 1048-1051.

Squillace, A.E. 1976. Geographic patterns of fusiform rust infection in loblolly and slash pine plantations. Res. Note SE-232. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 4 p.

Starkey, D.A.; Anderson, R.L.; Young, C.H. [and others]. 1997. Monitoring incidence of fusiform rust in the south and change over time. Protection Rep. R8-PR 30. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region Forest Health Protection. 29 p.

USDA Forest Service. 2006. FIA glossary. http://socrates.lv-hrc.nevada.edu/fia/ab/issues/pending/glossary/Glossary_5_30_06.pdf. [Date accessed: September 29, 2014].

U.S. Geological Survey. 1999. Digital representation of "Atlas of United States Trees" by Elbert L. Little, Jr. http://gec.cr.usgs.gov/data/little. [Date accessed: November 25, 2013].

Vergara, R.; White, T.L.; Huber, D.A.; Schmidt, R.A. 2007. Realized genetic gains of rust resistant selections of slash pine (*Pinus elliottii* var. *elliottii*) planted in high rust hazard sites. Silvae Genetica. 56(5): 231-242.

Zarnoch, S.J.; Bechtold, W.A. 2000. Estimating mapped-plot forest attributes with ratios of means. Canadian Journal of Forest Research. 30: 688-697.

Zhao, D.; Kane, M. 2012. Differences in growth dynamics of loblolly and slash pine plantations in the southeastern United States. Forest Ecology and Management. 281: 84-92.

FATE OF RESIDUAL CANOPY TREES FOLLOWING HARVESTING TO UNDERPLANT LONGLEAF PINE SEEDLINGS IN LOBLOLLY PINE STANDS IN GEORGIA

Benjamin O. Knapp, G. Geoff Wang, Joan L. Walker, and Robert N. Addington[1]

Abstract—Over the past few decades, reports of forest health problems have concerned scientists and forest managers in loblolly pine forests of the southeastern United States. Several interacting factors likely contribute to observed reductions in loblolly pine health, including low resource availability on many upland sites that were once dominated by longleaf pine. Currently, land managers are interested in converting such sites back to longleaf pine, while maintaining ecosystems services that are now provided by loblolly pine. Recent research suggests that underplanting longleaf pine in loblolly pine stands may be a viable solution for stand conversion, but it is not clear how such treatments affect the longevity or condition of residual canopy trees. In this study, we compared the effects of three levels of uniformly-distributed stand density (uncut Control, ~16 m^2/ha basal area; MedBA, ~9 m^2/ha basal area; LowBA, ~6 m^2/ha basal area) and three gap sizes (LG, radius of 40 m; MG, radius of 30 m; and SG, radius of 20 m) on the survival, growth, and canopy condition of residual trees through five years after harvest. Survival was not significantly affected by treatment ($p = 0.5899$), with an average of 96.8 percent survival. Tree growth during the study period was significantly greater on the LowBA plots than on the Control plots. By the end of the study period, LowBA plots had greater live crown ratios and less crown dieback than Control plots. Our results suggest that harvesting loblolly pine trees for underplanting longleaf pine does not accelerate pine decline in the short-term but does have the potential for growth release of residual trees.

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.) historically dominated the southeastern United States, occurring on site types that included xeric sandhills, coastal plain flatwoods, and mountainous portions of Georgia and Alabama (Peet 2006). Frequent surface fire regimes were common to longleaf pine ecosystems and maintained longleaf pine dominance throughout its range. In contrast, loblolly pine (*P. taeda* L.) was more commonly restricted to wetter sites that experienced relatively infrequent fire (Schultz 1999). Following widespread logging in the 1800s and early 1900s and fire exclusion policies of the early to mid-1900s, upland sites that had once supported longleaf pine were commonly reforested with loblolly pine through natural or artificial regeneration (Schultz 1999). As these stands developed through time, largely in the absence of fire, their resulting structure, composition, and function were notably different from frequently burned longleaf pine ecosystems. With current interest in the conservation value and ecosystem services provided by longleaf pine ecosystems, land managers throughout the southeast are interested in restoring longleaf pine to loblolly pine stands on upland sites.

Although longleaf pine seedlings are conventionally considered intolerant of competition (Boyer 1990), recent publications have discussed benefits of using a gradual approach to convert slash pine (*P. elliottii* Engelm.) (Kirkman and others 2007) and loblolly pine stands (Hu and others 2012, Knapp and others 2013) to longleaf pine by reducing canopy densities and underplanting longleaf pine seedlings. Retaining canopy pines during the restoration process may be desirable for several reasons. First, canopy pines reduce the rate of growth of hardwood seedlings and saplings that are commonly abundant on sites requiring restoration (Jack and others 2006, Kirkman and others 2007, Knapp and others 2014). In addition, needlefall from canopy pines provides an important source of fine fuel for frequent fire management (Mitchell and others 2009). Finally, the existing pines may be providing important wildlife habitat. This is the case at Fort Benning in Georgia and Alabama, where populations of the federally-endangered red cockaded woodpecker (*Picoides*

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borealis; RCW) currently utilize loblolly pine trees for nesting and foraging habitat as longleaf pine restoration occurs throughout the landscape (Fort Benning 2014).

Interest in converting loblolly pine stands to longleaf pine is heightened on sites believed to be susceptible to pine decline, a condition in which loblolly pines have a rapid reduction in growth followed by mortality (Ryu and others 2013). Reports of pine decline have been documented since the 1960s, with symptoms including short, chlorotic needles, sparse crowns, fine root deterioration, and reduced radial growth (Eckhardt and Menard 2008, Eckhardt and others 2010). Pine decline has been associated with the presence of insect and fungal species, including bark beetles (*Hylastes* spp.) and species of *Leptographium* fungi (Otrosina and others 1999, Eckhardt and others 2007). In addition, pine decline has been associated with low-quality sites that generally result in increased resource stress on trees and may be better suited for longleaf pine than loblolly pine (Eckhardt and Menard 2008, Eckhardt and others 2010, Ryu and others 2013). Given that pine decline appears to be incited by a complex of interacting stressors, it is possible that forest management practices that improve stand vigor, such as thinning in overstocked stands, may reduce susceptibility to decline (Eckhardt and others 2010).

The relatively xeric, upland sites commonly targeted for conversion from loblolly pine to longleaf pine are also likely to be the most susceptible to loblolly pine decline due to resource stress. The objectives related to a gradual conversion to longleaf pine on such sites may be compromised by unexpected mortality of residual canopy trees, especially if loblolly pine canopy trees are retained for other ecosystem services during restoration. The goal of this study was to evaluate the short-term effects of alternative harvesting treatments, used in conjunction with underplanting longleaf pine seedlings, on residual loblolly pine trees. Specifically, we quantified changes in 1) canopy health metrics; 2) survival; and 3) growth through five years following the harvesting treatments.

METHODS

This study was conducted at Fort Benning Military Installation in Muscogee and Chattahoochee Counties, GA and Russell County, AL (~32.38 °N, 84.88 °W). Fort Benning occupies approximately 74,000 ha, of which 30,000 ha are classified as pine forest. Longleaf pine is currently present on approximately 20,000 ha, and longleaf pine restoration is a primary objective of forest management on upland sites (Fort Benning 2014). The northern two-thirds of Fort Benning is classified as Sand Hills and the southern one-third is classified as Upper Loam Hills (Baily 1995). Upland soils in the area are generally low in organic matter and nutrient holding capacity, although the Upper Loam Hills region has higher silt and clay content than the coarse-textured, sandy soils of the Sand Hills.

The study used a randomized, complete block design, with each block located in a different loblolly pine stand on upland sites at Fort Benning. Each of five blocks were divided into seven experimental units that were 100 m × 100 m (1 ha), and experimental units were randomly assigned one of seven overstory harvesting treatments. Three treatments resulted in approximately uniform distribution of the residual canopy, including Control (uncut; residual basal area ~ 16 m^2/ha), MedBA (residual basal area of 9 m^2/ha), and LowBA (residual basal area of 6 m^2/ha). Harvesting focused on removing smaller trees or trees of poor form. Three treatments used group-selection to create canopy gaps of three sizes, including LG (large gap; radius of 40 m and total area of approximately 5027 m^2), MG (medium gap; radius of 30 m and total area of approximately 2827 m^2), and SG (small gap; radius of 20 m and total area of approximately 1257 m^2). An additional clearcut treatment was used in the experiment but is not relevant to this study because all canopy trees were removed. Harvesting was completed by December 2007, and container-grown longleaf pine seedlings were planted throughout the study area at 1.8 m × 3.7 m spacing in January 2008.

Immediately following harvesting in 2007, residual overstory trees (diameter at breast height (dbh) ≥ 10 cm) were each identified with an aluminum tag, and the species and dbh (cm) were recorded for each tree. In plots with uniformly distributed trees (Control, MedBA, and LowBA), all trees within each plot were measured. In group-selection plots (LG, MG, and SG), all trees within 20 m from the edge of the canopy gap were measured. The condition of each tree crown was assessed in July 2008 and again in July 2012 following protocol developed for the USDA Forest Service Forest Health Monitoring Program (Schomaker and others 2007). In this study, we report on the "uncompacted live crown ratio", measured as the ratio of live crown length to aboveground tree length (reported as percent of total tree length), "crown dieback", measured as the proportion of the crown that has experienced recent dieback from the upper and outer edges (reported as percent of crown area), and "crown density", measured as the amount of crown stem, branches, twigs, shoots, buds, foliage, and reproductive structures that block light penetration through the crown (reported as percent of crown area) (Schomaker and others 2007). In addition, we measured the crown diameter across the drip-line of each tree along two axes, the first of which was the widest crown diameter and the second being its perpendicular axis. In July 2012, the dbh of each tree was recorded, and the cause of death was noted for any tree that had died during the study period.

The data were summarized at the plot-level to determine effects of harvesting treatment on crown condition, survival, and growth. We used mixed-model Analysis of Variance (ANOVA) with a random block effect to determine treatment effects on uncompacted live crown ratio, crown dieback, and crown density in 2008 and in 2012. Using trees remaining alive in 2012, we calculated the difference from 2008 to 2012 for each variable and similarly tested for treatment effects on the change during the study period. Mixed-model ANOVA was used to test treatment effects on survival percentage from 2007 to 2012 and on dbh in 2007, 2012, and the change in dbh from 2007 to 2012. For each model with a significant treatment effect, pair-wise comparisons among treatments were made using Tukey's HSD test. To determine if crown condition or tree size were indicators of future mortality, each tree was classified as alive or dead based on status in 2012. Because of the low sample size for dead trees, we grouped trees across all treatments, and we tested for differences in crown condition metrics and dbh in 2007 between the two groups. Finally we used simple linear regression to determine relationships between dbh growth (change in dbh from 2007 to 2012) and crown condition metrics. We determined statistical significance when $p \leq 0.05$ for all analyses.

RESULTS

The harvesting treatments significantly reduced the basal area of the LowBA plots relative to all other treatments, resulting in approximately 6.4 m^2/ha basal area (table 1). The MedBA plots had significantly lower basal areas than the Control and all gap plots. The Control plots and the residual trees surrounding the group openings in LG, MG, and SG did not differ in basal area following harvest. The number of trees per hectare followed similar patterns among the treatments, with higher variability. There was no difference in the size (dbh) of the residual canopy trees following harvesting treatments (table 1).

During the first measurement period following harvest (2008 for the crown condition variables and 2007 for dbh), there were no significant differences in live crown ratio, crown dieback, crown density, or dbh (table 2). In 2012, there were significant treatment effects on live crown ratio and crown dieback. Live crown ratio was significantly greater on LowBA plots than on Control plots, and crown dieback was significantly lower on MedBA and LowBA plots than on Control plots. There were no effects of harvesting treatment on the stand-level averages of dbh or the crown densities of residual trees in 2012 (table 2). The response of live crown ratio, measured as change from 2008 to 2012, was significantly greater on LowBA, MG, and SG plots when compared to Control plots (fig. 1A). Changes in crown dieback (fig. 1B) and crown density (fig. 1C) were not affected by treatment and were highly variable within the treatments. The change in dbh from 2007 to 2012 was significantly greater on LowBA plots than on Control or MG plots (fig. 1D). Change in dbh was significantly related to the change in crown diameter from 2008 to 2012 and to live crown ratio in 2008 (fig. 2).

Survival of residual trees from 2007 through 2012 was high (mean = 96.8 percent) and did not significantly differ among treatments (table 2). The cause of mortality was not evident for the majority of the 59 trees that had died during the study period, although 20 percent of the dead trees had snapped boles by 2012. Three of the dead trees (5.1 percent) had apparent lightning strikes. The live crown ratios and crown densities from 2007 were significantly lower for trees that had died by 2012 than for trees that remained alive (table 3). Likewise, crown dieback from 2007 was significantly greater for trees that had died by 2012. There was no difference in the dbh from 2007 for trees that were alive vs dead in 2012 (table 3).

Table 1—Means and standard errors of stand structure attributes in 2007 (following initial harvest), including basal area (m^2/ha), trees per hectare, and dbh (cm). The p-values indicate significance from the global ANOVA test, and the same superscript letter within a column indicates no significant difference from pair-wise comparisons

	Basal area (m^2/ha)		Trees per hectare		DBH (cm)	
Treatment	Mean	SE	Mean	SE	Mean	SE
Control	17.5^{A}	1.2	293.4^{A}	68.7	30.7	3.0
MedBA	9.4^{B}	0.4	136.5^{BC}	23.4	31.2	2.8
LowBA	6.4^{C}	0.5	94.5^{C}	20.1	31.0	2.6
LG	16.5^{A}	0.9	269.9^{A}	31.3	30.8	2.1
MG	16.2^{A}	1.0	232.8^{A}	29.6	31.7	2.0
SG	15.0^{A}	1.1	203.7^{AB}	28.1	32.3	2.7
p-value	< 0.0001		0.0005		0.9779	

Table 2—Mean and standard errors of crown condition variables (live crown ratio, crown dieback, crown density) in 2008 and 2012, diameter at breast height in 2007 and 2012, and percent survival from 2007 through 2012. The p-values indicate significance from the global ANOVA test, and the same superscript letter within a row indicates no significant difference from pair-wise comparisons

	Control		MedBA		LowBA		LG		MG		SG		
Response variable	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	p-value
Live crown ratio (%)													
2008	36.2	1.4	38.6	0.8	38.2	1.4	37.1	1.0	36.0	1.3	37.3	0.6	0.4989
2012	37.0^{B}	1.5	40.7^{AB}	0.8	42.4^{A}	1.1	39.1^{AB}	0.9	39.6^{AB}	1.3	40.6^{AB}	1.1	0.0473
Crown dieback (%)													
2008	10.9	0.7	9.2	0.9	9.8	1.3	10.5	1.7	12.0	0.5	11.2	0.7	0.4287
2012	8.9^{A}	0.6	7.0^{B}	0.5	7.0^{B}	0.2	7.6^{AB}	0.5	7.4^{AB}	0.6	7.5^{AB}	0.3	0.0213
Crown density (%)													
2008	46.4	1.8	47.4	2.3	48.6	1.6	47.6	1.5	50.0	2.1	48.5	1.4	0.2989
2012	44.6	2.4	44.0	2.6	47.8	1.1	43.7	1.8	44.2	2.3	43.5	3.1	0.5502
DBH (cm)													
2007	30.7	3.0	31.2	2.8	31.0	2.6	30.8	2.1	31.7	2.0	32.3	2.7	0.9779
2012	32.7	3.3	34.7	2.7	34.2	2.7	33.1	2.3	33.5	2.1	35.0	2.7	0.9224
Survival													
2007 - 2012 (%)	99.0	0.7	95.5	2.1	96.6	1.0	97.0	1.2	97.5	1.2	95.2	2.8	0.5899

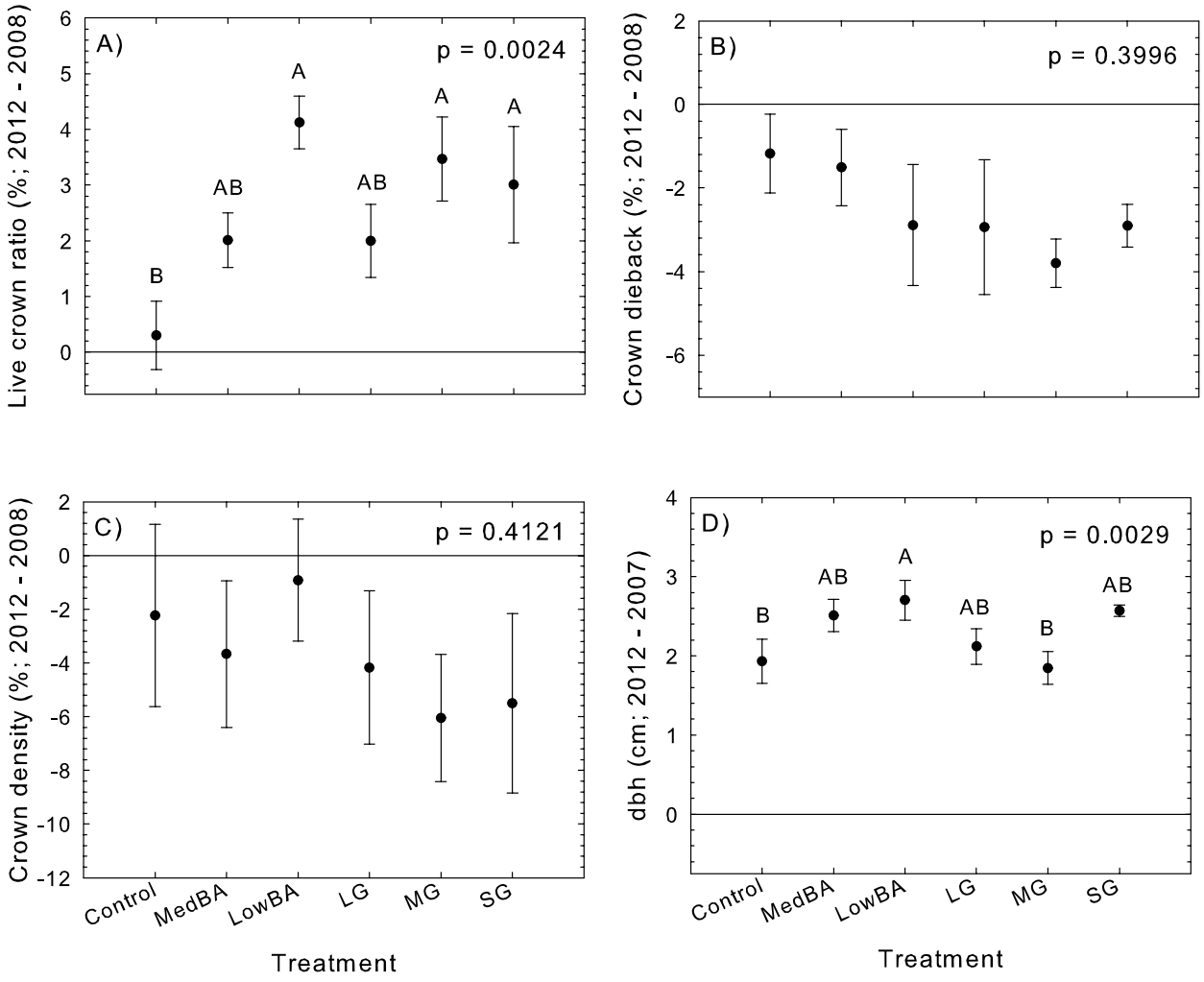


Figure 1—Means and standard errors by treatment for the change from 2008 to 2012 for A) live crown ratio (%), B) crown dieback (%), C) crown density (%), and D) change from 2007 to 2012 for diameter at breast height (dbh; cm). Within each panel, the p-values indicate significance from the global ANOVA test, and the same letter indicates no significant difference from pair-wise comparisons.

DISCUSSION

Unpredictable forest health may complicate efforts to integrate canopy retention into regeneration or restoration practices. Prescriptions for gradually converting stands of loblolly pine to longleaf pine have suggested target ranges of residual stand density intended to provide a balance between the desirable (e.g., reducing growth rates of hardwood regeneration) and undesirable (e.g., reducing growth rates of underplanted longleaf pine seedlings) effects on restoration targets (Knapp and others 2013, Knapp and others 2014). Although observations of pine decline have raised concerns about the longevity of loblolly pine forests at Fort Benning (Ryu and others 2013), we found little evidence of mortality from declining pine health in our study. The five-year mortality rate based on all trees sampled in this study was 4.8 percent, with wind damage (i.e., snapped boles) accounting for 20 percent of the mortality. In natural longleaf pine forests of the Gulf Coastal Plain region, lightning has been reported to cause greater mortality than wind damage (Platt and others 1988, Palik and Pederson 1996). Outcalt (2008) reported that lightning killed 1 tree/8 ha/year in longleaf pine forests in South Carolina, which was similar to the rate of lightning mortality observed in this study. It is possible that the higher incidence of wind snap in our study was related to the canopy reduction by harvesting, although no clear patterns between treatment and wind snap emerged in our study.

The crown condition metrics used in this study have been used to assess pine decline at Fort Benning (Menard 2007, Ryu and others 2013) and in central Alabama (Eckhardt and others 2007), and we found few indications of poor health. Live crown ratios greater than 30 percent generally indicate that crown size is

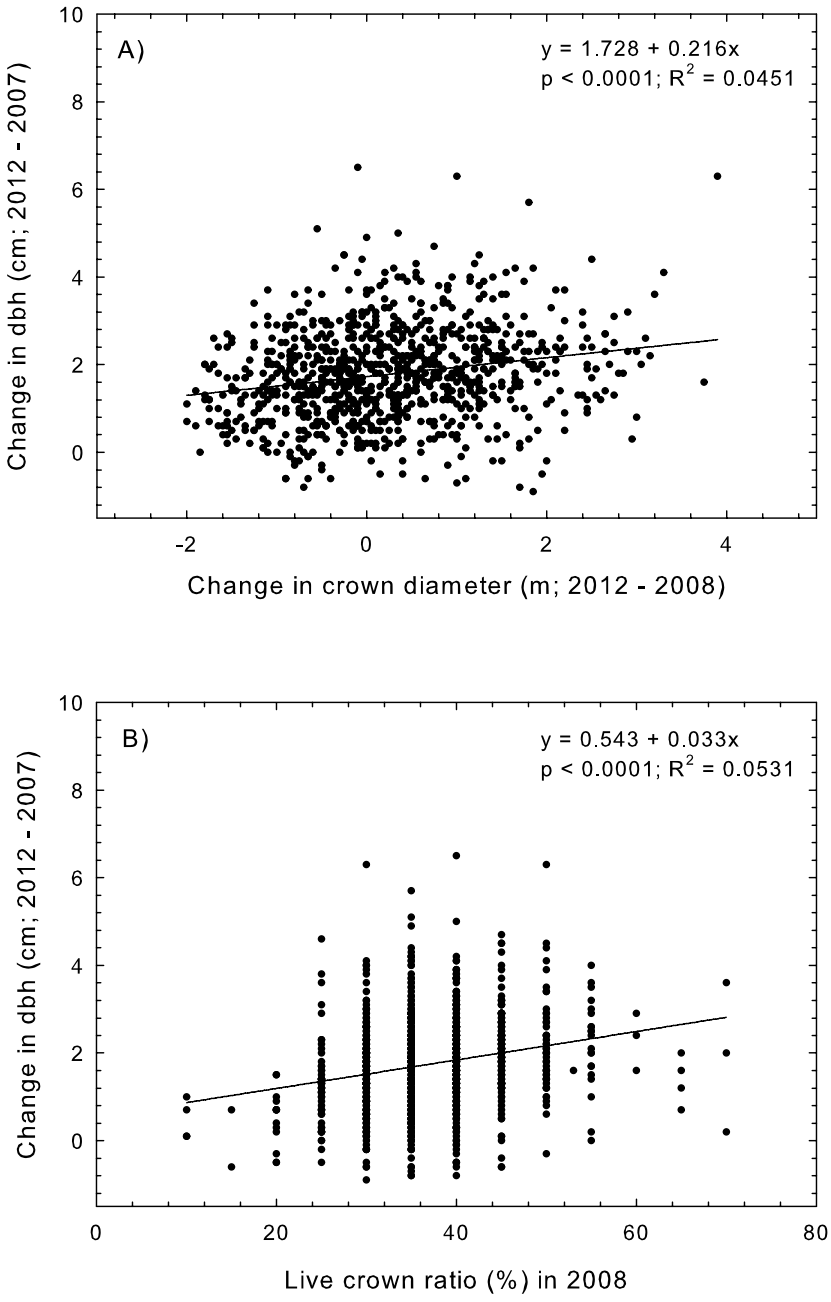


Figure 2—Scatterplots and linear regressions between diameter growth (change in dbh from 2007 to 2012) and A) change in crown diameter from 2008 to 2012 and B) the live crown ratio (%) in 2008.

Table 3—Means and standard errors of crown condition (live crown ratio, crown dieback, and crown density) and dbh at the initial measurement period for trees that were alive in 2012 and trees that were dead in 2012. All trees sampled were alive at the initial measurement period. The p-values indicate significant differences in the initial condition between live and dead trees in 2012 for each respective variable

	Status of tree in 2012				
	Live (n = 1163)		Dead (n = 59)		
Variable	Mean	SE	Mean	SE	p-value
Live crown ratio (%)	37.18	0.23	33.41	1.33	0.0031
Crown dieback (%)	10.80	0.24	21.36	3.09	< 0.0001
Crown density (%)	47.65	0.26	40.65	1.82	< 0.0001
Diameter at breast height (cm)	31.35	0.29	30.62	1.52	0.7891

not limiting to tree growth (Smith and others 1997), suggesting that stand densities prior to the harvesting treatments were not restricting crown development. Alexander and Palmer (1999) reported crown dieback values of around 5 percent from a series of loblolly pine plots in the southeastern United States but discuss that trees are not considered unhealthy if dieback is < 20 percent. The crown density values reported in our study are similar to values reported in a region-wide description of loblolly pine crown condition from Forest Inventory and Analysis plots measured in the late 1990s (Randolph 2006). Moreover, Menard (2007) reported crown density values similar to those found on our sites for loblolly pine trees that were asymptomatic for pine decline but crown densities from 35 to 40 percent for symptomatic pines at Fort Benning. Despite finding few indicators of poor tree health, the trees that died within our study period exhibited lower vigor at the first measurement period following harvest when compared to the trees that lived. In particular, crown dieback of trees that died averaged around 20 percent, providing support for the threshold discussed by Alexander and Palmer (1999), and crown density was closer to 40 percent, as found by Menard (2007).

Our results indicate that the harvesting stimulated a growth response from the residual trees, observed through the positive change of live crown ratio and dbh on the LowBA treatment. Trees generally respond to increased resource availability by increasing foliar production and crown size, resulting in the eventual allocation of carbon to diameter growth (Oliver and Larson 1996). This pattern was further supported in our data by the positive relationships between diameter growth response following harvesting and the change in crown size and the initial live crown ratio, although these relationships were weak. Our results also suggest that reducing canopy density resulted in increased tree vigor, with increased live crown ratio and reduced crown dieback on LowBA plots. In a region-wide analysis, Klos and others (2009) also found that stand density was negatively related to growth but positively related to mortality of pines. Given the relatively short response period of this study (five years), it is not clear if the magnitude of growth/vigor responses will become greater through time. However, loblolly pine stands with lower density are likely more resistant to stressors like drought (Klos and others 2009) and southern pine beetle (Belanger and others 1993).

CONCLUSION

There were few indications of forest health problems associated with pine decline on these study sites. However, this study did not cover a full range of site and stand conditions on which loblolly pine decline has been described, and it is not clear if similar results would occur on sites with higher tree stress or if pine decline was present initially. The associations of the crown condition metrics with tree mortality in this study support their utility in assessing tree vigor in loblolly pine forests. Treatments that reduced canopy density resulted in apparent increases in diameter growth and tree vigor, supporting previous reports that reducing stand density can likewise reduce risk of pine decline (Eckhardt and others 2010). Our results suggest that recommendations for reducing stand density to underplant longleaf pine seedlings during conversion of loblolly pine stands are compatible with improving the vigor of residual trees.

LITERATURE CITED

Alexander, S.A.; Palmer, C.J. 1999. Forest health monitoring in the United States: first four years. Environmental Monitoring and Assessment. 55: 267-277.

Baily, R.G. 1995. Description of the ecoregions of the United States, 2nd Edition. Misc. Pub. No 1391. Washington, DC: U.S. Department of Agriculture. 108 p.

Belanger, R.P.; Hedden, R.L.; Lorio, Jr., P.L. 1993. Management strategies to reduce losses from the southern pine beetle. Southern Journal of Applied Forestry. 17: 150-154.

Boyer, W.D. 1990. *Pinus palustris* Mill. Longleaf pine. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America. Vol. 1. Conifers. Agric. Handb. 65. Washington, DC: U.S. Department of Agriculture: 405-412.

Eckhardt, L.; Menard, R.D. 2008. Topographic features associated with loblolly pine decline in central Alabama. Forest Ecology and Management. 255: 1735-1739.

Eckhardt, L.G.; Weber, A.M.; Menard, R.D. [and others]. 2007. Insect-fungal complex associated with loblolly pine decline in central Alabama. Forest Science. 53: 84-92.

Eckhardt, L.; Sword Sayer, M.A.; Imm, D. 2010. State of pine decline in the southeastern United States. Southern Journal of Applied Forestry. 34: 138-141.

Fort Benning, 2014. Integrated natural resources management plan. Directorate of Public Works, Environmental Management Division, Fort Benning, GA: U.S. Department of the Army. 1033 p.

Hu, H.; Wang, G.G.; Walker, J.L.; Knapp, B.O. 2012. Silvicultural treatments for converting loblolly pine to longleaf pine dominance: effects on planted longleaf pine seedlings. Forest Ecology and Management. 276: 209-216.

Jack, S.B.; Mitchell, R.J.; Pecot, S.D. 2006. Silvicultural alternatives in a longleaf pine/wiregrass woodland in southwest Georgia: understory hardwood response to harvest-created gaps. In: Connor, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 85-89.

Kirkman, L.K.; Mitchell, R.J.; Kaeser, M.J. [and others]. 2007. The perpetual forest: using undesirable species to bridge restoration. Journal of Applied Ecology. 44: 604-614.

Klos, R.J.; Wang, G.G.; Bauerle, W.L.; Rieck, J.R. 2009. Drought impacts on forest growth and mortality in the southeast USA: an analysis using Forest Health and Monitoring data. Ecological Applications. 19: 699-708.

Knapp, B.O.; Wang, G.G.; Walker, J.L. 2013. Effects of canopy structure and cultural treatments on the survival and growth of *Pinus palustris* Mill. seedlings underplanted in *Pinus taeda* L. stands. Ecological Engineering. 57: 46-56.

Knapp, B.O.; Walker, J.L.; Wang, G.G. [and others]. 2014. Effects of overstory retention, herbicides, and fertilization on sub-canopy vegetation structure and functional group composition in loblolly pine forests restored to longleaf pine. Forest Ecology and Management. 320: 149-160.

Menard, R.D. 2007. An assessment of the risk mapping system for the use of managing loblolly pine decline sites within red-cockaded woodpecker habitat. Baton Rouge, LA: Louisiana State University. 45 p. M.S. thesis.

Mitchell, R.J.; Hiers, J.K; O'Brien, J.; Starr, G. 2009. Ecological forestry in the southeast: understanding the ecology of fuels. Journal of Forestry. 107: 391-397.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. 2d ed. New York: John Wiley & Sons, Inc. 520 p.

Otrosina, W.J.; Bannwart, D.; Roncadori, R.W. 1999. Root-infecting fungi associated with a decline of longleaf pine in the southeastern United States. Plant and Soil. 217: 145-150.

Outcalt, K.W. 2008. Lightning, fire, and longleaf pine: using natural disturbance to guide management. Forest Ecology and Management. 255: 3351-3359.

Palik, B.J.; Pederson, N. 1996. Overstory mortality and canopy disturbances in longleaf pine ecosystems. Canadian Journal of Forest Research. 26: 2035-2046.

Peet, R.K. 2006. Ecological classification of longleaf pine woodlands. In: Jose, S.; Jokela, E.J.; Miller, D.L., eds. The longleaf pine ecosystem: ecology, silviculture, and restoration. New York: Springer: 51-94.

Platt, W.J.; Evans, G.W.; Rathbun, S.L. 1988. The population dynamics of a long-lived conifer (*Pinus palustris*). The American Naturalist. 131: 491-525.

Randolph, K.C. 2006. Descriptive statistics of tree crown condition in the southern United States and impacts on data analysis and interpretation. Gen. Tech. Rep. SRS-94. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 17 p.

Ryu, S.R.; Wang, G.G.; Walker, J.L. 2013. Factors influencing loblolly pine stand health in Fort Benning, Georgia, USA. Forest Science and Technology. 9: 137-146.

Schomaker, M.E.; Zarnoch, S.J.; Bechtold, W.A. [and others]. 2007. Crown-condition classification: a guide to data collection and analysis. Gen. Tech. Rep. SRS-102. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 78 p.

Schultz, R.P. 1999. Loblolly – the pine for the twenty-first century. New Forests. 17: 71-88.

Smith, D.M.; Larson, B.C.; Kelty, M.J.; Ashton, P.M.S. 1997. The practice of silviculture: applied forest ecology. 9th ed. New York: John Wiley & Sons, Inc. 537 p.

EFFECTS OF ELEVATED TROPOSPHERIC OZONE AND FLUCTUATING MOISTURE SUPPLY ON LOBLOLLY PINE SEEDLINGS INOCULATED WITH ROOT INFECTING OPHIOSTOMATOID FUNGI

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Abstract—Southern Pine Decline is a cause of premature mortality of *Pinus* species in the Southeastern United States. While the pathogenicity of ophiostomatoid fungi, associated with declining *Pinus* species, has been observed both in the laboratory and the field the driving mechanisms for success of fungal infection, as well as the bark-beetle vectors is less understood. The goal of this research is to provide insight into the role of future climatic conditions, specifically elevated tropospheric ozone and altered precipitation patterns, in the progression of Southern Pine Decline on loblolly pine (*Pinus taeda* L.). Two key questions were addressed: (1) will predicted concentrations of tropospheric ozone affect loblolly pine vigor and increase susceptibility to fungi associated with Southern Pine Decline?; and (2) will predicted precipitation patterns affect loblolly pine vigor and increase susceptibility to fungi associated with Southern Pine Decline? Our results indicate seedlings selected for susceptibility to root infecting ophiostomatoid fungi were more sensitive to elevated ozone than tolerant seedlings, however, neither ozone nor fluctuating moisture supply resulted in seedlings to becoming more susceptible to root infecting ophiostomatoid fungi.

INTRODUCTION

The physical and chemical climate of the earth has changed rapidly over the last 100 years and is predicted to continue in the future (Christensen and others 2007, IPCC 2013). Global climate is changing and is apparent across a wide range of observations of which the warming over the past 50 years is primarily as a result of human activity (Walsh and others 2014.). It is likely that natural disturbances in forest ecosystems will be altered by climate change and there is evidence that warmer temperatures have already shifted suitable habitats and ranges of some forest species (Kirilenko and Sedjo 2007). Climate change-induced modifications of frequency and intensity of forest wildfires, outbreaks of insects and pathogens and extreme events such as high winds may be more important than the direct impact of higher temperatures and elevated carbon dioxide levels (Kirilenko and Sedjo 2007). The direct effects of climate change on individual plants and plant communities may occur in the absence of pathogens, but also may bring about alterations in plant metabolism that will affect their interactions with pathogens (Garrett and others 2006).

Pests that vector pathogenic fungi (e.g. ophiostomatoid fungi) (Bentz and others 2010, Kirisits 2004) are important and of concern in a changing climate. The influence of bark beetles, among other pests, is well established in the literature and typically, drought causes host plants to become stressed leading to greater infestations (Jactel and others 2012, Jones and others 2008, Klepzig and others 2004, Koricheva and others 1998). Host stress is one factor playing a part in this interaction, but insect physiology and ecologic shifts can also have drastic effects (Clarke and Fraser 2004, Gillooly and others 2001).

Water availability and tropospheric ozone are both issues of concern to the Southeastern U.S. ecosystems (Chameides and Cowling 1995, IPCC 2013, Jones and others 2001, Phillips and others 2009, Wear and Greis 2002). Water availability and ozone levels may alter loblolly pine vigor and increase susceptibility to root infecting ophiostomatoid fungi. The study addressed two scientific questions: (1) will elevated tropospheric ozone concentrations decrease loblolly pine vigor and increase susceptibility root infecting ophiostomatoid fungi? and (2) will fluctuating moisture supply decrease loblolly pine vigor and increase susceptibility to root infecting ophiostomatoid fungi?

MATERIALS AND METHODS

The first question was addressed in 2013 utilizing large open-top chambers, three ozone concentrations and stem inoculations of four families of loblolly pine. Two of the families used were selected for tolerance

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(T1 and T2) to root infecting ophiostomatoid fungi while the others were more susceptible (S1 and S2) (Singh and others 2014). Seedlings were inoculated with *Leptographium terebrantis* Barras and Perry, *Grosmannia huntii* (Rob.-Jeffr.) Zipfel, de Beer and Wingfield and three control treatments.

Seedlings were placed in open-top chambers and exposed to three ozone treatments: CF – charcoal-filtered air (~1/2 non-filtered), NF – non-filtered air (ambient ozone), 2× - twice ambient air. Non-filtered air is representative of concentrations currently found around large urban areas such as either Atlanta, GA or Birmingham, AL (Chameides and others 1988). The 2× is indicative of potential future ozone scenarios for rural Piedmont regions over the next 50 years (Thompson 1992, Vingarzan 2004).

The second question was addressed in 2014 utilizing capped open-top chambers, three irrigation treatments and stem inoculations of four families of loblolly pine. The same families of loblolly pine were used as in the 2013 study. Seedlings were inoculated with *Leptographium terebrantis, Grosmannia huntii* and three control treatments.

Seedlings were exposed to three irrigation treatments: 3D – irrigation three days.week^{-1}, 4D – irrigation four days.week^{-1} and 7D – irrigation seven days.week^{-1}. Each treatment had the same amount of moisture applied; only the intensity and frequency was altered to simulate flooding/drying periodicity (Westra and others 2014).

RESULTS AND DISCUSSION

Although, neither elevated ozone (question 1) nor moisture stress (question 2) resulted in increased susceptibility to either *L. terebrantis* or *G. huntii*, there is evidence to suggest sensitivity to root infecting ophiostomatoid fungi is linked to abiotic stresses such as moisture stress and ozone. Tropospheric ozone induced a host response, even at low concentrations, and caused visible foliar injury. Families selected for the susceptibility to root infecting ophiostomatoid fungi had significantly greater visible ozone injury, occurring on a higher percentage of the total plants. This indicates that families of loblolly pine that are more tolerant to root infecting ophiostomatoid fungi than others may withstand short-term exposure to elevated ozone concentrations. This relationship has been seen observed in a similar study with the pathogen *Fusarium circinatum* Nirenberg and O'Donnell (Carrey and Kelley 1994).

The interaction between family susceptibility and moisture stress when challenged with the fungi is weak. Typically root pathogens and moisture stress act independently, as observed by others (Croisé and others 2001, Goheen and others 1978, Joseph and others 1998, Matusick and others 2008). Seedlings exposed to intense and infrequent irrigation events began to reduce metabolic functions towards the end of the experimental period. This strategy would likely cause seedlings to be outcompeted by other more tolerant vegetation as well as result in mortality. Family affected the response of the seedlings to water stress treatments. One of the two susceptible families had less growth with infrequent moisture events, while both tolerant families had more growth with infrequent events. Seiler and Johnson (1988) found that seed source affects the response of loblolly pine to water stress. Our results agree, however, the response is not linked to tolerance to root infecting ophiostomatoid fungi.

Tropospheric ozone and drought are considered potential threats to forests in the Southeastern U.S. (Jones and others 2001). While ozone concentrations have been moderated by air pollution legislation,[2],[3] the increasing temperatures and human population could increase ozone concentrations in the future (Gonzalez-Abraham and others 2014, Milesi and others 2003, U.S. Bureau of the Census 2009, Wear and Greis 2002). Changes in precipitation also have been observed and are expected to become more intense in the future (IPCC 2013, MacCracken and others 2000, Seager and others 2009, Wang and others 2010,). The Southeastern U.S. is already experiencing climatic changes which have had detrimental effects to both humans and natural ecosystems (Wang and others 2010).

Shifts in climate will change the way species interact with each other and individually (Manning and von Tiedemann 1995). In the Southeastern U.S., exposure to elevated concentrations of ozone over multiple seasons is predicted to decrease loblolly pine vigor and increase the tree's susceptibility to root infecting ophiostomatoid fungi. Drought and altered precipitation regimes will likely have negative impacts as well. Typically, the attributors of host-pathogen-environment interactions are easily categorized. When examining insect-fungal disease complexes, such as Southern Pine Decline, there is greater complexity to be considered.

To better understand the relationship between disease tolerance and sensitivity to moisture stress, a more thorough approach would be recommended. Using either soil moisture probes in larger planting pots (as described by Matusick and others 2008), or conducting

[2]Clean Air Act of 1970. Pub. L. 159 (July 14, 1955) 69 Stat. 322, and the amendments made by subsequent enactments. 42 U.S.C. 7401–7626. Last accessed at http://www.epw.senate.gov/envlaws/cleanair.pdf on [May 1, 2015].

[3]Clean Air Act of 1990. Pub. L. 159 (July 14, 1995) 69 Stat. 322, and the amendments made by subsequent enactments. 42 U.S.C. 7401-7626. Last accessed at http://www.epw.senate.gov/envlaws/cleanair.pdf on [May 1, 2015].

a through-fall exclusion methodology experiment could be used. Because our seedlings were potted and placed in OTCs on uneven ground, there are water runoff issues that can affect the relative humidity uniquely during different time periods of the day. The OTCs themselves can also have drying effects on warm days. This can alter the rate of evaporation from seedlings and cause a chamber effect.

Future research should focus on the effects of elevated carbon dioxide and warming temperatures with root infecting ophiostomatoid fungi. Another component missing in the climate-SPD interaction is the role of the bark beetles vectoring the ophiostomatoid fungi. Current monitoring efforts should focus on changes in the chemical and physical climate during insect monitoring trials. Elevated carbon dioxide and warming will likely alter host vigor and productivity which may increase or decrease susceptibility to biotic and abiotic agents.

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LITERATURE CITED

Bentz, B.J.; Régnière, J.; Fettig, C.J. [and others]. 2010. Climate change and bark beetles of the Western United States and Canada: direct and indirect effects. BioScience. 60(8): 602-613.

Carey, W.A.; Kelley, W.D. 1994. Interaction of ozone exposure and *Fusarium subglutinanas* inoculation on growth and disease development of loblolly pine seedlings. Environmental Pollution. 84: 35-43

Chameides, W. L., R. W. Lindsay, J. Richardson and C. S. Kiang. 1988. The role of biogenic hydrocarbons in urban photochemical smog: Atlanta as a case study. Science. 241(4872): 1473-75.

Chameides, W.L.; Cowling, E.B. 1995. The state of the southern oxidants study (SOS): policy relevant findings in ozone pollution research. Raleigh, NC: College of Forestry, North Carolina State University: 1988-1994.

Christensen, J.H.; Hewitson, B.; Busuioc, A. [and others]. 2007. Regional climate projections. Climate change, 2007: the physical science basis. Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change. New York: Cambridge University Press: 847-940.

Clarke, A.; Fraser, K.P. 2004. Why does metabolism scale with temperature? Functional Ecology. 18(2): 243-51.

Croisé, L.; Lieutier, F.; Cochard, H.; Dreyer, E. 2001. Effects of drought stress and high density stem inoculations with *Leptographium wingfieldii* on hydraulic properties of young Scots pine trees. Tree Physiology. 21 (7): 427-36

Garrett, K.A.; Dendy, S.P.; Frank, E.E. [and others]. 2006. Climate change effects on plant disease: genomes to ecosystems. Annual Review Phytopathology. 44: 489-509.

Gillooly, J.F.; Brown, J.H.; West, G.B. [and others]. 2001. Effects of size and temperature on metabolic rate. Science. 293(5538): 2248-51.

Goheen, D.J.; Cobb Jr., F.W.; McKibbin, G.N. 1978. Influence of soil moisture on infection of Ponderosa pine by *Verticicladiella wagenerii*. Phytopathology. 68 (6): 913-916.

Gonzalez-Abraham, R.; Avise, J.; Chung, S.H. [and others]. 2014. The effects of global change upon United States air quality. Atmospheric Chemistry and Physics Discussions. 14(23): 31843-31897.

IPCC. 2013: Climate change 2013: the physical science basis. Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change. In: Stocker, T.F.; Qin, D.; Plattner, G.-K. [and others], eds. New York: Cambridge University Press. 1535 p.

Jactel, H.; Petit, J.; Desprez-Loustau, M. [and others]. 2012. Drought effects on damage by forest insects and pathogens: A meta-analysis. Global Change Biology. 18(1): 267-276.

Burkett, V.; Ritschard, R.; McNulty, S. [and others]. 2001. Chapter 5: Potential consequences of climate variability and change for the Southeastern United States. In: National assessment synthesis team, comps. Climate change impacts on the United States: the potential consequences of climate change variability and change. Report for the U.S. Global Change Research Program. Cambridge, UK: Cambridge University Press: 137-164. Available at: http://data.globalchange.gov/assets/e9/97/436129058f2107f4925aeec13ed8/nca-2000-foundation-report.pdf. [Date accessed: May 1, 2015].

Jones, M.E.; Paine, T.D.; Fenn, M.E.; Poth, M.A. 2008. Influence of ozone and nitrogen deposition on bark beetle activity under drought conditions. Forest Ecology and Management. 200(1): 67-76.

Joseph, G.; Kelsey, R.G.; Thies, W.G. 1998. Hydraulic conductivity in roots of Ponderosa pine infected with Black-Stain (*Leptographium wageneri*) or Annosus (*Heterobasidion annosum*) Root Disease. Tree Physiology. 18(5): 333-339.

Kirisits, T. 2004. Fungal associates of European bark beetles with special emphasis on the ophiostomatoid fungi. In: Bark and wood boring insects in living trees in Europe, a synthesis. The Netherlands: Springer: 181-236.

Kirilenko, A.P.; Sedjo, R.A. 2007. Climate change impacts on forestry. Proceedings of the National Academy of Sciences. 104(50): 19697-19702.

Klepzig, K.D.; Flores-Otero, J.; Hofstetter, R.W.; Ayres, M.P. 2004. Effects of available water on growth and competition of Southern Pine Beetle associated fungi. Mycological Research. 108(2): 183-188.

Koricheva, J.; Larsson, S.; Haukioja, E. 1998. Insect performance on experimentally stressed woody plants: a meta-analysis. Annual Review of Entomology. 43(1): 195-216.

MacCracken, M.; Barron, E.; Easterling, D. [and others]. 2001. Chapter 1: Scenarios for climate variability and change. . In: National Assessment Synthesis Team, comps. Climate change impacts on the United States: the potential consequences of climate change variability and change. Report for the U.S. Global Change Research Program. Cambridge, UK: Cambridge University Press: 13-71. Available at: http://data.globalchange.gov/assets/e9/97/436129058f2107f4925aeec13ed8/nca-2000-foundation-report.pdf. [Date accessed: May 1, 2015].

Manning, W.J.; von Tiedemann, A. 1995. Climate change: potential effects of increased atmospheric Carbon dioxide (CO_2), ozone (O_3), and ultraviolet-B (UV-B) radiation on plant diseases. Environmental Pollution. 88: 219-245.

Matusick, G.; Eckhardt, L.G.; Enebak, S.A. 2008. Virulence of *Leptographium serpens* on Longleaf pine seedlings under varying soil moisture regimes. Plant Disease. 92(11): 1574-1576.

Milesi, C.; Elvidge, C.D.; Nemani, R.R.; Running, S.W. 2003. Assessing the impact of urban land development on net primary productivity in the Southeastern United States. Remote Sensing of Environment. 86(3): 401-410.

Phillips, D.L.; Johnson, M.G.; Tingey, D.T.; Storm, M.J. 2009. Elevated CO_2 and O_3 effects on fine-root survivorship in Ponderosa pine mesocosms. Oecologia. 160(4): 827-837.

Seager, R.; Tzanova, A.; Nakamura, J. 2009. Drought in the Southeastern United States: causes, variability over the last millennium, and the potential for future hydroclimate change. Journal of Climate. 22(19): 5021-5045.

Seiler, J.R.; Johnson, J.D. 1988. Physiological and morphological responses of three half-sib families of Loblolly pine to water-stress conditioning. Forest Science. 34(2): 487-495.

Singh, A.; Anderson, D.; Eckhardt, L.G. 2014. Variation in resistance of Loblolly pine (*Pinus taeda* L.) families against *Leptographium* and *Grosmannia* root fungi. Forest Pathology. 44(4): 293-298.

Thompson, A.M. 1992. The oxidizing capacity of the earth's atmosphere: probable past and future changes. Science. 256(5060): 1157-1165.

U.S. Census Bureau. 2009. History: 2000 Census of population and housing. Volume 2. Washington, DC: U.S. Department of Commerce, Economics and Statistics Administration. 613 p. + appendixes.

Vingarzan, R. 2004. A review of surface ozone background levels and trends. Atmospheric Environment. 38(21): 3431-3442.

Walsh, J.; Wuebbles, D.: Hayhoe, K. [and others]. 2014. Appendix 3. Climate science supplement. In: Melillo, J.M.; Richmond, T.C.; Yohe, G.W., eds. Climate change impacts in the United States: the third national climate assessment. Washington, DC: United States Global Change Research Program: 735-789. Available at: doi:10.7930/J0Z31WJ2. [Date accessed: May 1, 2015].

Wang, H.; Fu, R.; Kumar, A.; Li, W. 2010. Intensification of summer rainfall variability in the Southeastern United States during recent decades. Journal of Hydrometeorology. 11(4): 1007-1018.

Wear, D.N.; Greis, J.G. 2002. Southern Forest Resource Assessment Technical Report. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 635 p.

Westra, S.; Fowler, H.J.; Evans, J.P [and others]. 2014. Future changes to the intensity and frequency of short-duration extreme rainfall. Reviews of Geophysics. 52(3): 522-555.

SPECIES COMPOSITION AND SUCCESSION IN YELLOW PINE STANDS FOLLOWING SOUTHERN PINE BEETLE OUTBREAKS IN TENNESSEE – PRELIMINARY RESULTS

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Abstract—The southern pine beetle (*Dendroctonus frontalis*) is a bark beetle that is native to the Southern United States, including Tennessee. The beetle is periodically epidemic and can cause high levels of mortality during epidemic years, particularly in dense or aging pine (*Pinus* spp.) stands. An epidemic outbreak of the Southern pine beetle occurred in 1999-2001. By 2001, at the peak of the epidemic, 55 counties in Tennessee were in outbreak status. Subsequent estimations suggest that over 350,000 acres of pine timber in the State were affected by the outbreak, causing hundreds of millions of dollars in damage. Given the relative scarcity of the softwood resource in the State compared to the abundance of hardwood species and the significant economic importance of softwoods in Tennessee, the composition and successional trajectory of pine stands impacted by southern pine beetle in the most recent 1999-2001 outbreak is of interest. Here, we measure and quantify the impacts of this southern pine beetle outbreak on the successional trajectory of impacted yellow pine stands. Plots from the Forest Service, U.S. Department of Agriculture Forest Inventory and Analysis Program measured prior, during, and after the outbreak are used to estimate the changes that occurred in southern yellow pine systems within Tennessee. The results from this study suggest that approximately 25 percent of the softwood-dominated forests in Tennessee was lost following the 2000 southern pine beetle event. The majority of that lost acreage transitioned into hardwood-dominated communities.

INTRODUCTION

The southern pine beetle (*Dendroctonus frontalis*) is a bark beetle that is native to the Southern United States, including Tennessee. The beetle is periodically epidemic (Thatcher and others 1980) and can cause high levels of mortality during epidemic years, particularly in dense or aging pine (*Pinus spp.*) stands (Harrington and others 2000). In the southern Appalachians, epidemic outbreaks are common every 10 to 25 years, while areas with warmer winters (Coastal Plain, Piedmont) may experience outbreaks every 7 to 12 years (Price and others 1998, Waldron and others 2007). The Southeastern States, including Tennessee, experienced an epidemic period from 1973 to 1977 that reportedly resulted in the mortality of about 4.5 billion board feet of pine timber (Thatcher and others 1982). Though Tennessee experienced a small outbreak in the mid-1980s, an epidemic outbreak occurred in 1999-2002 that rivaled the destruction of the mid-1970s epidemic (Oswalt and others 2009). By 2001, at the peak of the epidemic, 55 counties in Tennessee were in outbreak status. Subsequent estimates suggest that over 350,000 acres of pine timber in the State were affected by the outbreak, resulting in hundreds of millions of dollars in damage (Cassidy 2004).

Pine species occur throughout Tennessee, but overall the State is dominated by hardwood species (Oswalt and others 2009, Oswalt and others 2012). The most recent complete survey of Tennessee's forests indicated that in 2009 softwood forest acreage was at its lowest in over 50 years, while hardwood acreage continued to increase (Oswalt and others 2012). Although softwood acreage is low in comparison with hardwood acreage, softwood species are of economic and ecological importance and account for 22 percent of all timber product output in the State (Oswalt and others 2012).

Given the relative scarcity of the softwood resource in the State compared to the abundance of hardwood species, and given the significant economic importance of softwoods in Tennessee, the composition and successional trajectory of pine stands impacted by southern pine beetle in the most recent 1999-2002 outbreak (hereafter, 2000 outbreak) are of interest. Casual field observations during and following the 2000 outbreak suggested what appeared to be a large number of shade-tolerant hardwoods (e.g., *Quercus spp.*) in the understory of beetle-infested pine trees. To date, however, no empirical study has attempted to quantify the impact of the 2000 outbreak on the status of softwood-dominated forests in Tennessee.

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Our objective was to quantify changes to Tennessee forests that were softwood dominated prior to the 2000 southern pine beetle outbreak. Specifically, we were interested in quantifying the impact of the 2000 event on softwood forest, how those forests changed, and identifying southern pine beetle as culpable for those changes. We used broad-scale forest inventory data collected by the Forest Service, U.S. Department of Agriculture Forest Inventory and Analysis Program.

METHODS

The Forest Inventory and Analysis (FIA) Program is the primary source for information about the extent, condition, status, and trends of forest resources across all ownerships in the United States (Oswalt and others 2014). FIA applies a nationally consistent sampling protocol using a quasi-systematic design to conduct a multiphase inventory of all ownerships. The national sample intensity is approximately one plot per 6,000 acres of land (Bechtold and Patterson 2005), with plots consisting of four 24-foot fixed-radius subplots spaced 120 feet apart in a triangular arrangement with one subplot in the center (Woudenberg and others 2010). Forested plots, approximately 125,000 in total, are visited every 5 to 7 years in the Eastern United States and every 10 years in the West. Forest land is defined as areas at least 10 percent stocked with tree species, at least 1 acre in size, and at least 120 feet wide (Bechtold and Patterson 2005). Permanent fixed-area FIA inventory plots (approximately one-sixth of an acre in total size) are established in forested conditions when field crews visit plot locations that have accessible forest land. Field crews collect data on more than 300 variables, including ownership, forest type, tree species, and site conditions (Oswalt and others 2014; Woudenberg and others 2010).

Data were obtained from the publicly available Forest Inventory and Analysis Database (FIADB) (Woudenberg and others 2010). All plots measured during the 1999 inventory (using the current annual design) in the East, Plateau, and Central FIA Units (fig. 1) of Tennessee that contained at least one condition identified as softwood or a mixed forest type per FIA definitions (Woudenberg and others 2010) were identified for inclusion in the study. A total of 575 plots met the above criteria (table 1). Data were obtained from every measurement, beginning in 1999, of each of the selected plots through 2013. All plots were measured at least twice, and many plots had three measurements during the period from 1999 to 2013. While all plots were measured during the 1999 inventory (hereafter labeled as Time 1), remeasurement of each plot occurred at a rate of approximately 100-120 plots per year from 2000-2013.

Changes in forest type were tracked and compared across two points in time, the initial Time 1 measurement and the last measurement of the same plot, usually observed between 2009 and 2013 (hereafter labeled as Time 2). Changes in forest type condition proportion (the proportion of the plot accounted for by a particular forest type), softwood species stocking, and softwood species density were calculated using Time 1 and Time 2 measurements. Between Time 1 and Time 2, any recorded disturbance was noted. Paired t-tests were performed to identify any significant changes in condition proportion, softwood stocking, and softwood density through time for each forest type. One-way analysis of variance with Tukey's mean separation was used to identify differences among disturbance types (insects, weather, human, etc.) within given forest types. The R software environment (R Core Team 2013) was used for all statistical analyses.

To further quantify forest changes, plots included in this study were used to generate area estimates by forest type for each of three FIA inventory years, 2003, 2008, or 2012. Area estimates were generated using standard FIA area estimation procedures (Bechtold and Patterson 2005). Forest type changes were tracked across all three times by comparing each plot to its previous measurement within the dataset.

RESULTS AND DISCUSSION

According to estimates based on FIA data from 1999 to 2012, softwood-dominated forests have declined from 1.1 million acres to 848 thousand acres across Tennessee, a decline of 24 percent. At the same time, mixed (softwood and hardwood) forests declined 22 percent from 912 thousand acres to 711 thousand acres.



Figure 1—Forest Inventory and Analysis units in Tennessee.

Table 1—Number of Forest Inventory and Analysis plots measured by year in Tennessee identified as a softwood or mixed forest type

Year	Plots
1999	575
2000	118
2001	121
2002	103
2003	101
2004	121
2005	111
2006	119
2007	101
2008	101
2009	121
2010	108
2011	120
2012	93

Based on plots measured at Times 1 and 2, five softwood-dominated forest types and three mixed forest types experienced significant declines in condition proportion (the area of the plot accounted for by a given forest type) (fig. 2). Concomitantly, no hardwood-dominated forest types declined, and four hardwood-dominated forest types increased in condition proportion. The Yellow poplar/white oak/northern red oak and white oak forest types experienced the largest increases, while the Table Mountain pine, shortleaf pine, and shortleaf pine/oak forest types experienced the largest declines over the period of this study. The Table Mountain pine forest type declined in condition proportion by an average of 64 percent (table 2) per plot between Times 1 and 2. Each of the eight forest types that experienced significant declines shrank in condition proportion by at least 20 percent.

Changes in softwood stem density between Times 1 and 2 were found to be significant for only six forest types. All significant changes were negative (table 3). Mean stem-density decline was largest for the Table Mountain pine forest type (mean = -412 trees per acre). Of the six forest types identified as having significant average declines in stem density, three were softwood dominated (specifically pine dominated) and three were softwood-hardwood mixes.

Softwood species stocking significantly declined over the period of this study for a number of forest types (fig 3). Softwood stocking declines were largest for the Table Mountain pine, followed by eastern hemlock, Virginia pine, shortleaf pine, and loblolly pine forest types. Softwood stocking declined nearly 60 percent in plots where the Table Mountain pine forest type was found at Time 1. Softwood species stocking declines were not as severe for plots containing softwood-hardwood mixed forest types at the Time 1 measurement. However, softwood species stocking declines were still significant for four mixed forest types (fig. 3).

Softwood-dominated forest types experienced losses of approximately 219,000 acres during the early period of this study (approximately 1999-2003) (table 4). Area losses of mixed forests types were even larger, accounting for 272,000 acres during the same period. The largest changes were from mixed and softwood forest types transitioning to hardwood forest types. During the middle years of this study (approximately 2004-2008), considerable losses of softwood and mixed forest types were still occurring. The losses were not as large as during the early years. During the later years of this study (approximately 2009-2012), losses and transitions were essentially absent.

Softwood-dominated forests have declined significantly in the East, Plateau, and Central units of Tennessee following the 2000 southern pine beetle outbreak. In some cases, such as for Table Mountain pine (*Pinus pungens*), the resource was reduced beyond a level detectable by the FIA Program. While it is known through personal observation that Table Mountain pine stands still exist in the State of Tennessee, the area occupied by the Table Mountain pine forest type has declined to the point where it is no longer observed on any FIA plot in Tennessee (Oswalt and others 2012). In addition to the Table Mountain pine forest type, others such as the Virginia pine, shortleaf pine, and loblolly pine forest types have been significantly impacted and altered since 1999, prior to the 2000 southern pine beetle event.

It is clear from this study that many of the forests dominated by softwoods in 1999 no longer exist in the same condition. Many of those stands are now hardwood-dominated stands and are classified as a hardwood forest type. According to Oswalt and others (2012), approximately 1.1 million acres of softwood-dominated forests existed across the three units in this study in 1999. The results from this study suggest that approximately 25 percent of that resource was lost following the 2000 southern pine beetle event. The majority of that lost acreage transitioned into hardwood-dominated communities. Concomitantly,

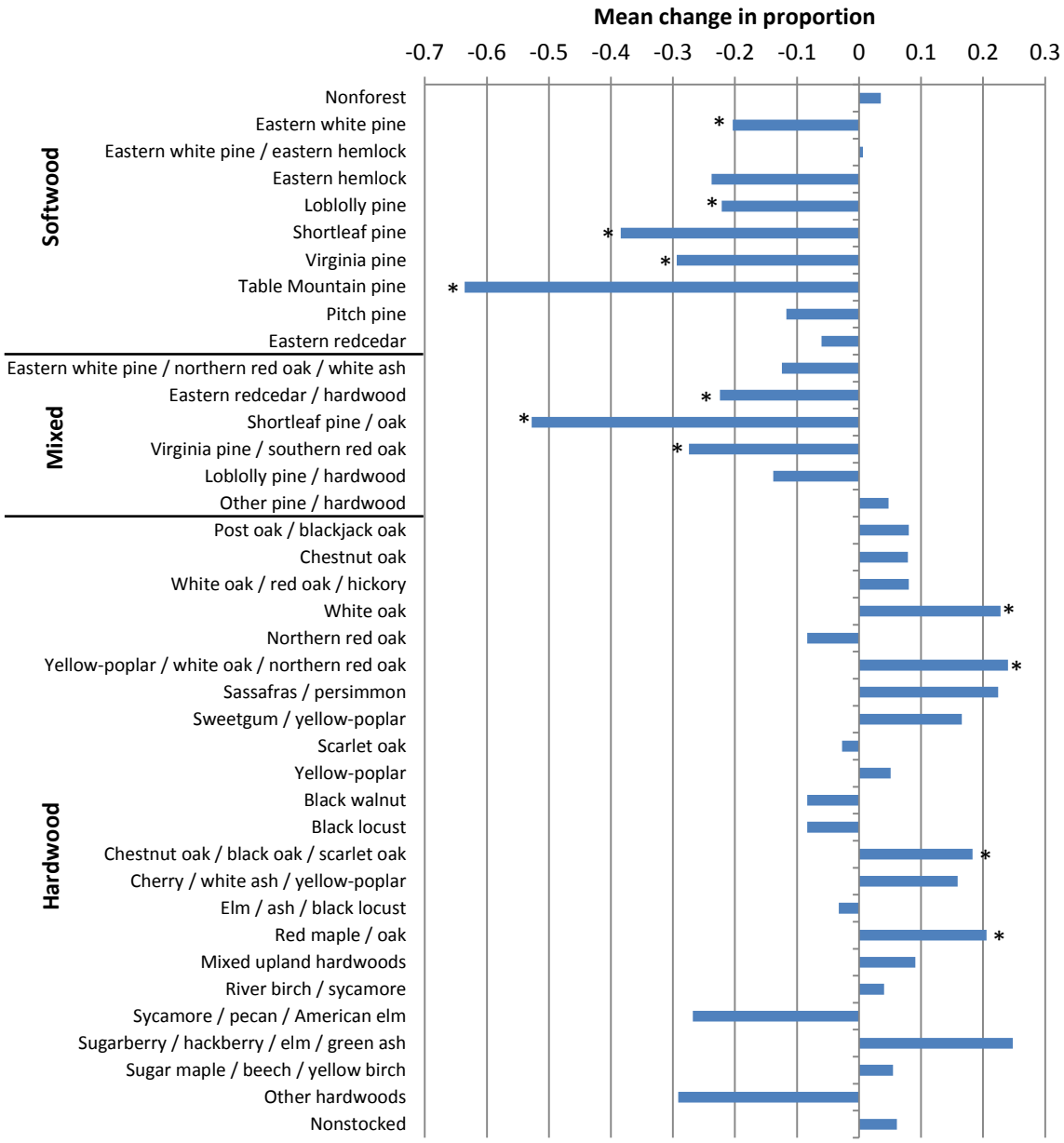


Figure 2—Mean change in condition proportion between 1999 and 2012 by forest type for plots dominated by softwood forest type in 1999 for the East, Plateau, and Central Forest Inventory units in Tennessee.

Table 2—Paired t-test results comparing condition proportion for each forest type for plots dominated by softwood forest type in 1999 for the East, Plateau, and Central Forest Inventory and Analysis units in Tennessee

Forest Type	MeanDiff	SE	t	p
Eastern white pine	-20%	9%	-2.2647	0.0337
Loblolly pine	-22%	7%	-3.0042	0.0039
Shortleaf pine	-38%	12%	-3.2972	0.0027
Virginia pine	-29%	4%	-7.5891	< 0.0001
Table Mountain pine	-64%	16%	-3.908	0.0174
Eastern redcedar / hardwood	-22%	6%	-3.9941	0.0001
Shortleaf pine / oak	-53%	10%	-5.4862	< 0.0001
Virginia pine / southern red oak	-27%	6%	-4.9784	< 0.0001
White oak	23%	10%	2.345	0.0269
Yellow-poplar / white oak / northern red oak	24%	10%	2.4369	0.022
Chestnut oak / black oak / scarlet oak	18%	8%	2.1785	0.0346
Red maple / oak	21%	9%	2.2628	0.0326

SE - standard error; t - t-test; p - p-value

Table 3—Paired t-test results comparing softwood tree density (trees per acre) for each forest type for plots dominated by softwood forest type in 1999 for the East, Plateau, and Central Forest Inventory and Analysis units in Tennessee

Forest Type	MeanDiff	SE	t	p
Shortleaf pine	-98.78	47.0213	-2.1008	0.0451
Virginia pine	-76.729	34.5387	-2.2215	0.028
Table Mountain pine	-414.2	134.529	-3.0789	0.037
Eastern redcedar / hardwood	-69.406	20.6003	-3.3692	0.0011
Shortleaf pine / oak	-62.874	19.2273	-3.2701	0.0024
Virginia pine / southern red oak	-55.102	16.9021	-3.2601	0.0015

SE - standard error; t - t-test; p - p-value

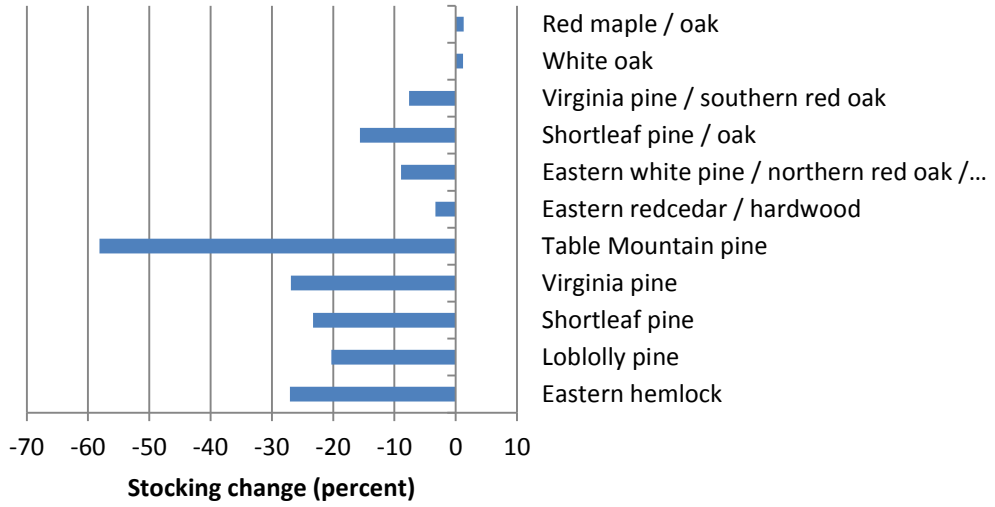


Figure 3—Mean change in relative stocking between 1999 and 2012 by forest type for plots dominated by softwood forest type in 1999 for the East, Plateau, and Central Forest Inventory and Analysis units in Tennessee.

Table 4—Broad forest type transitions for three different time periods for plots dominated by softwood forest type in 1999 for the East, Plateau, and Central Forest Inventory and Analysis units in Tennessee

Forest type transition	2003	2008	2012
Softwood to mixed	10%	6%	5%
Softwood to hardwood	14%	15%	6%
Mixed to hardwood	32%	29%	20%
Hardwood to mixed	8%	12%	9%
Hardwood to softwood	7%	7%	6%
Softwood to mixed	193,862	100,430	73,916
Softwood to hardwood	254,532	239,028	88,152
Mixed to hardwood	352,221	236,343	130,814
Hardwood to mixed	60,749	131,785	124,468
Hardwood to softwood	54,761	72,971	79,633
Softwood (net change)	(218,870)	(100,648)	35,132
Mixed (net change)	(272,373)	(169,967)	(49,997)
Hardwood (net change)	491,243	270,615	14,865

many mixed softwood-hardwood forests shifted to hardwood-dominated forests as well. At this time, these data cannot definitively identify the southern pine beetle as culpable for these changes. However, it can be inferred that the insect was at least one of the primary causes for this rapid shift of forest communities.

LITERATURE CITED

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

Cassidy, P.D. 2004. Dynamics and development of shortleaf pine in east Tennessee. Knoxville, TN: University of Tennessee. 99 p. Ph.D. dissertation.

Harrington, T.B.; Xu, M.; Edwards, M.B. 2000. Structural characteristics of late-successional pine-hardwood forest following recent infestation by southern pine beetle in the Georgia Piedmont, USA. Natural Areas Journal. 20: 360-365.

Oswalt, C.M.; Oswalt, S.N.; Johnson, T.G. [and others]. 2012. Tennessee's forests, 2009. Resour. Bull. RB-SRS-189. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 136 p.

Oswalt, C.M.; Oswalt, S.N.; Johnson, T.G. [and others]. 2009. Tennessee's Forests, 2004. Resour. Bull. SRS–144. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 96 p.

Oswalt, S.N.; Smith, W. B.; Miles, P.D. [and others]. 2014. Forest Resources of the United States, 2012: a technical document supporting the Forest Service 2015 update of the RPA Assessment. Gen. Tech. Rep. WO-91. Washington, DC: U.S. Department of Agriculture Forest Service. 218 p.

Price, T.S.; Doggett, C.; Pye, J.M.; Smith, B. 1998. A history of southern pine beetle outbreaks in the southeastern United States. Macon, GA: Georgia Forestry Commission. 71 p.

R Core Team .2013. R: A language and environment for statistical computing. R foundation for statistical computing, Vienna, Austria. URL http://www.R-project.org/.

Thatcher, .R.C.; Searcy J.L.; Coster, J.E.; Hertel, G.D., eds. 1980. The southern pine beetle. Tech. Bull. 1631. Pineville, LA: U.S. Department of Agriculture Forest Service, Expanded Southern Pine Beetle Research and Application Program, Forest Service, Science and Education Administration. 265 p.

Thatcher, R.C.; Barry, P.J. 1982. Southern pine beetle. Forest insect & disease leaflet 49. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 8 p.

Waldron, J.D.; Lafon, C.W.; Coulson, R.N. [and others]. 2007. Simulating the impacts of southern pine beetle and fire on the dynamics of xerophytic pine landscapes in the southern Appalachians. Applied Vegetation Science. 10: 53-64

Woudenberg, S.W.; Conkling, B.L.; O'Connell, B.M. [and others]. 2010. The forest inventory and analysis database: database description and users manual version 4.0 for Phase 2. Gen. Tech. Rep. RMRS-GTR-245. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 336 p.

OAK DECLINE ACROSS THE OZARK HIGHLANDS – FROM STAND TO LANDSCAPE AND REGIONAL SCALE PROCESSES

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Abstract—Oak decline has been a problem in forests of the Ozark Highlands (OzH) for decades. It has impacted upland oak-hickory forests, particularly species in the red oak group (*Quercus* section *Lobatae*) across the Ozark Highlands of Missouri, Arkansas, and Oklahoma. The oak decline complex is often described in terms of predisposing factors, inciting factors, and contributing factors. Drought is a common inciting factor in oak decline, while advanced tree age is considered a predisposing factor, and opportunistic organisms such as armillaria root fungi and wood boring insects are believed to contribute to the decline and demise of formerly stressed trees. Declining trees are initially indicated by foliage wilt and browning followed by progressive branch dieback. If crown dieback continues, trees can die. In this paper we synthesize four of our key research studies on oak decline, examining the occurrence, distribution, and characteristics of oak decline as it has impacted the OzH across space and time. Long-term climate forecasts for this region indicate decreasing precipitation and warming temperatures. Consequently, periodic droughts such as the widespread 2012 U.S. drought are expected to increase in frequency and intensity, and thereby exacerbate oak decline on millions of hectares of aging oak forests. Results from our research indicate that regular monitoring of forest conditions; increasing the proportion of species in the white oak relative to the red oak group; judicious application of prescribed fire; periodic thinning to favor species in the white oak group; and proactive harvest of aging red oak species anticipated to be at increased risk of mortality are methods that can help forest managers mitigate oak decline.

INTRODUCTION

Oak-hickory forests cover the largest area and contribute the greatest amount of biomass of any forest-type group in the Eastern United States (table 1). During the period 1856 to 1986, some 57 major oak mortality events were recorded across those forests (Millers and others 1989). The Ozark Highlands ecoregion (OzH) (McNab and Avers 1995, McNab and others 2005), located mainly in southern Missouri and northern Arkansas, contains some of the highest concentrations of oak with the mean basal area contributed by *Quercus* species estimated at 63percent (Oak and others 2015).

Widespread oak decline events of the OzH that began in 2000 have been particularly severe and continue today. Drought from 1998 through 2000 incited this oak decline episode. Drought has been the inciting factor for decline in the OzH in the past, and is most often cited as the predominant inciting factor for decline throughout the Southeastern United States (Millers and others 1989). Of the oak species, those in the red oak group (*Quercus* section *Lobatae*) are the most susceptible to decline while those in the white oak group (*Quercus* section *Quercus)* are relatively resilient during decline events (Fan and others 2012). The higher mortality of red oak species may be due in part to their high abundance on dry, rocky sites owing to widespread establishment on these areas following the extensive, exploitive logging of shortleaf pine in the early 1900s (Kabrick and others 2008). The past disturbance history of the Ozark Highlands has resulted in a disproportionate area of old forests that are susceptible to oak decline

Table 1—Biomass of Eastern U.S. forests by forest-type group. Oak dominated forests contribute 53 percent of the biomass of all of the forests in the eastern U.S. FIA assessment area (based on Oswalt et al. 2014)

Forest-Type Group	Biomass	Area
Oak-Hickory	39%	34%
Oak-Gum-Cypress	8%	6%
Oak-Pine	6%	7%
Total	**53%**	**47%**

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events. Nearly 60 percent of forests in this region are greater than 60 years old, an age cohort considered predisposed to oak decline. An additional 27 percent of OzH forests are between 40 and 60 years of age (Miles 2015). This situation assures that mortality associated with oak decline will be an important management issue for decades to come. Concern about the current and anticipated extent and severity of oak decline in the Ozark Highlands has prompted renewed collaborations, most notably the convergence of over 350 scientists, land managers, and landowners and the formation of the Upland Oak Ecology Symposium in 2002 (Spetich 2004b).

From a broader perspective, a recent paper on global drought and heat-induced tree mortality associated with climate change suggests that the composition and structure of forests worldwide may be altered by increases in heat and drought stress (Allen and others 2010). The authors suggest that some of the world's forests may already be responding to these changes in climate. Further, they conclude that if widespread tree mortality transpires there will be considerable impact to society and to the global ecosystem. Some climate models suggest that from 2000 to 2060 temperature in the OzH is expected to increase by about 3 degrees centigrade and precipitation is expected to decrease by at least 200 mm (McNab and others 2014), which will continue to predispose the OzH to future oak decline events.

In this paper, we synthesize four of our key OzH research studies on oak decline. These studies cover three spatial scales—stand, landscape, and region—examining past events and projecting conditions into the future. We examine the occurrence, distribution, and characteristics of oak decline as it has impacted the OzH across space and time, and we provide management suggestions aimed at mitigating future episodes of oak decline.

SITES

The Ozark Highlands ecological section contains over 12 million ha (30 million acres) and covers parts of southern Missouri, northern Arkansas, and a small portion of northeastern Oklahoma (McNab and Avers 1995, McNab and others 2005). The following field studies are based on data from Forest Inventory and Analysis (FIA) inventories (USDA Forest Service 2015) and other research plots across the OzH ecoregion in Missouri and Arkansas. These four studies cover three spatial scales: forest stand, landscape, and regional.

Stand Scale, Study 1:

The study site is a 32-ha area in an upland oak-hickory stand which was 74 years old in 2004 and located on a southwest facing slope with dry, rocky soil. The forest stand is dominated by oak (*Quercus* spp.) and hickory (*Carya* spp.) and was the center of a local patch of oak decline beginning in 2001. The site was undisturbed for more than 20 years prior to 2000 and had no deer browsing pressure on seedlings during the study.

Landscape Scale, Studies 2 and 3:

These two studies encompass a 427,660-ha area (1,056,771 acres) within the Ozark National Forest purchase boundaries and include three ranger districts: Bayou, Buffalo, and the Pleasant Hill (since the studies were completed, the Buffalo and Bayou ranger districts were combined to form the Big Piney ranger district). Dominant species are oak and hickory.

Regional Scale, Study 4:

This study included all forest within the OzH within Arkansas and Missouri that contained an oak component.

PROCEDURES

We examined dynamics of these forests at three spatial scales: forest stand (32 ha), intermediate landscape (428 thousand ha), and regional (12 million ha; the entire OzH across Arkansas and Missouri).

Stand Scale, Study 1 – regeneration:

In the winter of 2000, 24 overstory plots (0.16 ha) were established with 20 regeneration measurement plots (5.4 m^2) nested within each overstory plot (480 total regeneration plots). In the summer of 2000, trees < 5 cm diameter at breast height (dbh) were inventoried for all 480 regeneration plots. In the summer and early fall of 2000, standing trees were inventoried on all midstory (5- to 25-cm dbh trees) and overstory ($\geq$ 25-cm dbh trees) plots, and in the fall of 2001, 6 of the 24 overstory plots were remeasured. On March 12, 2004, prescribed fire was applied to one fourth of the site and encompassed 120 of the regeneration plots. Then in 2005 and again in 2008, all 480 regeneration plots were remeasured. Detailed methods of the study are described in Spetich (2004a, 2006, 2013, and 2014).

Landscape Scale, Study 2 – harvest methods:

The landscape disturbance model LANDIS PRO (He 2015) was used to simulate three harvest methods (clearcutting, thinning, and group-selection), and model long-term, landscape-scale effectiveness of treatments on oak decline mitigation 100 years into the future. During the simulated harvests, older red oaks were given priority for harvesting. Detailed study methods are described by Wang and others (2013).

Landscape Scale, Study 3 – prescribed fire:

The landscape disturbance model LANDIS (He and others 2005) was used to simulate the impact of two fire regimes on future oak decline risk 150 years into

the future over a 427,660-ha area. Additional details on study methods are provided by Spetich and He (2008).

Regional Scale, Study 4 – drought:

Spatial and temporal trends of oak decline and mortality across the Ozark Highlands were assessed using FIA data (USDA Forest Service 2015) from 1999 to 2010. We used data from trees on 6,997 spatially referenced FIA plots across the Ozark Highlands of Arkansas and Missouri. On each 0.4-ha plot, trees > 11 cm dbh were measured and periodically remeasured from 1999 to 2010. For analysis, the tree species were divided into three groups: white oak group, red oak group, and a non-oak species group. The percentage of dead trees was calculated in terms of basal area. Cross-correlation analysis was used to examine the relationship of growing season Palmer drought severity index (PDSI) to oak mortality (Palmer 1965). Additional details about methodology are available in Fan and others (2012).

RESULTS AND DISCUSSION

Stand Scale, Study 1:

Within one year, 2000 to 2001, the number of standing dead trees more than doubled from 23 trees per hectare to 51 trees per hectare ($p = 0.029$) for northern red oak *Quercus rubra* (Spetich 2004a). However, many of these trees were below the main canopy. This understory mortality is likely due to the fact that this stand was in the stem exclusion phase of stand development (Oliver and Larson 1990), putting this cohort of trees in a highly competitive condition prior to drought and the subsequent stand decline. Understory mortality on oak decline sites has been noted in at least one other decline study (Heitzman and others 2007).

One year after the 2004 prescribed fire, northern red oak regeneration in the unburned area had greater probability of survival in all diameter classes compared to the burned area. However, by 2008 this relationship changed where survival probability in the burned area remained relatively stable over all regeneration diameter classes, while in the unburned area there was a decrease in survival probability across all regeneration basal stem diameter classes. This decrease is likely due to a greater amount of competing vegetation remaining in the unburned area. In the burned area, the number of regeneration trees was 28,587 per hectare, while in the unburned area it was 44,718 trees per hectare. By 2008, seedlings greater or equal to a 6-mm basal stem diameter in the burned area had greater survival probability than regeneration in the unburned area (fig. 1).

For white oak (*Quercus alba)*, stem age (measured at the beginning of the study in 2000) was used to predict survival because basal diameter is not a useful predictor of regeneration survival for this species in the Ozarks due to its slow diameter growth rate at this stage of tree development (Spetich 2014). Survival probabilities of white oak regeneration remained relatively stable for all inventory years and stem age classes (1 to 15 years), and this is likely due to greater shade tolerance of white oak compared to northern red oak. In all cases, survival began to reach a maximum and level off at a stem age of 5 years with survival ranging from 74 to 78 percent at stem age 5, and 81 to 85 percent for stem age 15 (fig. 2). For detailed results of study 1, see Spetich (2004a, 2006, 2013, and 2014).

Landscape Scale Simulation, Study 2 – harvest methods:

By simulation year 100, on high risk sites, the three harvesting methods (clearcutting, thinning, and group-selection) reduced the percent of oak decline on the landscape by only three percent below that of doing nothing. On medium-risk sites, all three harvesting methods reduced the percent of oak decline on the landscape by 13 percent below that of doing nothing. However, the areas of potential oak decline decreased during the first five decades of simulation regardless of harvest method used, indicating that natural species dynamics and succession will also play a strong role in reducing the susceptibility of these forests to oak decline. For detailed results of study 2, see Wang and others (2013).

Landscape Scale Simulation, Study 3 – prescribed fire:

With prescribed fire administered on only half of the study area during the 150-year simulation period, all potential oak decline sites were reduced from covering 45 percent of the landscape at year 0, to 30 percent at year 150. By increasing prescribed fire to three times across the entire area during the 150-year period, all potential oak decline sites were reduced to only 20 percent of the landscape. High risk sites were reduced from 20 to 7 percent, medium risk sites from 15 to 7 percent, and low risk from 10 to 6 percent with three fires. The future impact of oak decline was alleviated as more frequent fire diminished red oak regeneration, thus shifting stand-level species dynamics. The result was a landscape with greater resistance to decline due to the replacement of red oak species with the more decline-resistant white oak species. For detailed results of study 3, see Spetich and He (2008).

Regional Scale, Study 4 – drought:

Beginning one year after the instigating drought, the proportion of dead basal area of white oak species remained relatively stable and was comparable to non-oak species from 1999 to 2010, with both white and non-oak averaging approximately 5 percent of dead basal area. The proportion of dead basal area was not significantly different than non-oak in at least eight of the twelve years. In fact, during three of the years in

which dead basal area differed more than one standard error (1999, 2008, and 2009), white oak mortality was lower than non-oaks.

However, dead basal area of red oak was significantly greater than white oak and non-oaks from 2000 to 2010, starting at a low of 8 percent dead basal area in 1999 and increasing to approximately 15 percent of dead basal area by 2008. Cross-correlation analysis revealed a significant relationship between red oak mortality and growing season PDSI. This indicated a two- to three-year lag of mortality after single droughts and a cumulative impact of droughts of up to ten years. For additional details, see Fan and others (2012).

Recommendations for Forest Land Managers:

Harvesting mature stands without regard to species is not likely to fully mitigate oak decline on high risk sites. However, over time even low frequency of prescribed fires could significantly decrease the area of potential oak decline. Time for a measurable response resulting from prescribed burning is likely measured in decades, because the change is based on the growth and succession of forest tree species. Prior to burning, managers should assess oak regeneration for sufficient density to meet site specific management objectives and locate stands with white oaks that are at least five years old. White oaks are more resistant to oak decline than red oaks, and white oaks will be favored

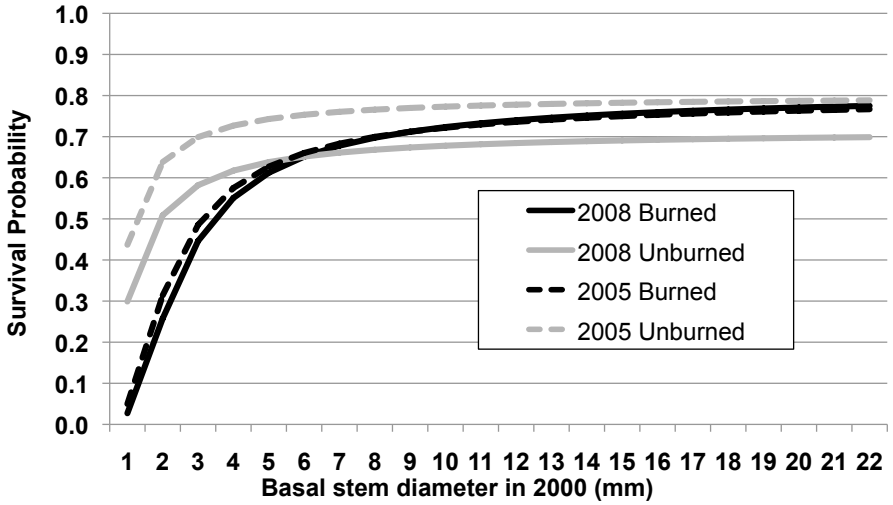


Figure 1—Study 1, oak regeneration. Red oak survival probability in the burned vs. unburned areas in 2005 and again in 2008.

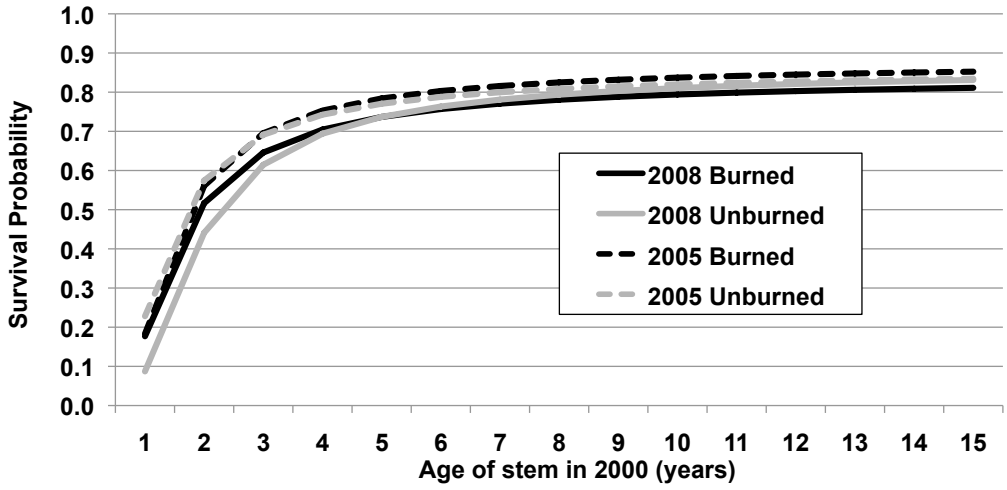


Figure 2—Study 1, oak regeneration. White oak survival probability in the burned vs. unburned areas in 2005 and again in 2008.

by prescribed burning in stands where white oak stems are mostly five years of age or older and red oaks are mostly smaller than 6 mm in basal diameter.

In light of a climate with increasing drought, increasing temperatures, and the lag time required for the impact of cumulative droughts on oak decline in the Ozarks to materialize, managers should consider proactive treatments for established oak stands. However, in study 2 above and in an applied harvesting study (Dwyer and others 2007), the mature stands that were harvested showed either modest or no impact on decline, so earlier intervention may be necessary, especially on high risk sites. In younger stands, this might be accomplished by harvesting red oak species on high risk sites before the trees reach a decline-susceptible age in order to help mitigate future oak decline effects. In high risk mature stands, it may be prudent to harvest most of the mature red oaks in a single treatment entry because sustaining red oaks through multiple thinning treatments is difficult on those sites. It is at maturity that red oak becomes most susceptible to oak decline (Manion 1991, Starkey and others 2004). Decline-resistant species like white oak could then be grown to rotation age while working to establish more white oak regeneration. The harvest of red oaks on decline-prone sites would likely be an intermediate harvest or a stand improvement harvest to address this forest health issue. Further, where there are limited resources, potential hot spots for oak decline can be identified through landscape-scale modeling [e.g., the LANDIS PRO model (He 2015)] so that limited forest management funding could be utilized most effectively.

Future studies should examine a combination of the above two methods by (a) using harvest methods to begin changing the overstory and understory species composition, and (b) applying prescribed fire to further encourage establishment and survival of decline-resistant species.

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LITERATURE CITED

Allen, C.D.; Macalady, A.K.; Chenchouni, H. [and others]. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management. 259(4): 660-684.

Dwyer, J.P.; Kabrick, J.M.; Wetteroff, J. 2007. Do improvement harvests mitigate oak decline in Missouri Ozark forests? Northern Journal of Applied Forestry. 24(2): 123-128.

Fan, Z.; Fan, X.; Crosby, M.K.; Moser W.K.; He, H.S.; Spetich, M.A.; and Shifley, S.R. 2012. Spatio-temporal trends of oak decline and mortality under periodic regional drought in the Ozark Highlands of Arkansas and Missouri. Forests. 3: 614-63.

He, H.S.; Li, W.; Sturtevant, B. R. [and others] 2005. LANDIS, a spatially explicit model of forest landscape disturbance, management, and succession—LANDIS 4.0 User's guide. North Central Research Station General Technical Report NC-263. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Research Station. 93 p.

He, H.S. 2015. LANDIS: A forest landscape model of succession, disturbance, and management. http://landis.missouri.edu/landispro70.php. [Date accessed: September 3, 2015].

Heitzman, E.; Grell, A.; Spetich, M.A.; and Starkey, D. 2007. Changes in forest structure associated with oak decline in severely impacted areas of northern Arkansas. Southern Journal of Applied Forestry. 31(1): 17-22.

Kabrick, J.M.; Dey, D.C.; Jensen, R.G.; Wallendorf, M. 2008. The role of environmental factors in oak decline and mortality in the Ozark Highlands. Forest Ecology and Management. 255(5): 1409-1417.

Manion, P.D. 1991. Tree Disease Concepts. Englewood Cliffs, NJ: Prentice Hall, Inc. 399 p.

McNab, W.H.; Avers, P.E., comps. 1995. Ecological subregions of the United States: section descriptions. Admin. Publ. WO-WSA-5. 1994. Washington, DC: U.S. Department of Agriculture Forest Service. 267 p.

McNab, W.H.; Cleland, D.T.; Freeouf, J.A. [and others], comps. 2005. Description of ecological subregions: sections of the conterminous United States [CD-ROM]. Washington, DC: U.S. Department of Agriculture Forest Service. 80 p.

McNab, W.H.; Spetich, M.A.; Perry, R.; Clark, S. 2014. Climate induced migration of native tree populations and consequences for forest composition. In: Vose, J.M.; Klepzig, K., eds. Climate change adaptation and mitigation management options: a guide for natural resource managers. Boca Raton, FL: CRC Press: 307-378. Chapter 11.

Miles, P.D. 2015. Forest Inventory EVALIDator web-application version 1.6.0.02. St. Paul, MN: U.S. Department of Agriculture Forest Service, Northern Research Station. Available only on internet: http://apps.fs.fed.us/Evalidator/evalidator.jsp. [Date accessed: September 4, 2015].

Millers, I.; Shriner, D.S.; Rizzo, D. 1989. History of hardwood decline in the Eastern United States. Gen. Tech. Rep. NE-126. Broomall, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 79 p.

Oak, S.W.; Spetich, M.A.; Morin, R.S. 2015. Oak decline in central hardwood forests: frequency, spatial extent, and scale. In: Greenberg, C.H.; Collins, B.S., eds. Natural disturbances and historic range of variation: type, frequency, severity, and post-disturbance structure in Central Hardwood Forests, USA.: Managing Forest Ecosystems, 2015, Vol. 32. 400 p.

Oliver, C.D.; Larson, B.C. 1990. Forest stand dynamics. New York: McGraw-Hill, Inc. 467p.

Oswalt, S.N.; Smith, W.B.; Miles, P.D.; Pugh, S.A. 2014. Forest resources of the United States, 2012: a technical document supporting the Forest Service 2015 update of the RPA Assessment. Gen. Tech. Rep. WO-91. Washington, DC: U.S. Department of Agriculture Forest Service, Washington Office. 218 p.

Palmer, W.C. 1965. Meteorological drought. Research Paper No. 45. Washington, DC: Department of Commerce Weather Bureau. 58 p.

Spetich, M.A. 2004a. Forest dynamics at the epicenter of an oak decline event in the Boston Mountains, Arkansas – year one. In: Yaussy, D.A.; Hix, D.M.; Long, R.P.; Goebel, P.C., eds. Proceedings of the 14th central hardwood forest conference. Gen. Tech. Rep. NE-316. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station. 21-26 [CD-ROM].

Spetich, M.A. 2004b. Upland oak ecology – a synthesis. In: Spetich, M.A., ed. Proceedings of the upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service; Southern Research Station: 3-9.

Spetich, M.A. 2006. Early changes in physical tree characteristics during an oak decline event in the Ozark Highlands. In: Connor, K.F., ed. 2006. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 424-427.

Spetich, M.A.; He, H.S. 2008. Oak decline in the Boston Mountains, Arkansas, USA: spatial and temporal patterns under two fire regimes. Forest Ecology and Management. 254: 454–462.

Spetich, M.A. 2013. Regeneration dynamics during oak decline and prescribed fire in Arkansas' Boston Mountains. In: Guldin, J.M., ed. Proceedings of the 15th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-175. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 255-259.

Spetich, M.A. 2014. Regeneration dynamics during oak decline in Arkansas. In: Groninger, J.W.; Holzmueller, E.J.; Nielsen, C.K.; Dey, D.C., eds. Proceedings of the 19th central hardwood forest conference. Gen. Tech. Rep. NRS-P-142. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 195-197.

Starkey, D.A.; Oliveria, F.; Mangini, A.; Mielke, M. 2004. Oak decline and red oak borer in interior highlands of Arkansas and Missouri: natural phenomena, severe occurrences. In: Spetich, M.A., ed. Proceedings of the upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 217–222.

U.S. Department of Agriculture (USDA) Forest Service. 2015. Forest Inventory and Analysis national program. http://www.fia.fs.fed.us/. [Date accessed: September 3, 2015].

Wang, W.J.; He, H.S.; Spetich, M.A. [and others]. 2013. Modeling the effects of harvest alternatives on mitigating oak decline in a central hardwood forest landscape. PLoS ONE. 8(6): e66713. 10 p. doi:10.1371/journal.pone.0066713. [Published online: 18 June 2013].

Threats
(Continued)

Moderator:

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MODELING STEM PROFILE OF *TRIADICA SEBIFERA* IN SOUTHERN FORESTLANDS OF MISSISSIPPI

Nana Tian and Zhaofei Fan[1]

Abstract—Chinese tallow (*Triadica sebifera*) is one of the most aggressive invasive species in the southern forestlands of United States. To explore the stem taper of tallow, outside-and-inside bark stem profile equations were fitted using Max and Burkhart (1976), Cao (2009) modified Max and Burkhart, and Clark, Souter, and Schlaegel (1991) segmented polynomial models. Sample trees were collected from oak-gum-cypress (*Quercus/Liquidambar styraciflua/Taxodium distichum*) and longleaf/slash pine (*Pinus taeda/ Pinus echinata*) forests in southern Mississippi using destructive sampling method. Results showed that: 1) Clark, Souter, and Schlaegel (1991) segmented polynomial model was the best fitted model for both DOB and DIB stem profile of tallow in these two forestlands; 2) the stem of tallow was generally sturdy in oak-gum-cypress forest while it was slender in longleaf/slash pine forest; however, no significant difference was found. Those models provide a tool for managers to project future growth stocking of tallow accurately and make management decision.

INTRODUCTION

The Chinese tallow tree (Triadica sebifera (L.)) in the spurge family (Euphorbiaceae) is a monoecious, deciduous tree, native to central and southern China (Zhang and others 1994). Since its introduction as an ornamental and potential oil producing species in the 1770s, it has become an invasive species in the southern forestlands of the United States (Bruce, 1993). Previous study recorded that approximately 185,000 acres of southern forests had been invaded by tallow tree (Tan, 2012) especially in the coastal prairies and plains (Gan and others 2009). Fast growth and spread are critical factors for colonization and establishment of Chinese tallow in the affected regions.*trst*

Stem growth described by profile functions is an important indicator of tree growth. Taper equations are the mathematical function of the diameter change with respect to height on the basis of species, age, and stand condition (Husch and others 1982; Brooks and others 2008). Numerous taper functions have been developed for different tree species with various forms from simple ones (Kozak and others 1969, Ormerod 1973, Hilt 1980, Zakrzewski and others 2006) to complex (Max and Burkhart 1976, Cao and others 2009 Clark and others 1991, Jiang and others 2005). Methol (2001) classified taper equations into four categories: single functions, within-tree variable form, between-tree variable form, and segmented polynomial models; moreover, Jiang and others (2005) summarized them into three classifications including simple taper functions, variable form taper functions, and segmented polynomial taper functions.

Simple taper functions mainly define tree profiles with a single continuous equation for the whole bole (Bruce and others 1968, Hilt 1980, Gordon and others 1995). However, these simple taper equations were unable to precisely describe the whole bole profile, although they could reflect the general stem form (Jiang and others 2005). Max and Burkhart (1976) developed a segmented polynomial regression method to build a profile equation for loblolly pine (*Pinus taeda*). Clark and others (1991) found it as a better performer in predicting diameter. Likewise, Sharma and Burkhart (2003) described it as the combination of three sub-models at two join points. However, the application of Max and Burkhart model is limited to a small range of diameters and heights because of the complexity (Matney and Parker 1992, Parker 1997). Cao (2009) modified the Max and Burkhart loblolly pine model by calibrating DBH and upper stem diameter. Then Clark and others (1991) developed a form-class segmented profile model which provided volume estimation more accurately than the segmented Max and Burkhart model. Souter (2003) employed a segmented profile function in southern tree species to predict diameter at a specified height. Overall, segmented polynomial models composed of a

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series of sub-models representing various sections of the stem are better than simple and variable form taper functions and are widely used (Ounekham, 2009).

To compare the stem taper of tallow in different forestlands in southern Mississippi, a group of profile models were constructed using Max and Burkhart (1976), Cao (2009) modified Max and Burkhart, and Clark, Souter, and Schlaegel (1991) segmented polynomial models.

METHODS

Data Collection

In total, 33 sample trees were collected in this study and 11 of them were from the oak/gum/cypress bottomland forest in southern Mississippi (Old River Wildlife Management Area in Poplarville city, Pearl River County, MS); 16 were sampled from the Grand Bay National Estuarine Research Reserve (NERR) (Jackson County, MS) and an additional 6 trees were obtained from the Mississippi Sandhill Crane National Wildlife Refuge (NWR) (Jackson County, MS) in the Gulf Coast Complex. The Grand Bay and Sandhill are primarily longleaf/slash pine forests.

Destructive sampling was used to obtain the diameter along different height of the stem. Before felling sample trees in the field, total height and diameter at breast height (DBH) was measured. Trees were then cut (to a stump height of approximately 10 cm) and the stem was divided into 1 m sections. Disks with 3-5 cm thickness were then extracted from the midpoint of each section. The diameter (inside bark and outside bark) was obtained at the upper end of each section. Diameter at selected height positions (0.8 m and 5.3 m) was also recorded.

Analysis

Three segmented polynomial approaches of Max and Burkhart (1976) (Equation 1), Cao (2009) (Equation 2), and Clark, Souter, and Schlaegel (1991) (Equation 3) were selected to fit the stem profile of tallow and they all fitted using TProfile (Matney 1992).

$$\frac{d}{D} = \left\{\left(b_1\left(\frac{h}{H}-1\right)+b_2\left(\frac{h^2}{H^2}-1\right)+b_3\left(\alpha_1-\frac{h}{H}\right)^2 I_1 + b_4\left(\alpha_2-\frac{h}{H}\right)^2 I_2\right)\right\}^{0.5} \quad (1)$$

Where: H indicates the total height (m) whereas D represents DBH (cm); d is diameter at height h (cm) and h is height above the ground to the measurement point (m). The indicate condition of this model is: $h = H$, $d = 0$, *and* $h = 1.37$, $d = D$. I_1=1 if $\frac{h}{H} \le a_1$, and $I_1 = 0$, otherwise; $I_2 = 1$ if $\frac{h}{H} \le a_2$, and $I_2 = 0$, otherwise. b_1, b_2, b_3, b_4, a_1, a_2 are all parameters to be estimated.

$$y^*\left(1-\frac{h}{H}\right) = b_1^* \times \left(1-\frac{h}{H}\right) + b_2(1-\frac{h}{H})^2 + b_3 \times I_1(1-\frac{h}{H}-a_1)^2 + b_4 \times I_2(1-\frac{h}{H}-a_2)^2 \quad (2)$$

$$b_1^* = \frac{y\left(1-\frac{1.37}{H}\right)-\hat{y}\left(1-\frac{1.37}{H}\right)+b_1\times(1-\frac{1.37}{H})}{(1-\frac{1.37}{H})}$$

Where: $y^* = d^{*2}/D^2$ and d^* is calibrated diameter; the parameter b_1 is modified to parameter b_1^*. The other variables had the same meaning with Max and Burkhart model.

$$d = \left\{I_S\left\{D^2\left(1+\frac{\left(b_2+\frac{b_3}{D^3}\right)\left(\left(1-\frac{h}{H}\right)^{b_1}-\left(1-\frac{1.37}{H}\right)^{b_1}\right)}{\left(1-\left(1-\frac{1.3}{H}\right)^{b_1}\right)}\right)\right\} + I_B\left\{D^2-\frac{(D^2-F^2)\left(\left(1-\frac{1.37}{H}\right)^{b_4}-\left(1-\frac{h}{H}\right)^{b_4}\right)}{\left(\left(1-\frac{1.37}{H}\right)^{b_4}-\left(1-\frac{5.3}{H}\right)^{b_4}\right)}\right\}^{0.5} + I_T\{F^2\left(b_6\left(\frac{h-5.3}{H-5.3}-1\right)^2+I_M\left(\frac{1-b_6}{{b_5}^2}\right)\left(b_5-\frac{h-5.3}{H-5.3}\right)^2\right)\}\} \quad (3)$$

Where: F is the diameter at 5.3 m, b_1,b_2,b_3 is the regression coefficient for butt section, b_4 is the coefficient for lower stem, and b_5, b_6 are the coefficients for height above 5.3 m. Hence, the four indicator variables in Equation (3) are defined as following:

$$I_S = \begin{cases}1 & h < 1.37\\ 0 & otherwise\end{cases}$$

$$I_B = \begin{cases}1 & 1.37 < h < 5.3\\ 0 & otherwise\end{cases}$$

$$I_T = \begin{cases}1 & h > 5.3\\ 0 & otherwis\end{cases}$$

$$I_M = \begin{cases}1 & h < (5.3 + b_5(H-5.3\\ 0 & otherwise\end{cases}$$

Final best fitted taper model selection was evaluated by root mean squared error (RMSE) (Equation 4) and a fit index (FI) (Equation 5) (Schlaegel 1981).

$$RMSE = \sqrt{\bar{e}^2 + SD^2} \quad (4)$$

$$FI = 1 - [\frac{\sum_{i=1}^{n}(Y_i-\hat{Y}_i)^2}{\sum_{i=1}^{n}(Y_i-\bar{Y})^2} \quad (5)$$

Where: SD is standard deviation of the predicted errors, e_i is the prediction error or difference between observations (Y_i) and the predictions $(\hat{Y}_i)$; $\bar{Y}$ is the mean of Y_i.

RESULTS

Both DOB and DIB profile models were fitted and for the inside-bark taper fitting process, DIB replaced DOB with the other variables being the same as with the DOB fitting process. Fitted profile models were presented in fig. 1. In addition, RMSE and FI showed that all three models performed well for DOB profile of tallow in oak/gum/cypress forest at Poplarville although Clark and others (1991) (RMSE = 0.052, FI = 0.974) was better than the Max and Burkhart taper model (RMSE = 0.060, FI = 0.964) and Cao (2009) modified Max and Burkhart model (RMSE = 0.060, FI = 0.964). However, no significant difference was found. Likewise, for DOB of tallow in longleaf/slash pine forest at Grand Bay and Sandhill, still no distinct difference was found among three profile models though Clark and others (1991) performed better (RMSE = 0.069, FI = 0.947) than Max and Burkhart (RMSE = 0.088, FI = 0.912) and the Cao (2009) modification model (RMSE = 0.090, FI = 0.910) according to RMSE and FI. Regarding the DIB profile models of tallow at Poplarville, there was also no evidence of difference among the three fitted models: Max and Burkhart (RMSE = 0.057, FI = 0.962), Cao (2009) modification model (RMSE = 0.061, FI = 0.956), and Clark and others (1991) (RMSE = 0.057, FI = 0.968). Nevertheless, Clark and others (1991) (RMSE = 0.078, FI = 0.930) performed better than the other two models which exhibited the same performance (RMSE = 0.088, FI = 0.880) for DIB profile model of tallow at longleaf/slash pine forest.

DISCUSSION AND CONCLUSIONS

Clark and others (1991) had better performance for both DOB and DIB profiles of tallow in oak/gum/cypress forest at Poplarville, but there was no significant difference among the three fitted profile equations. Similarly, Clark and others (1991) was also the best fitted model for tallow in longleaf/slash pine at Grand Bay and Sandhill as compared with the Max and Burkhart (1976) and the Cao (2009) modified Max and Burkhart models. Studies (Larson 1965, Garber and Maguire 2003, Bluhm and others 2007) have reported that stem profile varied with stand conditions. However, in this study there is no obvious difference for both DOB and DIB profile of tallow in the two different coastal forests. The importance of profile equations in improving the estimation of volume/biomass for managing and valuing

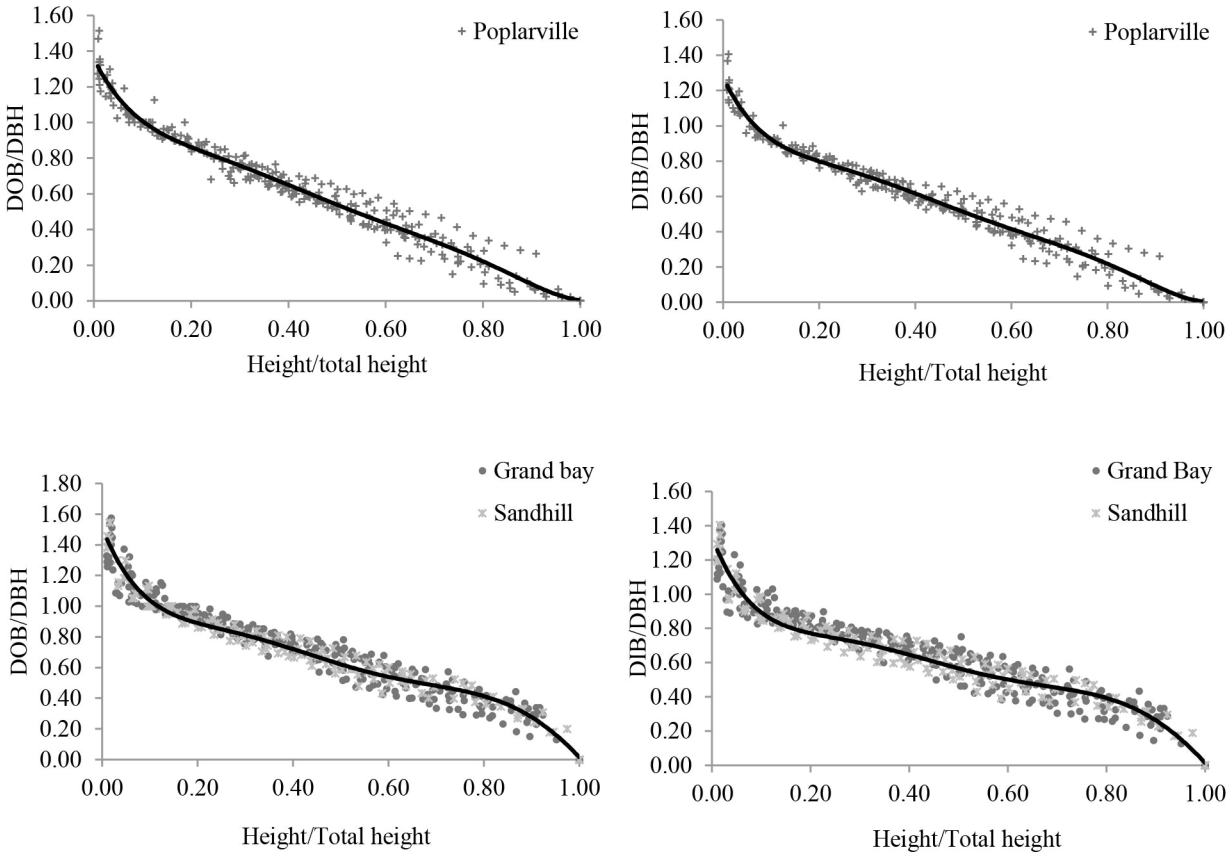


Figure 1—Plots of observed stem profile (DOB and DIB) at Poplarville, Grand Bay, and Sandhill.

forests and the difficulty of selecting an appropriate model that works well for multiple species and diverse site conditions makes the equations an exploited research topic (Clutter and others 1983, McClure and others 1986, Muhairwe 1999).

LITERATURE CITED

Bluhm, A.A.; Garber, S.M.; Hibbs, D.E. 2007. Taper equation and volume tables for plantation-grown red alder. PNW-GTR-735. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 74 p.

Bruce, K.A. 1993. Factors affecting the biological invasion of the exotic Chinese tallow tree, *Sapium sebiferum*, in the gulf coast prairie of Texas. Houston, TX: University of Houston. 155 p. M.S. thesis.

Brooks, J.R.; Jiang, L.; OzÇelik, R. 2008. Compatible stem volume and taper equations for Brutian pine, Cedar of Lebanon, and Cilicica fir in Turkey. Forest Ecology and Management. 256: 147-151.

Bruce, D.; Curtis, R.O.; VanCoevering, C. 1968. Volume and taper tables for red alder. Res. Pap. PNW-56. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 39 p.

Cao, Q.V. 2009. Calibrating a segmented taper equation with two diameter measurements. Southern Journal of Applied Forestry. 33(2): 58-61.

Clutter, J.L.; Fortson, J.C.; Pienaar, L.V. [and others]. 1983. Timber management: A quantitative approach. New York: John Wiley and Sons. 333 p.

Clark, A.; Souter, R.A.; Schlaegel, B.E. 1991. Stem profile equations for southern tree species. Res. Pap. SE-282. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 117 p.

Gan, J.; James, H.M.; Hsiaohsuan, W.; John, W.T. 2009. Invasion of tallow tree into southern US forests: influencing factors and implications for mitigation. Forest Resource. 39: 1346-1356.

Garber, S.M.; Maguire, D.A. 2003. Modeling stem taper of three central Oregon species using nonlinear mixed effects models and autoregressive error structures. Forest Ecology and Management. 179: 507-522.

Gordon, A.D.; Lundgren, C.; Hay, E. 1995. Development of a composite taper equation to predict over-and under-bark diameter and volume of *Eucalyptus salogna* in New Zealand. New Zealand Journal of Forestry Science. 25(3): 318-327.

Hilt, D.E. 1980. Taper-based systems for estimating stem volumes of upland oaks. Res. Pap. NE-458. Broomall, PA: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 15 p.

Husch, B.; Miller, C.I.; Beers, T.W. 1982. Forest mensuration. Malabar, FL: Krieger Publishing Company. 402 p.

Jiang, L.; Brooks, J.R.; Wang, J. 2005. Compatible taper and volume equations for yellow-poplar in West Virginia. Forest Ecology and Management. 213: 399-409.

Kozak, A.; Munro, D.O.; Smith, J.H.G. 1969. Taper functions and their application in forest inventory. Forest Chronicals. 45: 278-283.

Larson, P.R. 1965. Stem form of young Larix as influenced by wind and pruning. Forest Science. 11(4): 412-424.

Matney, T.G.; Parker, R.C. 1992. Profile equations for several hardwood species to variable top diameter limits. Southern Journal of Applied Forestry. 16(2): 75-78.

Max, T.A.; Burkhart, H.E. 1976. Segmented polynomial regression applied to taper equations. Forest Science. 22(3): 283-289.

McClure, J.P.; Czaplewski, R.L. 1986. Compatible taper equation for loblolly pine. Canadian Journal of Forest Research. 16: 1272-1277.

Methol, R.J. 2001. Comparisons of approaches to modeling tree taper, stand structure, and stand dynamics in forest plantations. Christchurch: New Zealand. 298 p. Ph.D. dissertation.

Muhairwe, C.K. 1999. Taper equations for Eucalyptus pilularis and Eucalyptus grandis for the north coast in New South Wales, Australia. Forest Ecology and Management. 113: 251-269.

Ormerod, D.W. 1973. A simple bole model. Forest Chronicals. 49: 136-138.

Ounekham, K. 2009. Developing volume and taper equations for Styrax tonkinensis in Laos. New Zealand: University of Canterbury. 83 p. M.S. thesis.

Parker, R.C. 1997. Nondestructive sampling applications of the Tele-Relaskop in forest inventory. Southern Journal of Applied Forestry. 21(2): 75-83.

Schlaegel, B. E., 1981. Testing, reporting, and using biomass estimation models. In: Proceedings of the southern forest biomass workshop, third meeting of the southern forest biomass working group. Hobcaw Barony, Georgetown, South Carolina, June 11-12, 1981. Clemson, SC: The Belle W. Barusch Forest: 95-112.

Sharma, M.; Burkhart, H.E. 2003. Selecting a level of conditioning for the segmented polynomial taper equation. Forest Science. 49(2): 324-330.

Souter, R.A. 2003. Taper and Volume Prediction in Southern Tree Species. Watkinsville: GA. ForesTech International, LLC: 1-19.

Tan, Y. 2012. Predicting the potential distribution of major invasive species using geospatial models in Southern forest lands. Starkville, MS: Mississippi State University. 76 p. M.S.thesis.

Zakrzewski, W.T.; MacFarlane, D.W. 2006. Regional stem profile model for crossborder comparisons of harvested red pine (*Pinus resinosa* Ait.) in Ontario and Michigan. Forest Science. 52: 468-475.

Zhang, K.; Lin, Y. 1994. Chinese Tallow. Beijing, China: China Forestry Press. 460 p.

WILD PIGS: INCITING FACTOR IN SOUTHERN PINE DECLINE?

Lori G. Eckhardt, Roger D. Menard, and Stephen S. Ditchkoff[1]

Abstract—During an investigation into southern pine decline at Fort Benning Georgia, the possibility of wild pigs (Sus scrofa) as an inciting factor became evident. Their rooting activity caused significant root damage on sites showing symptoms of pine decline. It was thought that perhaps the pigs may be moving around pathogenic fungi during their rooting activity in Pinus taeda (loblolly pine) stands. In 2008 and 2010, fungal isolates were obtained from the snouts of wild pigs captured from these stands, as well as, from root-feeding bark beetles and roots sampled 2003-2005. Micromorphology and DNA sequences of the ITS, elongation factor, and beta-tubulin gene regions were employed to identify the fungi recovered. Two new Ophiostoma species and a new Leptographium species were recovered. This study shows that wild pigs may exacerbate pine decline in this location by predisposing trees to bark beetles by reducing tree vigor when causing wounds for soil-borne or insect vectored pathogen infection and perhaps incidentally vectoring plant or tree pathogens during rooting activity.

INTRODUCTION

Several reports have indicated that localized forest health problems are increasing in southern pine forests (Eckhardt and others 2007, Hess and others 1999, 2005, Otrosina and others 1997. When these events are accompanied by sparse and chlorotic crowns, low annual stemwood growth, and isolation of fungal pathogens from the roots other than Phytophthora cinnamomi Rands or Heterobasidion annosum (Fr.) Bref., they are commonly referred to as pine decline (Eckhardt and others 2007, Menard 2007, Otrosina and others 1997). The etiology of forest declines typically involve complex of biotic and abiotic agents, which either exacerbate or mitigate the extent of tree growth reduction and tree mortality differentially. The decline model developed by Manion (1981) suggested that declines begin with predisposing factors related to host genotype, site or other typically abiotic factors which may be permanent or long term. Inciting factors are short term stressors such as insect defoliation, a frost event, drought or damage to the roots that may cause injury to the tree, which under ideal conditions would not lead to tree death. Predisposing factors reduce the defenses against contributing factors, which are typically biotic agents such as fungi, viruses, or insects that normally do not affect healthy trees, but may infect or infest trees that have been compromised by the predisposing and inciting factors (Manion, 1981).

It has been hypothesized that stress associated with abiotic and biotic factors attract root- and lower stem-feeding bark beetles (Coleoptera: Curculionidae) to southern pine stands (Eckhardt and others 2007). Infesting insects found associated with declining pines include Hylastes salebrosus Eichhoff, H. tenius Eichhoff, Pachylobius picivorus (Germar), and Hylobius pales (Herbst) (Eckhardt and others 2007, Menard 2007, Zanzot and others 2010). Hylastes species use the roots of both dead and living conifers for their maturation feeding and breeding activities, rarely causing damage (Milligan 1978). The insects associated with pine decline can also vector root-inhabiting ophiostomatoid fungal associates (Eckhardt and others 2007, Menard 2007, Zanzot 2009).

Wild pigs (Sus scrofa) may be an inciting factor in the pine decline complex, as they can be a particularly destructive exotic species in the places of introduction (Wickland 2014). Wild pigs negatively impact almost all aspects of ecosystem function and structure. Their feeding activities significantly disturb soil layers and natural decomposition cycles (Bratton 1975, Lacki and Lancia 1986). Wild pigs are commonly found in pine stands and their feeding activities damage roots (unearthing and removing bark) and could cause stress to trees that may predispose them to bark beetle attack. It is possible that there is a common attraction for both wild pigs and insects. Studies show that root-feeding insects are more abundant on symptomatic pine decline sites and these sites have a higher incidence of ophiostomatoid fungal infection (Eckhardt and others 2007, Menard 2007, Zanzot 2010). Studies suggest that

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trees in stress as a result of abiotic (i.e. drought) and biotic (i.e. fungi) factors are attractive to bark beetle activity (Kelsey and others 2014, Lahr and Sala 2014, Netherer and others 2015, Sopow and others 2015). The same pattern of attraction may play a part in the role of wild pig incidence on symptomatic pine decline sites as well. Pigs may have a palate for the taste of ophiostomatoid infected roots. The feeding activity of insects and wild pigs on ophiostomatoid infected roots presents interesting ecological comparisons. The host is affected similarly by the insect and wild pig as they both are using the same root substrate for feeding and it is possible that both are vectoring fungi in the process.

While both the incidence of pine decline symptoms and wild pig populations appear to be increasing, it is possible that the feeding activities of the wild pigs and the damage they cause, could compound the pine mortality that is occurring. This study was established to determine (1) if there is an association between wild pigs feeding and insect numbers and (2) whether wild pigs are acquiring and transporting the same fungi as the insects through their rooting behavior.

MATERIALS AND METHODS

Study Area and Wild Pig Trapping

Wild pig trapping took place 2004-2010 for a variety of studies (Hanson and others 2008, 2009, Holtfeter and others 2008, Jolley and others 2010, Sparklin and others 2009, Williams and others 2011) on approximately 737 km^2 of the Fort Benning military installation which is located in the Midwestern portion of Georgia's Muscogee and Chattahoochee counties that are mid-state on the eastern Alabama border. The predominant land base is Upper Coastal Plain with some Piedmont transition zone along the Fall Line. Fort Benning is planted in longleaf (Pinus palustris) and loblolly (Pinus taeda), separated by patches of bottomland hardwoods consisting of sweetgum (Liquidambar styraciflua), yellow poplar (Liriodendron tulipifera), hickory (Cary asp), and oak (Quercus sp.). Hanson and others (2008) reported pig densities between 1.2 and 6.1 pigs/km^2 in a similar area of Fort Benning in 2004.

Wild pigs were trapped in corral- and box-style traps that were prebaited with whole corn. Trapping occurred throughout 2008 and 2010, and fungal isolates were collected from wild pigs that were euthanized, and some that were captured and released according to protocols for other studies. A more detailed description of the trapping protocol is described in Williams and others (2011). All animal handling procedures were approved by the Auburn University Animal Care and Use Committee (PRN#2007-1196).

Isolation of Fungi from Wild Pig Snouts

One hundred wild pigs were trapped and the outer edge of the snout (where soil collects in the hairs) was swabbed with a sterile Q-tip moistened with sterile water in 2008 and 2010. Swabs were placed in sterile plastic bags (one swab per bag) and placed on ice for transport to the laboratory. Swabs were plated on malt extract agar (MEA, 20 g malt extract, 15 g agar/L) and MEA amended with cycloheximide and streptomycin (CSMA) (Hicks and others 1980). Ophiostomatoid fungi were identified by their characteristic perithecia or conidiophores and transferred to fresh CSMA or MEA. All isolates are maintained in the culture collection (CMW) of the Forestry and Agricultural Biotechnology Institute (FABI), University of Pretoria, South Africa.

Insect Trapping and Health Assessment

Within the wild pig trapping area were thirty-seven 1/6 acre plots (Forest Health Monitoring protocols, Dunn 1999) that were established in loblolly pine (2003-2005). Sites were predominately loblolly pine with a mix of competing species of pine, including shortleaf (Pinus eschinata Mill.) and longleaf as well as hardwood species such as dogwood (Cornus florida L.), red maple (Acer rubrum L.), water oak (Quercu nigra L.), southern red oak (Quercus falcate Michx.) and black jack oak (Quercus marilandica Muench.). Surface soil type varied from sandy through loam to clay. Topography was generally lower to moderate relief upland ridges with moderate drainages to flat alluvial plains. The location of the plots was determined using the Loblolly Pine Decline Risk Map (Eckhardt 2003) which identified a site as either symptomatic (decline [D]) or asymptomatic (healthy [H])). Each plot was categorized in one of four stand age classes; <10, 10 - 19, 20 - 40 and >40 years. Four replicates of each combination of stand class (D and H, n=2) and age class (n=4) were installed.

At each of these locations, a root health assessment was performed on three dominant or codominant pines nearest to the center plot location. Root sampling was done with the modified two-root excavation method (Otrosina and others, 1997) to examine for root damage (wild pig, fire, insect, mechanical) and fungal presence. Plot disturbance (wild pig, fire, thinning) was also recorded. Root-feeding insects were sampled on subplots using pitfall traps (3 subplots per plot, 192 total pitfall traps) from March to May for a 3 year period (loblolly 2003-2005). Insects were collected on biweekly basis for transport to the laboratory for identification and isolation of associated fungi.

RESULTS AND DISCUSSION

Insect Trapping

The total number of root-feeding insects (Hylastes spp.) and reproduction weevils (Hylobius pales and Pachylobius picivorus) captured in pitfall traps increased annually during the 3 years of trapping (1117 in 2003, 1253 in 2004 and 2423 in 2005). Mean insect

numbers were significantly greater on symptomatic pine decline plots than asymptomatic plots for study years 2003 ($F_{1,30}$=4.22, p=0.0495) and 2004 ($F_{1,30}$=4.33, p=0.0468).

Isolation of Fungi from Wild Pigs, Insects and Roots

Isolates of two unidentified Ophiostoma sp. (Sp. 1 53 percent and Sp. 2 48 percent) and one unidentified Leptographium sp. (59 percent), as well as, Leptographium terebrantis (68 percent) were recovered from the pig snouts. Isolations also yielded non-staining fungi such as Aspergillus spp., Aureobasidium spp., Gliocladium spp., Penicillium spp., and Trichoderma spp., along with several bacterial spp. Leptographium terebrantis, L. procerum and G. alacris were recovered from the surface of three species of root-feeding bark beetles [Dendroctonus terebrans Olivier (Coleoptera: Curculionidae), H. salebrosus, and H. tenuis], and two species of root-feeding weevils (H. pales and P. picivorus), as well as, Ophiostoma sp. 1 and the unidentified Leptographium sp. found on the wild pig snouts. Ophiostomatoid fungi were found in roots from 23 of the 36 plots which included Leptographium terebrantis, G. alacris and the unidentified Leptographium sp. The overall proportion of ophiostomatoid species isolated from all infected roots was greater from roots of trees on symptomatic plots (91 percent) than those from asymptomatic plots (0.08 percent).

These studies indicate that there are a number of fungi, including several species of ophiostomatoid fungi, associated with both snouts of wild pig in addition to the fungi carried on the body of root-feeding bark beetles. The attraction of pigs to pine decline sites may be associated to elevated insect incidence and roots infected by ophiostomatoid fungi that wild pigs actively seek out using their keen sense of smell. Damage to roots by wild pigs can be very extensive and cause root mortality putting host trees under additional stress. Symptomatic loblolly pine sites had 90 percent more wild pig damage compared to the asymptomatic loblolly pine sites. The uprooting activity of pigs while feeding exposes roots to more insect attack and fungal infection compounding host stress and increasing symptoms of pine decline. The implication of wild pigs as possible plant pathogenic fungal vectors may be weak, but pigs do cause significant physical root damage. That damage is a severe biotic stress that increases root-feeding bark beetle numbers ($F_{1,30}$=4.87, p=0.0283) and possibly the incidence of ophiostomatoid fungal infection facilitating the spread of pine decline.

CONCLUSIONS

This preliminary study illustrates two ways in which wild pigs are associated with the pine decline process, including (1) causing damage to roots which has the potential to attract insects as it stresses the tree and (2) transporting pathogenic fungi. More focused research needs to be completed to determine whether pigs vector fungi between trees.

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LITERATURE CITED

Bratton S. P. 1975. The effect of European wild boar (Sus serofa) on Gray Beech Forest in the Great Smoky Mountains. Ecology. 56: 1356-1366.

Dunn, P.H. 1999. Forest Health Monitoring Field Methods Guide. Washington, DC: U.S. Department of Agriculture Forest Service. 120 p.

Eckhardt, L.G.; Weber, A.M.; Menard, R.D. [and others]. 2007. Insect-fungal complex associated with loblolly pine decline in central Alabama. Forest Science. 53: 84-92.

Hanson, L.B.; Grand, J.B.; Mitchell, M.S. [and others]. 2008. Change-in-ratio density estimator for feral pigs is less biased than closed mark-recapture estimates. Wildlife Research. 35: 695-699.

Hanson, L.B.; Mitchell, M.S.; Grand, J.B. [and others]. 2009. Effect of experimental manipulation on survival and recruitment of feral pigs. Wildlife Research. 36: 185-191.

Hess, N.J.; Eckhardt, L.G.; Menard, R.D. [and others]. 2005. Assessment of loblolly pine decline on the Shoal Creek/ Talladega Ranger Districts, Talladega National Forest, Alabama and Choccolocca State Forest. FHP Rep. No. 2005-02-05. Montgomery, AL: U.S. Department of Agriculture Forest Service. 36 p.

Hess, N.J.; Otrosina, W.J.; Jones, J.P. [and others]. 1999. Reassessment of loblolly pine decline on the Oakmulgee District, Talladega National Forest, Alabama. Gen. Tech. Rep. SRS-50. Ashville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 560-564.

Hicks, R.R.; Howard, J.E.; Watterston, K.G.; Coster, J.E. 1980. Rating forest stand susceptibility to southern pine beetle in east Texas. Forest Ecology and Management. 2: 269-283.

Holtfreter, R.W.; Williams, B.L.; Ditchkoff, S.S.; Grand, J.B. 2008. Ferel pig detectability with game cameras. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies. 62: 17-21.

Jolley, D.B.; Ditchkoff, S.S.; Sparklin, B.D. [and others]. 2010. Estimate of herpetofauna depredation by a population of wild pigs. Journal of Mammalogy. 91: 519-524.

Kelsey, R.G.; Gallego, D.; Sanchez-Garcia, F.J.; Pajares, J.A. 2014. Ethanol accumulation during severe drought may signal tree vulnerability to detection and attack by bark beetles. Canadian Journal of Forest Research. 44: 554-561.

Lacki M. J.; Lancia R. A. 1986. Effects of wild pigs on beech growth in Great Smoky Mountains National Park. Journal of Wildlife Management. 50: 655-659.

Lahr, E.; Sala, A. 2014. Species, elevation, and diameter affect whitebark pine and lodgepole pine stored resources in the sapwood and phloem: implications for bark beetle outbreaks. Canadian Journal of Forest Research. 44: 1312-1319.

Manion, P.D., 1981. Tree disease concepts. Englewood Cliffs, NJ: Prentice-Hall. 399 p.

Menard, R.D. 2007. An assessment of the risk mapping system for the use of managing loblolly pine decline sites within red-cockaded woodpecker habitat. Baton Rouge, LA: Louisiana State University. 54 p. M.S. thesis.

Milligan, R.H. 1978. Black pine bark beetle. Forest and Timber Insects in New Zealand. 29: 1-6.

Netherer, S.; Matthews, B.; Katzensteiner, K. [and others]. 2015. Do water-limiting conditions predispose Norway spruce to bark beetle attack? New Phytologist. 205: 1128-1141.

Otrosina, W.J.; Hess, N.I.; Zarnoch, S.I. [and others]. 1997. Blue-stain fungi associated with roots of southern pine trees attacked by the southern pine beetle, Dendroctonus frontalis. Plant Disease. 81: 942-945.

Sparklin, B.D.; Mitchell, M.S.; Hanson, L.B. [and others]. 2009. Territoriality of feral pigs in a highly persecuted population on Fort Benning, Georgia. Journal of Wildlife Management. 73: 497-502.

Sopow, S.L.; Bader, M.K.-F.; Brockerhoff, E.G. 2015. Bark beetles attacking conifer seedlings: picking the weakest or feasting upon the fittest? Journal of Applied Ecology. 52: 220-227.

Wickland, K. 2014. Sus scrofa, wild boar. Animal Diversity Web. University of Michigan. Museum of Zoology. http://animaldiversity.org/accounts/Sus_scrofa/ [Date accessed: May 12, 2015].

Williams, B.L.; Holtfreter, R.W.; Ditchkoff, S.S.; Grand, J.B. 2011. Trap style influences wild pig behavior and trayping success. Journal of Wildlife Management. 75: 432-436.

Zanzot, J.W. 2009. Biology and ecology of root-feeding beetles and ophiostomatoid fungi in sandhills longleaf pine stands. Auburn, AL: Auburn University. 242 p. Ph.D. dissertation.

Zanzot, J.W.; Matusick, G.; Eckhardt, L.G. 2010. Ecology of root-feeding beetles and their associated fungi on longleaf pine in Georgia. Environmental Entomology. 39: 415-423.

Conservation

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BREEDING BIRD RESPONSE TO A SECOND-STAGE SHELTERWOOD HARVEST IN AN UPLAND HARDWOOD FOREST

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Abstract—Post-logging sites were historically assumed to provide unfavorable habitat for songbirds. Timber harvests have always been important for species that require disturbances, but while most studies focus on clearcuts, few examine the harvesting methods ranging between clearcutting and undisturbed forests; such as those created with shelterwood prescriptions. We studied breeding bird community response to different basal area retention levels during the second stage of a two-stage shelterwood harvest, 12 and 13 years post-initial harvest. Study sites were located in an upland hardwood forest in northern Alabama on the southern end of the mid-Cumberland Plateau through territory mapping. This study examined how different retention cuts affect bird species with varying habitat requirements. Bird diversity (calculated using the Brillouin Index) was compared among treatments using analysis of variance with a post-hoc LSD test. The relationship between habitat features and bird diversity were examined using correlation analysis. Breeding bird diversity was significantly lower in the control stands than the 50 percent retention stands (2013 p = 0.002 – 2014 p = 0.007) in both years. Breeding bird diversity was positively correlated with sapling density in 2013 (r = 0.65); showed a quadratic correlation with sapling density in 2014 (r = 0.85), and canopy cover in both years (2013 r = 0.96; – 2014 r = 0.96). Breeding bird diversity was similar across all four shelterwood treatments in both years, although these treatments had structural differences among them. These finding suggest that breeding bird diversity is dynamic and temporally dependent upon timber harvest and subsequent succession.

INTRODUCTION

Neotropical songbird conservation continues to be a topic receiving considerable focus and attention due to habitat degradation and/or transformation over the past century (Augenfeld and others 2008, Schlossberg and King 2007). Neotropical songbirds are primarily habitat specialists, each requiring a specific habitat during the breeding period (Perry and Thill 2013). MacArthur (1964) stated that the vegetation structure of an area can affect the abundance, richness and diversity of the breeding songbird community. Due to entire bird communities' wide array of habitat preferences, it is difficult for managers to create suitable habitat for all species. Regenerating clearcuts are considered detrimental to mature/interior forest birds, but beneficial for early-successional/disturbance dependent species immediately following harvest (Gram and others 2003). Silviculture practices differ in their effects on bird communities, primarily shelterwood prescriptions. Because shelterwood harvests are a two stage process, where the initial disturbance or harvest reduces overstory to create favorable light conditions for understory regeneration and reproduction, some favorable habitats are retained by the remaining canopy trees (Newell and Rodewald 2012). Over the course of 10 years, the understory is allowed to grow under the protection of the overstory; thus, creating a vertically stratified forest structure. The shift in plant communities can alter bird communities due to birds' preferences associated with vegetation complexity (Brawn and others 2001). The habitat heterogeneity created through varying levels of structural complexity provides viable habitat for a large suite of bird species that require differing habitat (Twedt and Somershoe 2009). With shelterwood prescriptions, we can manage for a myriad of songbird species that vary in their habitat preferences. We seek to explore the specific habitat associations driving bird species composition changes across various levels of shelterwood harvest.

METHODOLOGY

Study Site and Design

Study sites were located in northern Jackson County, AL on the southern end of the mid-Cumberland Plateau. Three block replications at 2 different sites (1 at Miller Mountain (MM) and 2 at Jack Gap (JG)) with 5 stands each comprise the study areas (totaling 3 block replications and 15 stands). Miller Mountain has a mean elevation of 500 m with a southwestern aspect, and Jack Gap has elevations of 450 and 360 m respectively with both having northern aspects. Study sites were mainly composed of oaks (*Quercus velutina, Q. rubra, Q. alba, Q, prinus*), yellow-poplar

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(*Liriodendron tulipifera*), hickories (*Carya* spp.), and sugar maple (*Acer saccharum*) (Schweitzer 2004). The study was established in 2001, with each stand within a block randomly receiving one of five possible basal area retention treatments (0, 25, 50, 75, and 100 percent retention). Trees in the 0, 25, and 50 percent retention stands were harvested by chain saw felling and grapple skidding. Stands receiving the 75 percent retention treatments had the midstory removed by means of the hack-and-squirt technique using an herbicide (Arsenal®, active ingredient imazapyr) in order to mimic overstory removal. Stands within the 100 percent retention category were undisturbed. All fifteen stands were allowed to grow for ten years, prior to final harvest in 2011. Canopies in the 25, 50, 75, and 100 percent retention treatments were removed while the 0 percent retention treatments were left undisturbed; three new control stands (one each per block replication) were put in place for bird community surveying purposes. This created five forest cohorts: (1) ten-year old regeneration clearcut ("old clearcut") (consisted of 0 percent retention stands), (2) released regeneration 1, with more vertical structure due to understory/midstory persistence through the second harvest and new sprouts from midstory and overstory trees (consisted of 25 and 50 percent retention stands), (3) released regeneration 2, with less vertical structure due to lack of midstory from herbicide during the initial harvest which subdued resprouting (consisted of 75 percent retention stands), (4) "new clearcut" or "old control", due to the lack of treatment during the initial phase, this was essentially a clearcut ten years after the first clearcut (consisted of 100 percent retention stands), and (5) mature or "new control", a forest that had not been disturbed for greater than 40 years (herein referred to as "2011 control stands") (Personal communication. Callie J. Schweitzer. 2014. Research Forester, Southern Research Station, 730-D Cook Avenue, Huntsville, AL 35801).

Bird Community Assessment

The spot-mapping technique was used to determine territory densities of breeding songbirds during the 2013 and 2014 season (May 1 – June 30). The spot-mapping method allows for robust estimators of species diversity, richness, and evenness because sites are visited 10 times throughout the survey period, which accounts for temporal variation within the breeding period. This technique is superior to single-visit point counts and line transects for observing behavioral patterns during the breeding period (Ralph and others 1993). Surveys were conducted beginning around 0500-0530 (CST) and lasted about five hours, with each stand taking 45-60 minutes. During surveys, territorial behaviors (songs, calls, and/or displays) were recorded on a topographic map with each stand visit being recorded on its own separate map. After the breeding season finished, maps were digitally transferred into ArcMAP v. 10.1. For a species to register a territory, it must have met the following requirements: (1) individuals must be detected on three or more visits, with two detections being at least ten days apart, (2) territories in one stand must be evaluated simultaneously with territories of an adjacent stand, and (3) if there are multiple territories of the same species within a single stand, one of the following requirements must be met: a minimum of one simultaneous interaction with a conspecific or at least two pairs of non-simultaneous registrations (Lesak 2004).

Vegetation Assessment

Habitat was assessed using a modified James and Shugart method (1970) 0.4 ha circular plot method. Each stand had three vegetation plots that were randomly generated in ArcMap v. 10.1. Percent canopy cover was recorded using a hand-held spherical densitometer at the center of each plot and at 11.3 m from the plot center in the 4 cardinal directions. Understory sapling density was measured by walking along transects that were 11.3 m from the plot center in the 4 cardinal directions and counting all woody stems less than 30 mm in diameter at 0.25 m above the ground and were within 0.85 m on each side of the transect. Percent canopy cover and understory density were grouped and averaged for each stand.

Statistical Analysis

Spot-mapping assumes a census of the entire songbird community for a study site (Pielou 1975). As such, the Brillouin diversity index was used to describe the songbird community for each treatment based on territories of each species (Zar 1996). A one-way analysis of variance (ANOVA) of the Brillouin diversity index was used with a post-hoc least squared difference test (LSD) to test for differences between treatments. A correlation analysis was used to determine the relationship between songbird diversity and habitat features (canopy cover and understory density), with an R value being reported for goodness of fit. Each season (2013 and 2014) was analyzed individually. All statistical analyses were computed in SPSS v. 20. All statistical tests were deemed significant at $p \leq 0.05$, unless otherwise stated.

RESULTS

Vegetation varied depending on whether a treatment had canopy removal during the second phase of this study. Treatments subjected to canopy removal during the second phase had greater understory vegetation density and less canopy closure than treatments that were left undisturbed during the second phase harvest (table 1). Understory density in the 25, 50, 75, and 100 percent retention stands were significantly greater than in the 2011 control stands in 2013 ($F = 6.40$, $p = 0.000$) and 2014 ($F = 6.12$, $p = 0.005$). Canopy cover was

Table 1—Percent canopy cover and understory sapling density for 2013 and 2014 seasons. Letters denote significant differences at p ≤ 0.050. Table depicts mean ± standard error

Treatment	Brillouin Diversity-2013	Canopy Cover-2013	Sapling Density-2013	Brillouin Diversity-2014	Canopy Cover-2014	Sapling Density-2014
0 % Retention	0.96 ± 0.07^{bc}	94.08 ± 2.99^{b}	21.69 ± 5.82^{abc}	0.89 ± 0.12^{bc}	93.41 ± 2.06^{a}	27.61 ± 8.47^{ab}
25 % Retention	1.07 ± 0.06^{ab}	73.21 ± 5.09^{ab}	36.19 ± 3.73^{c}	1.06 ± 0.05^{ab}	78.24 ± 10.76^{a}	44.89 ± 8.82^{bc}
50 % Retention	1.13 ± 0.01^{a}	68.63 ± 6.02^{ab}	17.17 ± 3.99^{ab}	1.13 ± 0.03^{a}	80.22 ± 5.06^{a}	43.36 ± 6.68^{b}
75 % Retention	1.07 ± 0.05^{ab}	41.19 ± 7.87^{a}	34.97 ± 7.51^{bc}	1.05 ± 0.03^{ab}	60.42 ± 11.68^{a}	68.64 ± 13.44^{c}
100 % Retention	1.13 ± 0.02^{a}	54.28 ± 7.74^{ab}	35.06 ± 7.55^{bc}	1.11 ± 0.01^{a}	65.98 ± 13.04^{a}	48.53 ± 4.36^{bc}
2011 Control	0.89 ± 0.02^{c}	92.62 ± 2.89^{ab}	8.86 ± 1.22^{a}	0.87 ± 0.01^{c}	92.51 ± 1.74^{a}	9.39 ± 1.25^{a}

significantly different between the 0 percent retention stands and the 75 percent retention stands in 2013 ($F = 7.98$, $p = 0.000$); 0 percent retention and 2011 control stands were significantly different from the 75 and 100 percent retention stands in 2014 ($F = 2.39$, $p = 0.101$).

Brillouin diversity was significantly different across treatments in both 2013 ($F = 5.23$, $p = 0.009$) (fig. 1) and 2014 ($F = 3.84$, $p = 0.026$) (fig. 2) seasons, with treatments subjected to the second stage canopy release having higher diversity values than treatments not receiving any second-stage harvest for both 2013 and 2014. In 2013, diversity showed a quadratic relationship with canopy cover ($r = 0.96$) (fig. 3) and a positive linear relationship with understory sapling density ($r = 0.65$) (fig. 4). In 2014, diversity showed a quadratic relationship with canopy cover ($r = 0.96$) (fig. 5) and understory density ($r = 0.85$) (fig. 6).

Songbird species were mainly composed of early-successional and edge habitat breeders (table 2); with the Indigo Bunting (*Passerina cyanea*) (2013-34 territories, 2014-26 territories), Prairie Warbler (*Setophaga discolor*) (2013-26 territories, 2014-29 territories), and Yellow-breasted Chat (*Icteria virens*) (2013-30 territories, 2014-30 territories) having the greatest territory abundances for both years, though abundances varied from year-to-year. Birds associated with mature/interior forest habitat had lower abundances (White-breasted Nuthatch: 2 and 6 territories in 2013 and 2014, Worm-eating Warbler: 10 and 12 territories in 2013 and 2014, Wood Thrush: 4 and 4 territories in 2013 and 2014) and fewer mature forest species were present (8 mature forest species were detected in both 2013 and 2014) than early-successional species (10 species and 11 species in 2013 and 2014 respectively) and edge habitat species (12 species in both 2013 and 2014).

DISCUSSION

Treatments that showed minimal vertical complexity (0 percent retention and 2011 control) had lower breeding bird diversity. Treatments that were subjected to canopy removal during the second stage of the shelterwood harvest (25, 50, 75, and 100 percent retention) showed a greater amount of vertical structure, and had greater songbird diversity within their treatments. Because of this study's implementation, we were able to manage for an entire songbird community that has varying habitat requirements. The staggered levels of canopy removal at the beginning of the study allowed us to mimic different stages of forest succession, thus meet the breeding needs of species from early-successional to mature forests immediately following implementation and 10+ years after. Yahner (2003) stated that managing for early-successional habitats in forested landscapes can benefit the long term conservation of both early-successional and mature forest bird species. While clearcutting may not provide immediate benefit for some songbird species, it is an integral part of the system that should remain in management plans to increase habitat heterogeneity and improve habitat suitability for Neotropical migrants. This study shows that with time habitats created by differing regeneration methods change in their desirability to an array of wildlife species that shift in time from birds that prefer open environments to those who prefer mature forests. In other words, regenerated stands progress through a sequence of preferred habitat depending on the bird species requirements for nesting or survival and growth in certain life stages. Therefore, we must be careful not to associate a specific treatment type with certain songbird species or community indices (diversity, richness, and evenness), and with the stage of stand development following regeneration.

Rather, we can look at direct effects (vegetation features) which might explain what drives the bird community. Songbird diversity showed a quadratic relationship with percent canopy cover, with an optimal canopy range between 40-80 percent canopy cover. Beyond 80 percent canopy closure we began to see a sharp decrease in songbird diversity, most likely due to a lack of sunlight penetrating through the canopy into

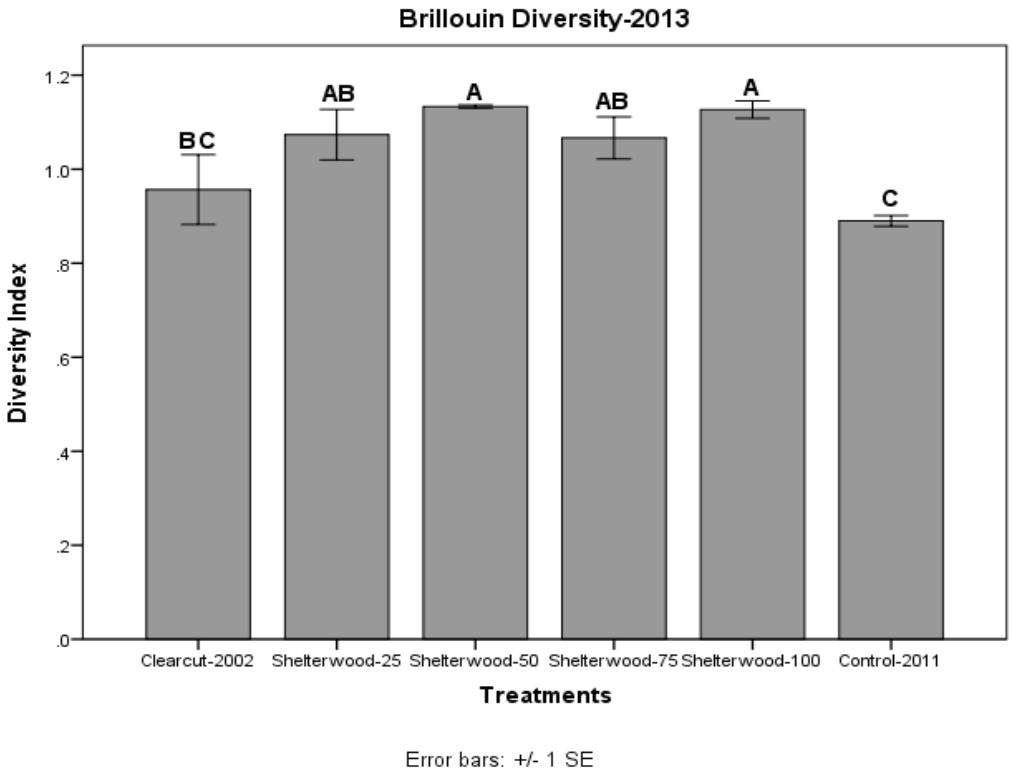


Figure 1—Brillouin diversity of treatments for the 2013 season. Letters denote significant differences at the 0.05 level. Graph depicts mean and standard error.

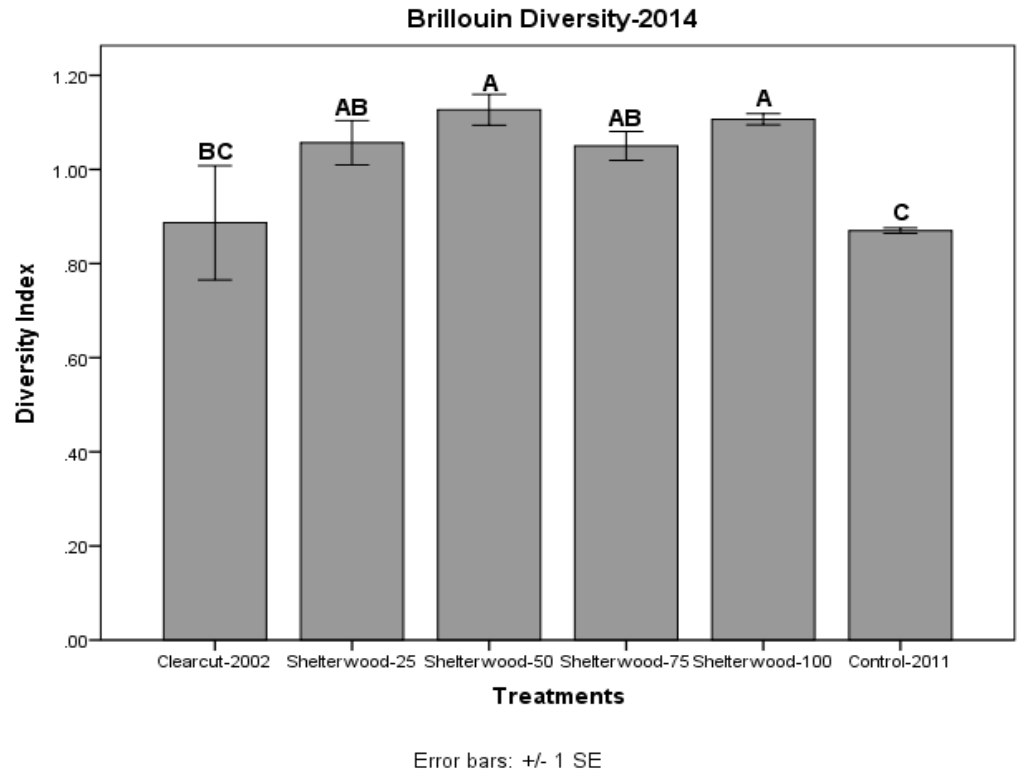


Figure 2—Brillouin diversity of treatments for the 2014 season. Letters denote significant differences at the 0.05 level. Graph depicts mean and standard error.

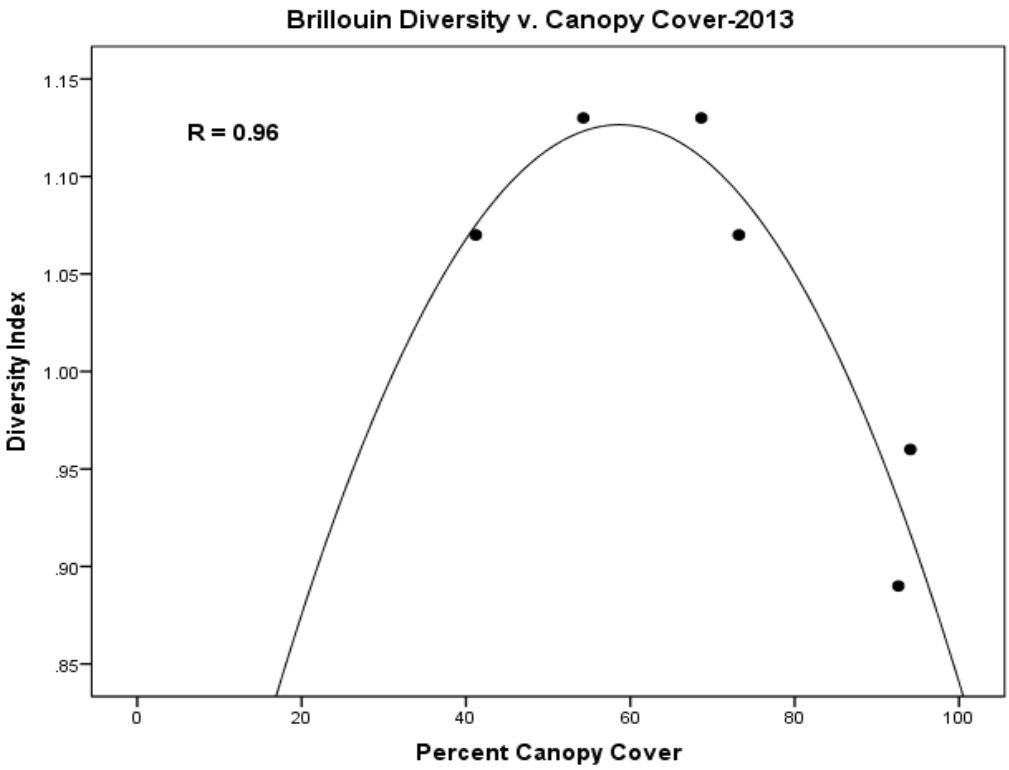


Figure 3—Brillouin diversity in relation to canopy cover for the 2013 season.

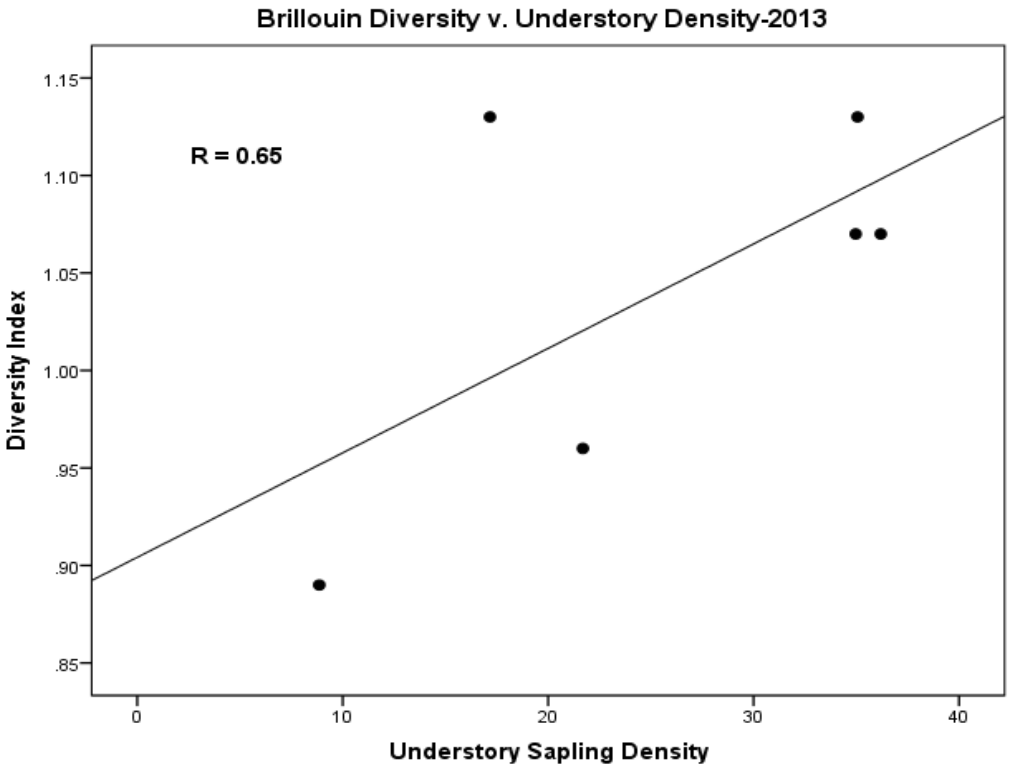


Figure 4—Brillouin diversity in relation to understory sapling density for the 2013 season.

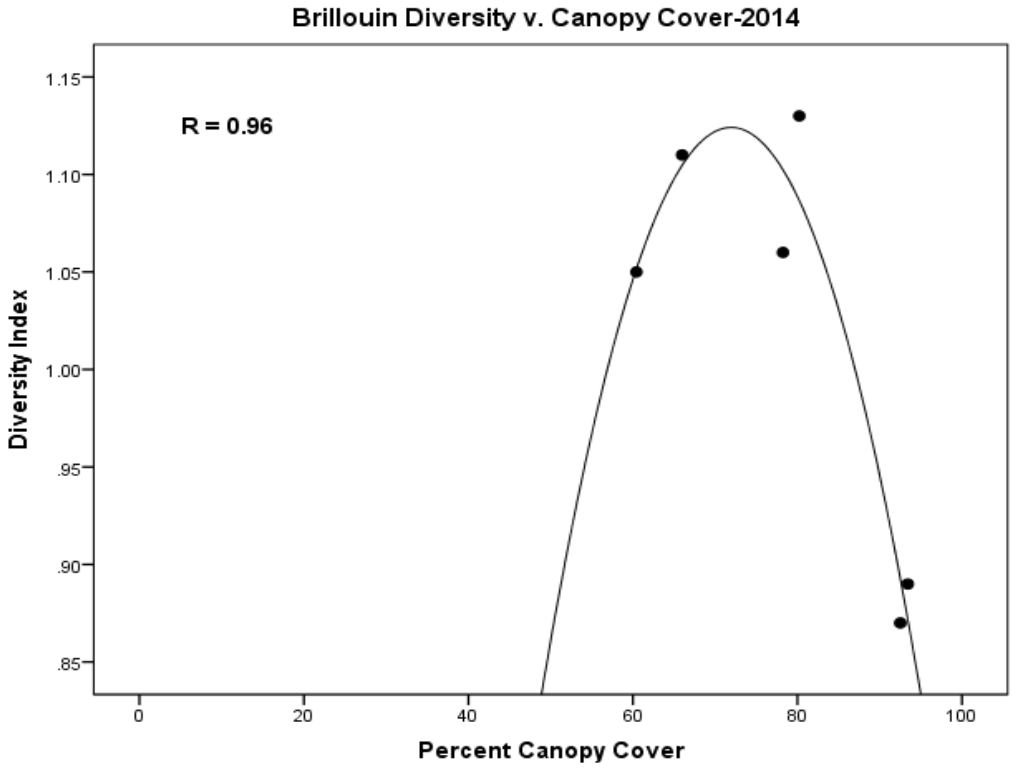


Figure 5—Brillouin diversity in relation to canopy cover for the 2014 season.

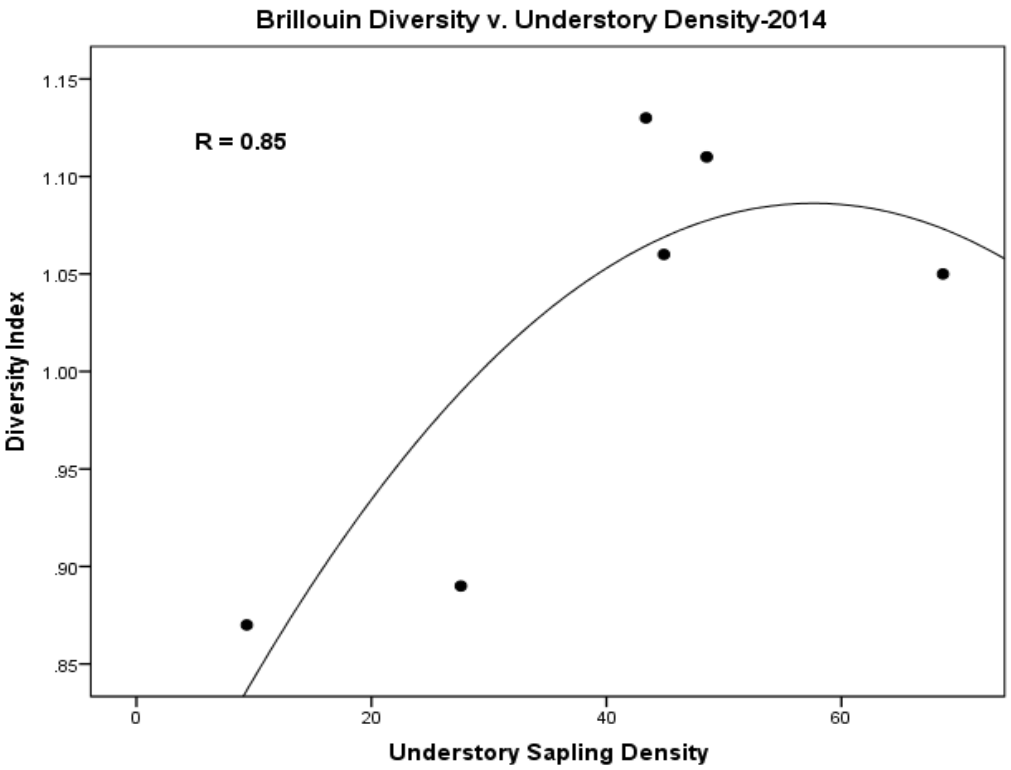


Figure 6—Brillouin diversity in relation to understory sapling density for the 2014 season.

Table 2—Territory abundances of songbird species for the 2013 and 2014 seasons. Habitat categories for individual species are as follows: O/E-open/edge habitat, I/E-interior/edge habitat, I-interior forest

2013			2014		
Species	Habitat	Territories	Species	Habitat	Territories
Indigo Bunting	O/E	34	Yellow-breasted Chat	O/E	30
Yellow-breasted Chat	O/E	30	Prairie Warbler	O/E	29
Prairie Warbler	O/E	26	Indigo Bunting	O/E	26
Northern Cardinal	I/E	25	White-eyed Vireo	O/E	23
Eastern Towhee	I/E	23	Eastern Towhee	I/E	19
Kentucky Warbler	I/E	21	Red-eyed Vireo	I/E	19
Red-eyed Vireo	I/E	18	Hooded Warbler	I	18
Tufted Titmouse	I/E	17	Tufted Titmouse	I/E	17
Carolina Wren	O/E	16	Northern Cardinal	I/E	17
White-eyed Vireo	O/E	16	Kentucky Warbler	I/E	16
Yellow-throated Vireo	I/E	16	Carolina Wren	O/E	14
Hooded Warbler	I	14	Worm-eating Warbler	I	12
Scarlet Tanager	I	13	Scarlet Tanager	I	12
Red-bellied Woodpecker	I/E	13	Summer Tanager	I/E	11
Summer Tanager	I/E	12	American Goldfinch	O/E	10
Eastern Wood-pewee	I/E	10	Blue-gray Gnatcatcher	I/E	10
Worm-eating Warbler	I	10	Red-bellied Woodpecker	I/E	9
Carolina Chickadee	I/E	9	Eastern Wood-pewee	I/E	7
Black-and-white Warbler	I	8	Carolina Chickadee	I/E	7
Hairy Woodpecker	I	7	White-breasted Nuthatch	I	6
American Goldfinch	O/E	7	Black-and-white Warbler	I	6
Blue-gray Gnatcatcher	I/E	6	Wood Thrush	I	4
Downy Woodpecker	I/E	5	Pileated Woodpecker	I	4
Pileated Woodpecker	I	4	Downy Woodpecker	I/E	4
Wood Thrush	I	4	Hairy Woodpecker	I	4
Field Sparrow	O/E	3	Yellow-throated Vireo	I/E	3
Orchard Oriole	O/E	3	Blue Jay	I/E	3
Chipping Sparrow	O/E	3	Orchard Oriole	O/E	3
White-breasted Nuthatch	I	2	Common Yellowthroat	O/E	2
Eastern Bluebird	O/E	1	Field Sparrow	O/E	1
American Redstart	I	1			
Blue Jay	I/E	1			

the understory and subsequent decline in vegetation structure in the ground and lower canopy positions. These canopy values themselves might be somewhat misleading because values were obtained at 1.5 m above ground level and reflect any canopy above that height. Productivity at these study sites is high, and we see noticeable changes in the habitat from year-to-year. From 2013 to 2014, the amount of canopy closure increased 5-20 percent in the shelterwood treatments, but these treatments had only been subjected to this increase overstory for a short period of time. The understory had not yet been affected by the decreased amount of light for us to see any concurrent detrimental effects on the bird community. Songbird diversity showed a positive relationship with the amount of understory sapling density. As the understory becomes denser, the amount of foraging and nesting opportunities increases for a large portion of these species. Foraging is increased with the amount of resources immediately available, which becomes important during the breeding and post-breeding season (Chandler and others 2012). As many of these species are Neotropical migrants and must prepare for trans-continental migration, areas where food is immediately available and in high amounts is important for successful travel (Marshall and others 2003). Areas with dense understory also provide nesting opportunities for a variety of species, which can use different types of vegetation to their benefit (Vitz and Rodewald 2011). Predator avoidance is also an important component with which dense vegetation can offer cover or protection. During the post-breeding period, when juveniles are beginning to dispersal but do not have the mobility that adults do can be a period where foliage from vegetation can increase survivability (King and others 2006).

Canopy cover seemed to be a factor that indirectly affects diversity of the songbird community through influence on understory sapling density, which in turn directly affects bird community diversity. With increasing light penetration to the understory, early-successional plant species are stimulated. This creates a dense level of understory vegetation, which can benefit the foraging and breeding for songbirds and influence community diversity. Understory sapling density showed a strong positive relationship with community diversity. Due to the creation of habitat that met the needs of many bird species and for different periods during the summer (breeding, post-breeding, pre-migration preparation), we believe dense understory habitat with herbaceous vegetation to be a good habitat driver of the bird community. Measures of the amount of herbaceous and forb material associated with each treatment may have improved our ability to detect changes in bird diversity among treatments. Sapling density only tells a portion of the story, as we observed an increase in herbaceous growth following the final harvests. To truly understand the relationships that birds have with their habitat, we would have to account for non-woody species as well. For example, *Rubus* spp. is a key nesting substrate for Yellow-breasted Chats and offers substantial foraging opportunities for many bird species (Ricketts and Ritchison 2000), but is difficult to quantify in early-successional systems due in part to dense thickets associated with early-successional habitat. We noted thickets of *Rubus* spp. and other dense herbaceous areas that colonized the stands following the second harvest, particularly in the 75 percent retention stands. These areas were also where we found the greatest activity of Yellow-breasted Chats and other early-successional bird species. This research, which documented habitat and concurrent bird community usage, showed that these disturbance-dependent birds responded to disturbance caused by tree harvesting. The unique vegetation structure created using a two-stage shelterwood prescription allowed for higher bird diversity compared to less-frequently disturbed sites. Also of interest is the loss of suitable habitat due to vegetation regrowth. The 11-year old clearcut has now reached the stem exclusion stage, and no longer provides suitable habitat for early-successional bird species, but is now beginning to be used by mature forest bird species. The creation of habitat has also been seen to be beneficial for not only early-successional species immediately following harvest, but beneficial to mature forest species years later.

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LITERATURE CITED

Augenfeld, K. H.; Franklin, S. B.; Snyder, D. H. 2008. Breeding bird communities of upland hardwood forest 12 years after shelterwood logging. Forest Ecology and Management. 255: 1271-1282.

Brawn, J. D.; Robinson, S. K.; Thompson III, F. R. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics. 32: 251-276.

Chandler, C. C.; King, D. I.; Chandler, R. B. 2012. Do mature forest birds prefer early-successional habitat during the post-fledging period? Forest Ecology and Management. 264: 1-9.

Gram, W. K.; Porneluzi, P. A.; Clawson, R. L. [and others]. 2003. Effects of experimental forest management on density and nesting success of bird species in Missouri Ozark forests. Conservation Biology. 17: 1324-1337.

James, F. C.; Shugart, H. H. 1970. A quantitative method for habitat description. Audubon Field Notes. 24: 727-736.

King, D. I.; Degraaf, R. M.; Smith, M. L.; Buonaccorsi, J. P. 2006. Habitat selection and habitat-specific survival of fledgling ovenbirds (*Seirus aurocapilla*). Journal of Zoology (London). 269: 414-421.

Lesak, A. A. 2004. Response of Forest Songbird Communities to a Gradient of Overstory Retention in Northeastern Alabama. Normal, AL: Alabama A&M University. 72 p. M.S. thesis.

MacArthur, R. H. 1964. Environmental factors affecting bird species diversity. American Naturalist. 98: 387-397.

Marshall, M. R.; DeCecco, J. A.; Williams, A. B. [and others]. 2003. Use of regenerating clearcuts by late-successional bird species and their young during the post-fledging period. Forest Ecology and Management. 183: 127-135.

Newell, F. L.; Rodewald, A. D. 2012. Management for oak regeneration: short-term effects on the bird community and suitability of shelterwood harvests for canopy songbirds. The Journal of Wildlife Management. 76: 683-693.

Perry, R. W.; Thill, R. E. 2013. Long-term responses of disturbance-associated birds after different timber harvests. Forest Ecology and Management. 307: 274-283.

Pielou, E. C. 1975. Ecological Diversity. New York: Wiley.

Ralph, C. J.; Geupel, G. R.; Pyle, P.; [and others]. 1993. Handbook of field methods for monitoring landbirds. Gen. Tech. Rep. PSW-GTR144-www. Albany, CA: U.S. Department of Agriculture Forest Service, Pacific Southwest Research Station. 41 p.

Ricketts, M. S.; Ritchison, G. 2000. Nesting success of Yellow-breasted Chats: effects of nest site and territory vegetation structure. The Wilson Bulletin. 112: 510-516.

Schweitzer, C. J. 2004. First-year response of an upland hardwood forest to five levels of overstory tree retention. In: Conner, K. F., ed. Proceedings of the 12th biennial southern silviculture research conference. Gen. Tech, Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 287-291.

Schlossberg, S.; King, D. I. 2007. Postlogging succession and habitat usage of shrubland birds. Journal of Wildlife Management. 73: 226-231.

Twedt, D. J; Somershoe, S. G. 2009. Bird response to prescribed silvicultural treatments in bottomland hardwood forests. Journal of Wildlife Management. 73: 1140-1150.

Vitz, A. C.; Rodewald, A. D. 2011. Movement of fledging ovenbirds (*Seirus aurocapilla*) and worm-eating warblers (*Helmitheros vermivorum*) within and beyond the natal home range. The Auk. 127: 364-371.

Yahner, R. H. 2003. Response of bird communities to early successional habitats in a managed landscape. Wilson Bulletin. 115: 292-298.

Zar, J. H. 1996. Biostatistical Analysis. New Jersey: Upper Saddle River.

POTENTIAL EFFECTS OF FORESTRY OPERATIONS AND ASSOCIATED BEST MANAGEMENT PRACTICES ON RIPARIAN WILDLIFE SPECIES IN THE SOUTHEASTERN UNITED STATES

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Abstract—The US Fish and Wildlife Service is considering the addition of 374 riparian and aquatic species in the southeastern United States to the federal Threated and Endangered Species List. This recommendation is a result of a 2011 petition, which recognized forest operations as having negative effects on 51 percent of the listed species, citing research conducted in the absence of Best Management Practices (BMPs) (Federal Register 76(187):59836-59862). We conducted a literature review to evaluate how BMPs might benefit these species, but found that information specific to these riparian species and forest operations was generally limited. Available literature pertaining to BMP effects and these riparian species generally contained broader conclusions, which were often conducted at higher taxonomic levels. We were able to develop some broad interpretations that support the benefits of BMP implementation to many of these species. Our review indicated that BMPs (i.e., streamside management zones) can limit sediment and nutrient inputs, reduce thermal pollution, enhance water quality, and safeguard riparian ecosystems to a degree that should provide some level of protection for most of the investigated species. Stream crossing BMPs and stream crossing designs should be beneficial by restricting sediment input and by minimizing potentially negative changes to stream channel hydrology. Our findings generally support the need for additional research regarding the specific effects of BMPs on stream and riparian biota.

INTRODUCTION

Following the Clean Water Act of 1972, Forestry Best Management Practices (BMPs) have been developed to address potential water quality issues during forestry operations (Aust and Blinn 2004, Ice 2004, Shepard 2006). Potential impacts of sedimentation, temperature change, and chemical regimes have been addressed through the implementation of forestry BMPs (Anderson and Lockaby 2011, Aust and Blinn 2004). Streamside Management Zones (SMZs), sediment control mechanisms, and stream crossing designs can reduce, prevent, or eliminate negative ecological alterations that would otherwise be associated with harvesting operations (Aust and others 2011, Lakel and others 2006). A significant amount of research has shown that BMP implementation is associated with good water quality (Shepard 2006). When compared to natural stands, managed stands can support similar species diversity (Wigley 1997).

In 2011, the US Fish and Wildlife Service began the review process on a petition that requested 374 riparian and stream species from the southeastern United States be added to the federal Threatened and Endangered species list. This petition identified logging as a threat for 51 percent of the listed species and supported this statement predominantly with research conducted in the absence of appropriate forestry BMPs. The overall objective of our project was to conduct a literature review to evaluate how BMP implementation could potentially enhance water quality, preserve natural riparian habitat, and safeguard these petitioned species during harvesting operations. For this paper we chose to concentrate on the animal species.

METHODS

Our literature search included peer reviewed articles, government publications, theses, dissertations, and books. Specific information on habitat needs, life history, and home ranges of these species were obtained and placed into an appendix. This information was predominantly acquired through peer reviewed journal articles, textbooks, and government publications. Because significant research gaps existed

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between forestry operations and potential species level impacts, we frequently reviewed research findings for larger taxonomic classifications and for studies that were conducted at sites similar to conditions in the southeastern United States. With over 200 references, we compiled a significant amount of information regarding BMP implementation and resulting impacts on wildlife populations within the southeastern United States. The organization of our results does not necessarily follow a strict taxonomic classification grouping, but instead are organized based on the organization in the Federal Register listing (Federal Register 76(187):59836-59862).

RESULTS

Mammals

Insular cotton rat (*Sigmodon hispidus insulicola*), Sherman's short tailed shrew (*Blarina carolinensis shermani*), and the Pine Island Rice Rat or Marsh Rice Rat (*Oryzomys palustris*) (2 populations) were mammals listed in this petition. These rodents are successional species that thrive in harvested environments. The cotton rat prefers dense vegetation found in 3 to 10 year old clearcuts (Mengak and Laerm 2007). Small mammal abundance is generally, positively affected by 10 m riparian buffers (Constantine and others 2004). SMZs in the southeastern United States have been shown to be sufficient to support small mammal populations (Miller and others 2004). Forest edges, skid roads, and clearcut openings support higher densities, higher species richness, and greater diversity in small mammal populations (Moseley and others 2009).

Birds

Black rails (*Laterallus jamaicensis*), sandhill cranes (*Grus canadensis pratensis*), and seaside sparrows (*Ammodrammus maritimus macgillivraii*) are restricted to marshland habitats, and are not associated with forested habitat. In fact, woodland expansion and encroachment negatively impacts sandhill crane and seaside sparrow habitat (Elderd and Nott 2008, Johnson 2000). In order to preserve sandhill crane habitat at Rowe Sanctuary in Nebraska, woody vegetation that encroaches into water channels is managed through the use of heavy machinery (Kinzel and others 2009). Timber harvesting does not typically occur in marshlands, and therefore would not have an effect on these avian species.

Reptiles

To protect reptile habitat and water quality, a 30 to 60 m wide SMZ has been recommended (Rudolph and Dickson 1990, Semlitsch and Bodie 2003). Vegetative structure has been shown to influence reptile populations. Narrow SMZs can produce dense understory, which negatively influences reptile abundance (Rudolph and Dickson 1990). Maintaining riparian buffers increases turtle diversity (Sterrett and others 2010). In the southeastern United States, timber harvesting can create heterogenic environments, providing reptiles with egg laying and basking habitat in an otherwise forested environment (Moseley and others 2009, Russell and others 2002).

Amphibians

The southeastern United States is the "hotspot" of the nation's amphibian biodiversity (Weir and Greis, 2002). This diverse population is primarily threatened by habitat loss from conversion of forested land to urban, industrial, and agricultural uses (Weir and Greis 2002). The width of an SMZ influences amphibian abundance. Wider SMZs (30 to 95 m) support the greatest abundance, while narrower zones (0 to 20 m) commonly have the lowest abundance (Rudolph and Dickson 1990). The recommended width of SMZ to maintain amphibian abundance is at least 30 m (Rudolph and Dickson, 1990). Potential effects of harvesting operations on amphibians are complex, and could vary at a species level as well as by region (Alix and others 2014, deMaynadier and Hunger 1995). Foley (1994) found in his study in Texas that there were no differences in numbers of amphibians inside and outside a 65 foot SMZ among control, select cut, and clearcut treatments. Forest road and skid trails can potentially create suitable habitat for amphibians by creating artificial aquatic habitats and vernal pools for reproduction (Adams and Hook 1993, deMaynadier and Hunter 2000, Russell and others 2002).

Fish

General potential threats to the 48 species of fish listed in this petition include impoundment, channelization, altered hydrology, sedimentation, thermal pollution, dissolved oxygen, and changes in watershed land use. Riparian buffers as well as stream crossing selection and design specifically address the aforementioned potential threats to fish populations. Riparian buffers and SMZs safeguard fish from potential sedimentation, temperature level alterations, and detritus input by providing fish habitat, promoting species diversity, and maintaining water quality (Anbumozhi and others 2005). Moring (1982) found a 30 m wide unharvested riparian buffer was sufficient to protect salmon eggs from sedimentation. Clearing of riparian strips in excess of 1km. can substantially affect fish assemblage (Jones and others 1999, Quist and Schultz 2014). Stream crossing BMPs address fish mobility and potential threats of impoundment. While bridges commonly allow for adequate fish mobility, culverts may require certain design specifications to properly enable fish passage (Aust and others 2011, Kidd and others 2014). A significant number of studies have been conducted to investigate fish mobility in the presence of culverts (David and others 2014, Foster and Keller 2011, Jensen

2014). Culvert designs should focus on hydraulic velocity, entrance attractiveness, and species specific fish swimming abilities (Hotchkiss and Frei 2007, Jensen 2014). Culvert length, velocity, and depth are important factors to consider when designing a culvert for fish populations. Studies have shown that retrofitting could enhance fish passage in existing culverts (David and others 2014). Spoiler baffles and even mussel spats can enable fish passage (David and others 2014, Feurich and others 2012).

Amphipods and Isopods

As cave dwelling species, the amphipods and isopods listed in this petition can be impacted by energy base alterations, physical and hydrological disturbances, and temperature alterations (Holsinger 1972). In many states, forestry BMPs are in place to protect karst environments and cave entrances. Fifty foot riparian buffers are recommended for sensitive areas such as cave entrances, sinkholes, and areas above the cave passage (Personal communication. Daniel Feller. 2013. Western Region Ecologist, MD Department of Natural Resources, Appalachian Lab, 301 Braddock Rd., Frostburg, MD 21532). Consultation by a professional is an important component of preharvest planning in these areas (Zokaites 1997). Erosion control implementations and proper waste wood placement ensure that environmental conditions and energy base levels are not influenced by sedimentation or debris (Zokaites 1997).

Mussels

Dams are a major factor in freshwater mussel decline in the southeastern United States. Before impoundment of the Tennessee River, there were 100 species of mussels. Following impoundment by hydrological dams, those numbers were reduced to less than half, with only 44 species of mussels remaining (Watters 2009). Impoundments, sedimentation, channelization, dredging, loss of riparian buffers, and invasive exotics threaten mussel populations (Clayton and others 2001, Poole and Downing 2004, Thorp and Rogers 2011, Watters and others 2009). Stream crossing design and riparian buffers are BMPs that address potential sedimentation and hydrological issues. SMZs and riparian buffers can preserve mussel habitat, reduce potential sedimentation, preserve temperature levels, and enhance in channel diversity. Riparian buffers have been shown to maintain species richness and abundance. Poole and Downing (2004) found that having ≥ 50 percent forested riparian buffers had sustained mussel diversity, whereas ≤ 50 percent riparian buffers lost mussel diversity. Stream reaches having 80 percent riparian buffers lost almost no species (Poole and Downing 2004). Where possible, to promote motility, bridges are the preferred stream crossing. Culverts can be designed in a way that prevents potential impoundment, with bottomless culverts being the preferred design. By focusing on hydraulic velocity, turbulence scour and the threat of impoundment can be eliminated. Mussel habitat can be preserved by ensuring heterogeneous substrate and allowing for fish host passage.

Crayfish

Threats to the 83 listed crayfish include sedimentation, limited ranges, invasive species, and a loss of riparian buffers (Herrig and Shute 2002, Lodge and others 2000, Parkyn and Collier 2004). Adequate riparian buffers and proper stream crossings can reduce or eliminate these potential threats during harvesting operations and protect essential habitat (Graynoth 1979, Parkyn and Collier 2004). If not properly designed, culverts could potentially restrict crayfish motility, limit dispersal, increase predation, alter riverbed channels, and create substrate homogeneity. Elevated water velocity beyond a species motility threshold can cause an inability for crayfish to maintain their position within a culvert (known as “slippage”). Slippage can occur at velocities as low as 2 cm/s downstream, and upstream at velocities of between 30 and 40 cm/s, depending on the species (Foster and Keller 2011, Louca and others 2014). Higher culvert velocities can serve as a selective environmental filter, selecting for the more tolerant of velocity, and far more aggressive nonnative rusty crayfish (Foster and Keller 2011). In a mixed agricultural and urban environment, there was an increase in brown rat predation on crayfish within and around culverts whose increased water velocity inhibited crayfish movement (Louca and others 2014). Corrugations, oxidized culvert bottoms, or natural substrate on the bottom of culverts can assist with reducing this slippage and enabling crayfish passage (Foster and Keller 2011). Louca and others (2014) recommended that water velocity be less than 30 cm/s to mitigate flow effects on crayfish passage. While further research is needed on individual species and their motility thresholds, proper culvert design ensures crayfish population motility during and following harvesting operations.

Snails

Snails have very limited ranges, and potential threats to populations include impoundments, sediment, and chemicals (Johnson 2009). BMPs that minimize sediment, such as riparian buffers, should be beneficial to snail populations (Herrig and Shute 2002, Johnson 2009). If water velocities are too high within culverts, slippage may occur, resulting in an inability to move through a culvert (Resh 2005, Rivera 2008). High velocity culverts can potentially serve as a biological filter, perhaps even selecting for more robust, invasive species of snail (Clennon and others 2007, Rivera 2008). Dissipating erosional energy, providing substrate heterogeneity, and implementing a culvert at a similar slope to the streambed should reduce stream velocity

and enable snail passage (Resh 2005). There is very little information on forestry BMPs in relation to snails, particularly in North America, and additional research is needed.

Butterflies and Moths

In North America, moths are the better Lepidoptera biodiversity indicator, as compared to butterflies in tropical regions (Summerville and others 2004). Vegetative changes (as a result of host plants), changes in light penetration, habitat heterogeneity, moisture, temperature, humidity, and canopy cover changes may alter butterfly and moth abundance, diversity, and community composition (Hamer and others 2003, Summerville and others 2004, Summerville and others 2009). Research suggests that studies showing a decline in lepidopteran biodiversity following harvesting may be a result of homogenized tree communities in secondary successional forests (Summerville and others 2008). Lepidoptera in Indiana were found to be more resilient to shelterwood harvests than clearcut harvests (Summerville 2013). Plant community composition, biogeographic history, and spatial heterogeneity of host plants for female oviposition are all important components in maintaining lepidopteran community structure (Summerville and others 2008). BMPs that maintain SMZs and promote a diversity of forest classes should safeguard moth and butterfly species by maintaining host plant requirements and protecting and potential streamside changes (Summerville and others 2009).

Based upon studies in other countries and with different species, forestry effects on butterflies are complex: some species are favored by light increases, while others can potentially be harmed. Hamer and others (2003) found that in Borneo's selectively logged areas, butterfly diversity increased, yet assemblages differed significantly. Some North American butterflies, such as skippers, thrive in sunny environments and managed disturbance can encourage localized skipper recolonization and increase skipper population numbers (Swengel 2001).

Stoneflies, Dragonflies, and Caddisflies

Potential threats for stoneflies, dragonflies, and caddisflies include sedimentation, vegetative alteration, and altered hydrology. Leaving riparian buffers have been shown to benefit these invertebrates, as buffers can stabilize benthic food webs, ensure temperature stability, and maintain detritus input, light levels, and algae production. Quinn and others (2004) found that clearcut sites with continuous riparian buffers had stable populations of caddisflies while clearcut sites with patchy buffers had lower numbers. Davies (1994) found that SMZ width influenced stoneflies in Australia, and that they were most affected by buffers less than 30 m. Although not in North America, a study in South Africa suggested that riparian buffers should be at least 20 m to protect riparian vegetation for dragonfly populations, as smaller buffers could shift species compositions to that of a generalist community (Samways 1996). Culverts may affect caddisfly flight. Blakely and others (2003) found that road culverts in New Jersey reduced upstream adult caddisfly abundance by 250 percent when compared to downstream, but as the upstream numbers did not fall to zero, culverts were not found to be absolute dispersal barriers. Harding and others (2005) found that culvert hindrance of upstream flight in caddisflies was predominately due to urban surroundings, predation, confounding flight cues, and gradients in humidity, light, and temperature. In an urban environment, only 30 to 50 percent of caddisflies enter the culvert, and of these, about 10 to 30 percent do not reach the exit, partially due to spider predation (Harding and others 2005). Further research is needed to assess potential culvert impacts in a forested environment. BMPs like SMZs that reduce sediment, manage for habitat heterogeneity, and preserve litter food sources will benefit these species.

Beetles

Of the petitioned beetles, 17 out of 18 are cave beetles. Potential threats and appropriate BMPs previously mentioned for amphipods and isopods can also be applied to cave beetles, such as a recommended 50 foot SMZ. Cobblestone tiger beetles (*Cicindela marginipennis*) have and very restricted habitat and are adapted to natural disturbances such as flooding, fire, and ice scouring (Hudgins and others 2012, Leonard and Bell 1999). Although it is unlikely to be affiliated with forestry operations, the Cobblestone tiger beetle's need for low vegetation and cobblestone habitat could make it susceptible to sediment deposits burying cobble and allowing vegetation growth (Hudgins and others 2011). BMPs that address sedimentation issues (i.e., riparian buffers) would benefit this species by minimizing channelization and substrate alteration.

Vascular and Nonvascular Plants

The greatest potential threat for riparian vascular plants is direct removal, so SMZs would likely greatly benefit the species listed in this petition. Riparian buffers have been shown to benefit riparian vascular species. Hibbs and Bower (2001) examined an Oregon riparian forest and found conversion to a riparian buffer maintained species structure and composition. Habitat alteration is the greatest potential threat for nonvascular plants in this petition. Two aquatic species (moss and hornwort) are limited by sediment and temperature increases. Two riparian liverworts are limited by canopy removals. BMPs that reduce sediment and SMZs appear to be beneficial for these species.

CONCLUSION

In conducting our literature review, we found that the effects of BMPs on the vast majority of the species listed in this petition have not been evaluated. Even interpretations of effects on larger taxonomic classifications require additional research, particularly in the southeastern United States. Numerous studies on potential forestry impacts on wildlife did not employ BMPs, including those studies cited in the 2011 petition, of which ascribed negative impacts to forestry related practices. Negative harvesting impacts that were ascribed to 51 percent of the species cited studies that were conducted in the absence of BMPs, and many of these same species were not affiliated with forestry operations.

Forestry BMPs are designed to limit sediment, nutrients, and pollutants from entering the streams, protecting water quality and safeguarding riparian habitat. Typically, the most important operations for reducing sediment involve roads, skid trails, and stream crossings. Since many of the species are potentially negatively affected by sediment and state BMP programs are specifically directed towards the development and implementation of forestry BMP on such operational areas, BMPs almost undoubtedly benefit species negatively affected by sediment.

Riparian buffers and SMZs have been consistently shown to benefit riparian and stream dwelling species. They provide heterogeneous vegetation and riparian habitat, trap sediment and attached nutrients, provide thermal protection for streams, and serve as low impact zones in managed landscapes. The simple act of maintaining SMZs, as is recommended by all southeastern states, should enhance the habitat and stream conditions for many of these species. Sensitive areas such as karsts and sinkholes should generally be avoided, and riparian buffers should be considered during forestry operations.

Stream crossing BMPs, such as portable panel bridges or geoweb fords, which do not restrict channel flow, are a beneficial BMPs to riparian wildlife that may be impacted by channel restrictions, scouring, habitat homogeneity, and velocity changes. However, culvert options that reduce or eliminate potential mobility impacts on wildlife are available.

In providing economic incentives to landowners, timber harvesting maintains forested land and retains forested buffers along streams. Many land use changes, such as urbanization, had a greater potential to negatively impact wildlife populations. Forest management can reduce fragmentation, maintain riparian buffers, and control sediment in ways that may not necessarily be required by other land use changes. In maintaining forested habitat, applying appropriate BMPs to reduce sediment, and by minimizing and implementing appropriate and adequately designed stream crossings, riparian wildlife can be safeguarded and preserved in the presence of forestry operations.

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LITERATURE CITED

Adams, T.; Hook, D.D. 1993. Implementation and effectiveness monitoring of forestry best management practices on harvested sites in South Carolina. Best Management Practices Monitoring Report Number: BMP-1. Columbia, SC: South Carolina Forestry Commission, South Carolina State Documents Depository. 32 p.

Alix, D.M.; Anderson, C.J.; Grand, J.B.; Guyer, C. 2014. Evaluating the effects of land use on headwater wetland amphibian assemblages in Coastal Alabama. Wetlands. 34(5): 917-26.

Anbumozhi, V.; Radhakrishnan, J.; Yamaji, E. 2005. Impact of riparian buffer zones on water quality and associated management considerations. Ecological Engineering. 24(5): 517-523.

Anderson, C.J.; Lockaby, B.G. 2011. The effectiveness of forestry best management practices for sediment control in the southeastern United States: a literature review. Southern Journal of Applied Forestry. 35(4): 170-177.

Aust, W.M.; Carroll, M.B.; Bolding, M.C.; Dolloff, C.A. 2011. Operational forest stream crossings effects on water quality in the Virginia Piedmont. Southern Journal of Applied Forestry. 35(3): 123-130.

Blakely, T.; Harding, J.; McIntosh, A. 2003. Impacts of urbanisation in Okeover stream, Christchurch. New Zealand: Freshwater Ecology Research Group, Department of Zoology, University of Canterbury. 25p.

Clayton, J.L.; Stihler, C.W.; Wallace, J.L. 2001. Status of and potential impacts to the freshwater bivalves (Unionidae) in Patterson Creek, West Virginia. Northeastern Naturalist. 8(2): 179-88.

Clennon, J.A.; King, C.H.; Muchiri, E.M.; Kitron, U. 2007. Hydrological modelling of snail dispersal patterns in Msambweni, Kenya and potential resurgence of *Schistosoma haematobium* transmission. Parasitology. 134(05): 683-93.

Constantine, N.L.; Campbell, T.A.; Baughman, W.M [and others]. 2004. Effects of clearcutting with corridor retention on abundance, richness, and diversity of small mammals in the Coastal Plain of South Carolina, USA. Forest Ecology and Management. 202(1-3): 293-300.

David, B.O.; Tonkin, J.D.; Taipeti, K.W.T.; Hokianga, H.T. 2014. Learning the ropes: mussel spat ropes improve fish and shrimp passage through culverts. Journal of Applied Ecology. 51(1): 214-23.

Davies, P.; Nelson, M. 1994. Relationships between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. Marine and Freshwater Research. 45(7): 1289-1305.

deMaynadier, P.G.; Hunter, M.L., Jr. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environmental Reviews. 3(3-4): 230-261.

deMaynadier, P.G., Hunter, M.L., Jr. 2000. Road effects on amphibian movements in a forested landscape. Natural Areas Journal. 20(1): 56-65.

Elderd, B.D.; Nott, M.P. 2008. Hydrology, habitat change and population demography: an individual-based model for the endangered Cape Sable seaside sparrow *Ammodramus maritimus mirabilis*. Journal of Applied Ecology. 45(1): 258-268.

Feurich, R.; Boubée, J.; Olsen, N.R.B. 2012. Improvement of fish passage in culverts using CFD. Ecological Engineering. 47: 1-8.

Foley, D.H. 1994. Short-term response of herpetofauna to timber harvesting in conjunction with streamside-management zones in seasonally-flooded bottomland-hardwood forests of southeast Texas. College Station, TX: Texas A&M University, Department of Wildlife and Fisheries Sciences. 93 p.

Foster, H.R.; Keller, T.A. 2011. Flow in culverts as a potential mechanism of stream fragmentation for native and nonindigenous crayfish species. Journal of the North American Benthological Society. 30(4): 1129-1137.

Graynoth, E. 1979. Effects of logging on stream environments and faunas in Nelson. New Zealand Journal of Marine and Freshwater Research. 13(1): 79-109.

Hamer, K.C.; Hill, J.K.; Benedick, S. [and others]. 2003. Ecology of butterflies in natural and selectively logged forests of northern Borneo: the importance of habitat heterogeneity. Journal of Applied Ecology. 40(1): 150-162.

Harding J.S.; Neumegen, R.E.; van den Braak I.L. 2005. Where have all the caddis gone? The role of culverts, and spiders [Abstract]. In: American geophysical union spring meeting abstracts. Abstract No.: NB14C-01. Christchurch, New Zealand: School of Biological Sciences, University of Canterbury. [Not paged].

Herrig J.; Shute P. 2002. Aquatic animals and their habitats. In: Wear, D.N.; Greis, J.G., eds. Southern forest resource assessment, Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 537-580 p.

Hibbs, D.E.; Bower, A.L. 2001. Riparian forests in the Oregon Coast Range. Forest Ecology and Management. 154(1-2): 201-213.

Holsinger, J.R. 1972. Freshwater amphipod crustaceans (Gammaridae) of North America. Biota of freshwater ecosystems, identification manual. Cincinnati, OH: U.S. Environmental Protection Agency, Office of Research and Development, Aquatic Biology Section. 89 p.

Hotchkiss, R.H.; Frei, C.M. 2007. Design for fish passage at roadway-stream crossings: synthesis report. Report No.: FHWA-HIF-07-033. McClean, VA: U.S. Department of Transportation, Federal Highway Administration. 280 p.

Hudgins, R.; Norment, C.; Schlesinger, M. 2012. Assessing detectability for monitoring of rare species: a case study of the cobblestone tiger beetle (*Cicindela marginipennis* Dejean). Journal of Insect Conservation. 16(3): 447-55.

Hudgins, R.; Norment, C.; Schlesinger, M.D.; Novak, P.G. 2011. Habitat selection and dispersal of the Cobblestone Tiger Beetle (*Cicindela marginipennis* Dejean) along the Genesee River, New York. The American Midland Naturalist. 165(2): 304-318.

Ice, G. 2004. History of innovative best management practice development and its role in addressing water quality limited waterbodies. Journal of Environmental Engineering. 130(6): 684-689.

Jensen, K.M. 2014. Velocity reduction factors in near boundary flow and the effect on fish passage through culverts. Provo, UT: Brigham Young University, Department of Civil and Environmental Engineering. 44 p. M.S. thesis.

Johnson, P.D. 2009. Sustaining America's aquatic biodiversity: freshwater snail biodiversity and conservation. Publication No: 420-530. Blacksburg, VA: Virginia Polytechnic Institute and State University, Virginia Cooperative Extension. 7 p.

Johnson, W.C. 2000. Tree recruitment and survival in rivers: influence of hydrological processes. Hydrological Processes. 14(16-17): 3051-3074.

Jones, E.B.D.; Helfman, G.S.; Harper, J.O.; Bolstad, P.V. 1999. Effects of riparian forest removal on fish assemblages in Southern Appalachian Streams. Conservation Biology. 13(6): 1454-1465.

Kidd, K.R.; Aust, W.M.; Copenheaver, C.A. 2014. Recreational Stream Crossing Effects on Sediment Delivery and Macroinvertebrates in Southwestern Virginia, USA. Environmental Management. 54(3): 505-16.

Kinzel, P.J. Nelson, J.M.; Heckman, A.K. 2009. Response of sandhill crane (*Grus canadensis*) riverine roosting habitat to changes in stage and sandbar morphology. River Research and Applications. 25(2): 135-152.

Lakel, W.A.; Aust, W.M.; Dolloff, A.C. 2006. Seeing the trees along the streamside: Forested streamside management zones are one of the more commonly recommended forestry best management practices for the protection of water quality. Journal of Soil and Water Conservation. 61(1): 22A-29A.

Leonard, J.G.; Bell R.T. 1999. Northeastern Tiger Beetles: a field guide to Tiger Beetles of New England and Eastern Canada. Boca Raton, FL: CRC Press LLC. 192 p.

Lodge, D.M.; Taylor, C.A.; Holdich, D.M.; Skurdal, J. 2000. Reducing impacts of exotic crayfish introductions. Fisheries. 25(8): 21-3.

Louca, V.; Ream, H.M.; Findlay, J.D.; Latham, D.; Lucas, M.C. 2014. Do culverts impact the movements of the endangered white-clawed crayfish? Knowledge and Management of Aquatic Ecosystems. Number 414, 2014(14). 17 p.

Mengak, M.T.; Laerm, J. 2007. Hispid Cotton Rat: *Sigmodon hispidus*. In: Trani, M.K.; Ford, W.M.; Chapman, B.R., eds. The land manager's guide to the mammals of the South. Durnham, NC: The Nature Conservancy, Southeastern Region: 374-379.

Moring, J.R. 1982. Decrease in stream gravel permeability after clear-cut logging: an indication of intragravel conditions for developing salmonid eggs and alevins. Hydrobiologia. 88(3): 295-8.

Moseley, K.; Ford, W.M.; Edwards, J. 2009. Local and landscape scale factors influencing edge effects on woodland salamanders. Environmental and Monitoring Assessment. 151(1-4): 425-435.

Miller, D.A.; Thill, R.E.; Melchiors, M.A.; Wigley, T.B.; Tappe, P.A. 2004. Small mammal communities of streamside management zones in intensively managed pine forests of Arkansas. Forest Ecology and Management 203(1-3): 381-393.

Parkyn, S.M.; Collier, K.J. 2004. Interaction of press and pulse disturbance on crayfish populations: flood impacts in pasture and forest streams. Hydrobiologia. 527(1): 113-24.

Poole, K.E.; Downing, J.A. 2004. Relationship of declining mussel biodiversity to stream-reach and watershed characteristics in an agricultural landscape. Journal of the North American Benthological Society. 23(1): 114-25.

Quinn, J.M.; Boothroyd, I.K.G.; Smith, B.J. 2004. Riparian buffers mitigate effects of pine plantation logging on New Zealand streams: 2. Invertebrate communities. Forest Ecology and Management. 191(1-3): 129-146.

Quist M.C.; Schultz, R.D. 2014. Effects of management legacies on stream fish and aquatic benthic macroinvertebrate assemblages. Environmental Management. 54(3): 449-464.

Resh, V.H. 2005. Stream crossings and the conservation of diadromous invertebrates in South Pacific island streams. Aquatic Conservation: Marine and Freshwater Ecosystems. 15(3): 313-317.

Rivera, C.J.R. 2008. Obstruction of the upstream migration of the invasive snail *Cipangopaludina chinensis* by high water currents. Summer UNDERC Project. South Bend, IN: University of Notre Dame, Department of Biological Sciences, Galvin Life Sciences Center. 13 p

Rudolph, D.C., Dickson, J.G. 1990. Streamside zone width and amphibian and reptile abundance. The Southwestern Naturalist. 35(4): 472-476.

Russell, K.R., Hanlin, H.G., Wigley, T.B., Guynn, D.C., Jr. 2002. Responses of isolated wetland herpetofauna to upland forest management. The Journal of Wildlife Management. 66(3): 603-617.

Samways, M.J., Steytler, N.S. 1996. Dragonfly (Odonata) distribution patterns in urban and forest landscapes, and recommendations for riparian management. Biological Conservation. 78(3): 279-288.

Semlitsch, R.D., Bodie, J.R. 2003. Biological criteria for buffer zones around wetlands and riparian habitats for amphibians and reptiles. Conservation Biology. 17(5): 1219-1228.

Shepard, J.P. 2006. Water quality protection in bioenergy production: the US system of forestry best management practices. Biomass and Bioenergy. 30(4): 378-384.

Sterrett, S.C., Smith, L.L., Schweitzer, S.H., Maerz, J.C. 2010. An assessment of two methods for sampling river turtle assemblages. Herpetological Conservation and Biology. 5(3): 490-497.

Summerville, K.S.; Courard-Hauri D.; Dupont, M.M. 2009. The legacy of timber harvest: do patterns of species dominance suggest recovery of lepidopteran communities in managed hardwood stands? Forest Ecology and Management. 259(1): 8-13.

Summerville, K.S.; Dupont, M.M.; Johnson, A.V.; Krehbiel, R.L. 2008. Spatial dtructure of forest Lepidopteran communities in oak hickory forests of Indiana. Environmental Entomology. 37(5): 1224-1230.

Summerville, K.S., Ritter, L.M., Crist, T.O. 2004. Forest moth taxa as indicators of lepidopteran richness and habitat disturbance: a preliminary assessment. Biological Conservation. 116(1): 9-18.

Summerville, K.S. 2013. Forest lepidopteran communities are more resilient to shelterwood harvests compared to more intensive logging regimes. Ecological Applications. 23(5): 1101-1112.

Swengel, A.B. 2001. A literature review of insect responses to fire, compared to other conservation managements of open habitat. Biodiversity and Conservation. 10(7): 1141-1169.

Thorp, J.H.; Rogers, C.D. 2011. Field guide to freshwater invertebrates of North America. San Diego, CA: Elsevier Inc. 274 p.

Watters, T.G.; Hoggarth, M.A.; Stansbery, D.H. 2009. The freshwater mussel of Ohio. Columbus, OH: The Ohio State University Press. 421 p.

Weir, D.; Greis, J. 2002. Southern forest resource assessment: summary report. Gen. Tech. Rep. SRS-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 103 p.

Wigley, T.B.; Roberts, T.H. 1997. Landscape-level effects of forest management on faunal diversity in bottomland hardwoods. Forest Ecology and Management. 90(2-3): 141-154.

Zokaites C.; Brown, T.; Wills, G. [and others]. 1997. Living on karst: a reference guide for landowners in limestone regions. Glen Allen, VA: Cave Conservancy of the Virginias. 33 p.

FOREST STRUCTURE OF OAK PLANTATIONS AFTER SILVICULTURAL TREATMENT TO ENHANCE HABITAT FOR WILDLIFE

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Abstract—During the past 30 years, thousands of hectares of oak-dominated bottomland hardwood plantations have been planted on agricultural fields in the Mississippi Alluvial Valley. Many of these plantations now have closed canopies and sparse understories. Silvicultural treatments could create a more heterogeneous forest structure, with canopy gaps and increased understory vegetation for wildlife. Lack of volume sufficient for commercial harvest in hardwood plantations has impeded treatments, but demand for woody biomass for energy production may provide a viable means to introduce disturbance beneficial for wildlife. We assessed forest structure in response to prescribed pre-commercial perturbations in hardwood plantations resulting from silvicultural treatments: 1) row thinning by felling every fourth planted row; 2) multiple patch cuts with canopy gaps of <1 ha; and 3) tree removal on intersecting corridors diagonal to planted rows. These 3 treatments, and an untreated control, were applied to oak plantations (20 - 30 years post-planting) on three National Wildlife Refuges (Cache River, AR; Grand Cote, LA; and Yazoo, MS) during summer 2010. We sampled habitat using fixed-radius plots in 2009 (pre-treatment) and in 2012 (post-treatment) at random locations. Retained basal area was least in diagonal corridor treatments but had greater variance in patch-cut treatments. All treatments increased canopy openness and the volume of coarse woody debris. Occurrence of birds using early successional habitats was greater on sites treated with patch cuts and diagonal intersection. Canopy openings on row-thinned stands are being filled by lateral crown growth of retained trees whereas patch cut and diagonal intersection gaps appear likely to be filled by regenerating saplings.

INTRODUCTION

Afforestation has been undertaken by the U.S. Fish and Wildlife Service, state wildlife agencies, other conservation groups, and private landowners (often in conjunction with U.S. Department of Agriculture programs such as Wetland Reserve Program) to initiate restoration of forested wetlands on thousands of hectares within the Mississippi Alluvial Valley. Planting hardwood tree seedlings or seeds on restoration sites has resulted in thousands of hectares of oak-dominated, bottomland hardwood plantations. Many of these initial plantings are >20 years old. These maturing stands are often entering the stem-exclusion stage of stand development with closed canopies and sparse understories (Johnson 2004, Oliver and Larson 1996). During this stage, competition for light hinders the growth of canopy trees. Silvicultural treatments prescribed to enhance structural heterogeneity (both vertical and horizontal) and increase the floristic complexity within these stands may increase availability of food and cover for wildlife species within these stands. Although the effects of forest management on forest structure have been examined (Lorimer 1989, Meadows 1996) and the suitability of this resultant habitat for wildlife species has been evaluated (Twedt 2012, Twedt and Somershoe 2009, Wigley and Roberts 1994), little is known of the effects of forest management within relatively young (<30 year-old) plantations. Despite lack of empirical studies, enhanced wildlife habitat and improved timber production are expected to result from prescribed silvicultural treatments.

Analogous forest modification has been ongoing within coniferous forests where managers have used early-stage thinning in young (33-45 years old) stands to emulate late-successional forest conditions: Thinning was deemed successful as a preliminary restoration treatment (Plummer and others 2012). Although we are not advocating late-successional conditions within young (20-30 years old) bottomland hardwood plantations, silvicultural treatment may result in more rapid attainment of stand conditions desired for wildlife - as identified by the Lower Mississippi Valley Joint

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Venture (LMVJV Forest Resource Conservation Working Group 2007).

Despite the perceived benefit from reducing canopy cover within these hardwood stands, silvicultural treatments have rarely been undertaken because most young oak-dominated plantations lack sufficient volume of forest products (timber or pulpwood) for commercial harvest. However, where markets exist, harvest for woody biomass may provide an economically viable means to introduce beneficial disturbance in hardwood plantations. Because prescribed perturbations in young, oak-dominated, hardwood plantations are not common, subsequent stand development has not been widely studied. Evaluating stand development and wildlife response to prescribed silvicultural treatments is needed to inform development of proactive silvicultural treatments for oak-dominated hardwood plantations within the Mississippi Alluvial Valley.

OBJECTIVES

This study was intended to evaluate pre-treatment and post-treatment forest stand conditions within oak-dominated hardwood plantations as a base upon which subsequent stand development can be evaluated. We sought to gain insight regarding the ability of silvicultural prescriptions to improve wildlife habitat within hardwood plantations that differ in amount and configuration of canopy removed. We evaluated pre-commercial silvicultural treatments applied to oak-dominated hardwood plantations on National Wildlife Refuges (NWR) to assess initial changes in forest structure, woody species recruitment, and avian abundance.

METHODS

Study Areas

We identified >4250 ha that were afforested before 1991 through query of a forest restoration database maintained by the Lower Mississippi Valley Joint Venture. Most restorations on federally owned land were on Tensas River NWR (~900 ha), Theodore Roosevelt NWR Complex (~900 ha), Central Louisiana NWR Complex (~670 ha), and Cache River NWR (~300 ha). From these restorations, cooperating refuge personnel selected 3 plantations with density and height of trees such that prescribed silvicultural treatments would likely improve forest structure for wildlife by increasing within-stand structural heterogeneity. Selected study sites were:

Cache River NWR (Fig. 1) – A site of ~88 ha that was planted in 1990, located ~8 km north of Biscoe, AR. UTM coordinates (NAD83; Zone 15N): N3862700; E649190; S3861600; W648140,

Grand Cote NWR – A site of ~190 ha that was planted in 1990, located ~ 5 km west of Marksville, LA at T2N, R3E, Section 24: N3445800; E585660; S3443640; W585480, and

Yazoo NWR – A site of ~40 ha that combined small, adjoining oak plantings that were established during February or March of 1981, 1987, and 1990. Located ~18 km SW of Hollandale, MS: N3661900; E686180; S3660600; W685460.

Prior to implementation of treatments, forest managers randomly selected 10 permanent plots, using Universal Transverse Mercator (UTM) coordinates in the 1983 North American Datum (NAD83), within each treated stand at each study site (10 plots per stand = 40 plots per study site). Plot centers were located using hand held global positioning systems (GPS). To facilitate relocation, plot centers were marked with buried steel rebar or flat fiberglass pole and adjacent trees were basally marked with numbered aluminum tags. Notably, compared to post-treatment assessments which used international system (SI) units, pre-treatment data included measurement of smaller diameter trees within smaller sample plots and were recorded using English measurement units.

Pre-treatment Assessment

Within 0.05 acre circular sample plots, relative canopy closure and ground cover were visually estimated within 10 percent increments. At the Yazoo site, canopy closure was also assessed using hemispherical photography obtained from vertical projection of camera located 5 feet above forest floor (Chianucci and Cutini 2012). For each tree within sampled plot with diameter at breast height (dbh) ≥2 inch, observers recorded species, dbh, total tree height, crown class (dominant, co-dominant, intermediate, or suppressed), live crown ratio (10 percent increments of tree height that supported live green foliage), tree condition (1= no dieback; 2 = lower crown dieback; 3 = <1/3 crown dieback; 4 = >1/3 crown dieback; 5 = dead, twigs remain; 6 = dead, large limbs remain; 7 = dead, bole only; 8 = down wood ≥4 inch diameter), and distance (feet) and azimuth (degrees) from plot center. Diameters were recorded to 0.01 inch at Cache River and Yazoo, but to 1 inch at Grand Cote and are reported as the dbh of the tree of average basal area (quadratic mean diameter; QMD).

Tree regeneration was assessed within a nested 0.01 acre circular plot and recorded by number and species of seedlings (<3 feet tall) and saplings (>3 feet tall with dbh <2 inch). The relative volume (percent of vegetation that filled the cylindrical space) of each regeneration plot within 2 height classes (0-3 feet; 3-6 feet) was estimated in 10 percent increments for 6 vegetation classes (trees/shrubs; forbs/ferns; grass/sedge; blackberry; vines; other).

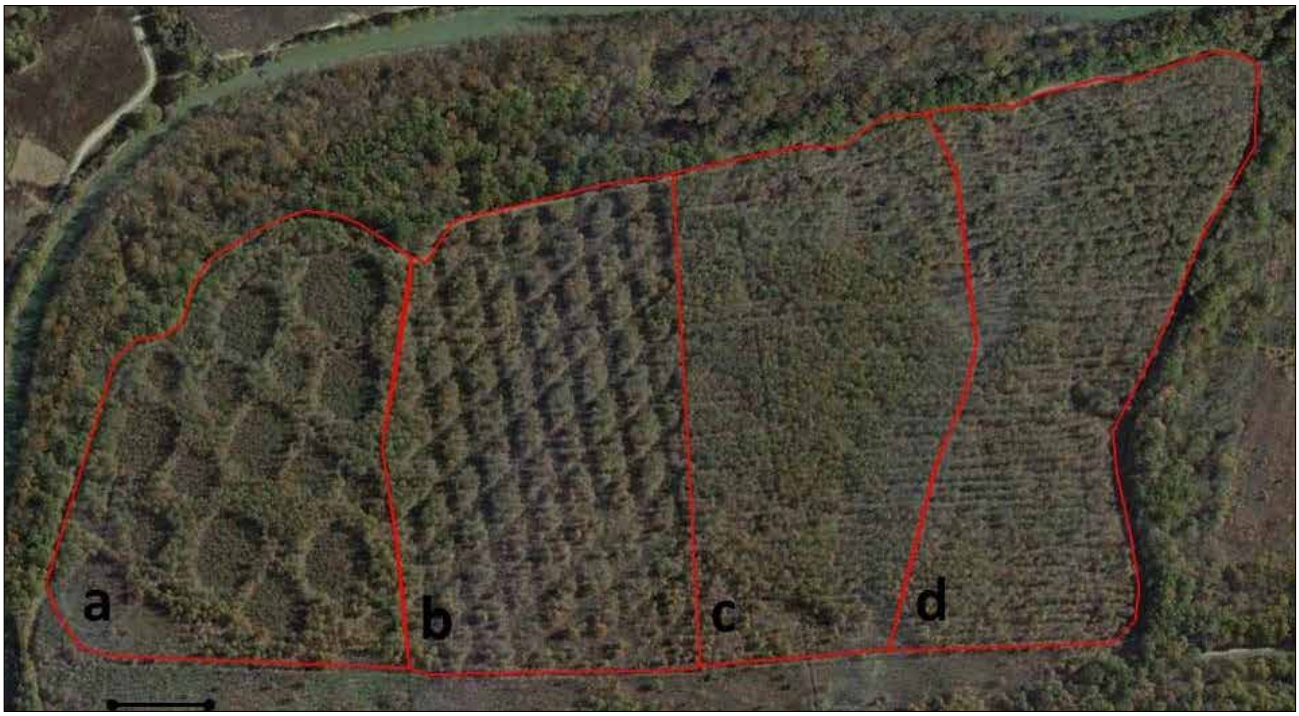


Figure 1—Aerial view (2012) of 4 treatments applied to ≥20 year-old oak plantation during 2010 to enhance wildlife habitat on Cache River National Wildlife Refuge (NWR), Arkansas. Treatments from left to right were: (a) variable patch clearcuts, (b) diagonal corridor thinning, (c) untreated control, and (d) single (1 of 4) row thinning.

Treatments

Each study site was divided and quadrates subjected to 1 of 4 treatments during summer 2010: (1) control with no harvest, (2) row thinning, (3) patch clearcuts, and (4) diagonal cross-row harvest. Due to the small acreage and relatively small volume of timber harvested during treatments, all treatments were pre-commercial with trees felled and left in situ as woody debris.

The long-term desired objective of these silvicultural treatments was to move stand structure toward desired forest conditions for wildlife which include: 1) maintaining or increasing species diversity, 2) increasing vertical and horizontal structural diversity, 3) retaining or increasing tree cavities, snags, and coarse woody debris, and 4) ensuring adequate (≥1000 stems/ha) regeneration of shade-intolerant tree species on 40 percent of stand (table 1; LMVJV Forest Resource Conservation Working Group 2007). Presumably, achieving these conditions will require major disturbance from a combination of clustered harvest (e.g., small, patch clearcuts) and an area-wide thinning (e.g., removal of individual trees and/or rows of trees). As such, one appropriate prescription may be a variable harvest that combines irregularly shaped group cuts and patch clearcuts (<2 ha) and thinning of the remaining stand. Recognizing that such a prescription may require considerable effort by forest managers, we opted to use 3 more easily implemented silvicultural treatments:

Row thinning—wherein ~25 percent of the existing canopy on treated stand was prescribed to be removed via felling of all trees within every fourth row of the original planting. Although some increase in species diversity and structural complexity was expected within treated stands, this treatment was presumed more beneficial for timber production than for wildlife habitat. We anticipated increased growth and vigor of retained trees but that understory development within the narrow openings created by removal of a single row of trees would be limited by lateral crown growth of residual trees.

Variable patch clearcut—wherein ~30-40 percent of the existing canopy on treated stand was prescribed to be removed via in patch clearcuts that ranged in area from 0.1 ha to 0.8 ha. Prescription stipulated that ≥2 patch clearcuts within each treated stand must be >0.4 ha, that ≥20 m of forest must be retained between edges of patch clearcuts, and that openings should be distributed throughout the entirety of the treated stand. Anticipated outcome was that natural regeneration and stump-sprouts within opening will increase species diversity and structural complexity within treated stand. Increased growth and vigor was anticipated for residual trees along edges of openings. Increased competition among regenerating saplings within openings may result in better bole quality of future timber than was present in felled trees.

Table 1—Mean forest metrics within ≥20 year-old oak plantations in Arkansas, Louisiana, and Mississippi during 2009 pre-treatment assessment of stands prescribed for treatment to enhance wildlife habitat. Prescribed treatments included untreated control, diagonal corridor thinning, variable patch clearcuts, and single (1 of 4) row thinning. P-values are for $F_{3,6}$ from analysis of variance comparing treatments

Metric	Control	Diagonal	Patch	Row	P
Canopy Closure (%; ocular)	77.0 ± 3.2	69.0 ± 3.3	67.3 ± 3.8	68.6 ± 4.3	0.218
Canopy Closure (%; photo)[a]	74.0	73.7	74.7	76.4	
Ground Cover (%)	28.3 ± 4.6	51.8 ± 4.9	54.0 ± 5.5	31.5 ± 5.5	0.118
Basal Area (m^2/ha)	14.1 ± 2.5	14.1 ± 4.1	12.1 ± 1.9	14.7 ± 3.8	0.678
Density (stems >5 cm / ha)	866 ± 198	678 ± 154	652 ± 236	812 ± 162	0.087
QMD (cm)	15.1 ± 2.0	16.4 ± 2.6	16.2 ± 1.7	15.7 ± 3.3	0.694
Tree height (m)	10.8 ± 1.6	11.9 ± 2.7	10.6 ± 1.4	12.3 ± 2.7	0.427
Snags (stems dbh > 5 cm / ha)	13.2 ± 6.7	14.8 ± 6.8	1.6 ± 1.6	13.2 ± 7.5	0.789
Dead wood (m^3/ha)	0.42 ± 0.30	0.79 ± 0.46	0.01 ± 0.01	0.65 ± 0.53	0.740
Regeneration (stems/ha)	2372 ± 535	2438 ± 427	2743 ± 562	3583 ± 776	0.620
Shade intolerant regeneration	1433 ± 357	1516 ± 314	1952 ± 450	2232 ± 511	0.388
Shade tolerant regeneration	939 ± 298	922 ± 207	791 ± 192	1351 ± 407	0.763

[a]Yazoo NWR only.

Diagonal corridor thinning—wherein ~56 percent of overstory canopy was prescribed to be removed by cutting cross-diagonal swaths as wide as the average canopy tree height with residual leave-tree patches of width approximately twice the canopy height. Where possible, cross-diagonal cut swaths were oriented 60°-240° and 120°-300° to enhance sunlight penetration of openings, but generally harvest was implemented diagonal to the planted tree rows. Because tree rows may be of a single species due to original planting protocol, tree harvest diagonal to planted rows lessens the likelihood that all or a large percentage of the stems of any tree species are removed during harvest. Finally, because diagonal harvests are implemented in a crossing pattern (i.e., an X shape), the intersection of harvest diagonals may simulate larger canopy gaps such as those of group harvests or small patch clearcuts.

Control—wherein no trees were felled or otherwise manipulated. Because management of reforested bottomland hardwood stands is poorly understood, a portion of each study area was retained as an untreated control. Control stands were used to compare post-treatment stand conditions with conditions where no treatments were implemented.

At the Grand Cote NWR site, concern regarding Chinese tallow (*Triadica sebifera*) establishment within treated stands prompted managers to aerially apply Clearcast™ herbicide (imazamox, SePRO Corp., Carmel, IN) at the rate of 4.68 L/ha after silvicultural treatments (September 2010). Mortality of Chinese tallow, green ash (*Fraxinus pennsylvanica*), and black willow (*Salix nigra*) appeared to result from this treatment.

Post-treatment Assessment

During summer 2012, we used circular sampling plots of 0.05 ha (>twice the area of pre-treatment plots) that were centered at previously marked plot locations. If we were unable to relocate a permanent plot marker, either due to discrepancy in recorded coordinates or due to presence of harvest debris, we located sample plot at GPS coordinates recorded during pre-treatment assessment.

For each tree within sampled plots with dbh ≥6 cm (note slightly larger dbh than used for pre-treatment assessment), observers recorded the same metrics as recorded during pre-treatment assessments. We measured dbh using metric tree calipers and recorded to 1.0 cm. Distance and height were measured using a laser rangefinder and recorded to 0.1 m.

Tree regeneration and vegetation density estimates were assessed within four 20 m^2 circular plots (2.52 m radius) located at cardinal directions each 5 m from 0.05 ha plot center. Number and species of woody plants ≥0.5 m tall were recorded within 0.5 m height classes up to 2.5 m and taller stems were recorded within 1.0 m height classes. We visually estimated, within 1 percent increments, the relative percent of cylindrical space

(volume) that was filled by vegetation within 0-1 m and 1-2 m above ground level for the same 6 vegetation classes used during pre-treatment assessments. Finally, at each regeneration plot, percent canopy closure was estimated using a concave densiometer.

The 12 experimental units used for analysis were the 4 treated stands at 3 study site locations. We used analysis of variance (ANOVA; Proc GLM) to compare forest metrics among treatments (df = 3) and locations (df = 2) wherein the experimental error term was the treatment by location interaction (df = 6).

Bird Counts

From 2012-2014, within each treated stand we annually conducted 2 or 3 point counts for breeding birds between May 10 and June 30. Each count was 10-minutes in duration with detections recorded in time-distance intervals (Farnsworth and others 2005). Count locations were selected to optimize coverage of treated stands while maintaining independence of observations among bird counts. We tested for differences in bird communities among treatments, and among years post-treatment, using nonparametric multivariate analysis of variance (Anderson 2001). We used indicator species analysis (Dufréne and Legendre 1997) to evaluate the relationships between species and silvicultural treatments while blocking by location. Bird data were restricted to detections within 100 m and standardized to mean number of detections per species per count per year within a stand before analyses.

RESULTS

We detected no significant pre-treatment differences among treatments for any of the forest metrics measured (table 1); but stands prescribed for control and row thinning treatments may have had slightly greater tree density ($F_{3,6}$= 3.55, P = 0.09) and less ground cover ($F_{3,6}$= 2.98, P = 0.11). Study sites, however, differed markedly in pre-treatment structure (table 2). Grand Cote NWR had less canopy closure at just under 60 percent ($F_{2,6}$ = 13.33, P <0.01), less basal area with 9.0 m^2 per ha ($F_{2,6}$ = 14.73, P < 0.01), and less shade intolerant regeneration at only 43 stems/ha ($F_{2,6}$ = 26.03, P < 0.01). Greatest stem densities of trees >5 cm dbh at >1100 stems/ha ($F_{2,6}$ = 45.21, P<0.01) and greatest total regeneration of ~5600 stems/ha ($F_{2,6}$ = 18.14, P <0.01) were found at Cache River NWR. Yazoo NWR had the largest trees with mean height >15 m, QMD >20 cm, and mean basal area >19 m^2 per ha (table 2). Dead standing trees (snags) were rare (~1 percent of stems >5 cm dbh; table 1) and coarse woody debris was sparse (<1 m^3/ha; table 2).

Post-treatment stand conditions were assessed from 116 of the original 120 sample plots: Flooding prevented access to 1 sample plot in variable patch clearcut treatment at Grand Cote NWR and logistic-time constraints resulted in 3 permanent plots (1 each in control, variable patch clearcut, and diagonal corridor thinning treatments) not being resampled at Cache River NWR. Stands subjected to variable patch clearcut and diagonal corridor thinning had reduced canopy

Table 2—Mean forest metrics within ≥20 year-old oak plantations on National Wildlife Refuges (NWR) in Arkansas, Louisiana, and Mississippi during 2009 pre-treatment assessment of stands prescribed for treatment to enhance wildlife habitat. P-values are for $F_{2,6}$ from analysis of variance comparing study site locations

Metric	Cache River NWR	Grand Cote NWR	Yazoo NWR	P
Canopy Closure (ocular)	73.5 ± 3.1	59.4 ± 3.2	78.5 ± 2.6	0.006
Ground Cover	49.7 ± 5.2	43.5 ± 4.9	31.0 ± 3.7	0.212
Basal Area (m^2/ha)	13.1 ± 0.7	9.1 ± 0.3	19.2 ± 2.0	0.005
Density (stems dbh > 5 cm / ha)	1119 ± 58	540 ± 77	598 ± 57	<0.001
QMD (cm)	12.3 ± 0.3	14.8 ± 0.7	20.4 ± 0.8	<0.001
Tree height (m)	9.6 ± 0.2	9.0 ± 0.2	15.6 ± 1.1	<0.001
Snags (stems/ha)	9.9 ± 5.4	9.9 ± 5.1	12.4 ± 5.5	0.975
Dead wood (m^3/ha)	0.10 ± 0.05	0.59 ± 0.35	0.72 ± 0.45	0.613
Regeneration (stems/ha)	5591 ± 552	463 ± 190	2298 ± 323	0.003
Shade intolerant regeneration	3193 ± 398	43 ± 23	2113 ± 320	<0.001
Shade tolerant regeneration	2397 ± 277	420 ± 189	185 ± 43	0.006

closure (56 and 68 percent, respectively) compared to 92 percent canopy closure on post-treatment control stands ($F_{3,6}$ = 7.43, P=0.019; table 3). Density of trees with dbh ≥ 6 cm ($F_{3,6}$ = 3.96, P=0.071) appeared to be reduced and understory vegetation volume 0-1 m above ground ($F_{3,6}$ = 3.29, P=0.100) likely increased in response to silvicultural treatments (table 3). Density of snags did not differ among treatments ($F_{3,6}$ = 1.06, P=0.431), but relative to control stands, coarse woody debris appeared to be more abundant (table 3; $F_{3,6}$ = 3.42, P=0.093). Regeneration did not differ among treatments (table 3; $F_{3,6}$ < 2.36, P>0.17).

Larger trees remained present on Yazoo NWR post-treatment compared to other study sites (table 4), with taller trees ($F_{2,6}$=23.3, P=0.001), greater diameters ($F_{2,6}$ = 18.5, P=0.003), and more basal area ($F_{2,6}$ = 3.7, P=0.089). Greater understory vegetation volume 1-2 m above ground ($F_{2,6}$ = 7.80, P=0.021) and greater regeneration ($F_{2,6}$>40.2, P<0.001) were present at Cache River NWR (table 4). The number of snags at Grand Cote NWR increased markedly to 97 stems/ha after treatment ($F_{2,6}$ = 7.84, P=0.021), whereas coarse woody debris increased most at Yazoo NWR ($F_{2,6}$ = 6.67, P=0.03).

As one of the objectives of these treatments was to promote increased species diversity, we evaluated the density and stature of oaks ≥6 cm dbh (presumed to be planted), with those of non-oak species (presumed to be colonizers). Although density of non-oak species was low within stands treated with variable patch clearcuts (80 stems/ha) and diagonal corridor thinning (84 stems/ha), density did not increase in response to treatments (table 5; $F_{3,6}$ = 2.85, P=0.127). Moreover, non-oak species were markedly shorter, with heights <9 m compared to oaks at >10 m, and had smaller diameters, with QMD of <10 cm compared to oaks at >16 cm (table 5).

During point counts of breeding birds we detected 42 bird species (table 6). Empirically, we detected fewer birds and species within control stands than within treated stands. For 28 species with >2 detections, we noted bird community differences among treatments (F=2.93, P<0.01) but no differences among years since treatment (F=1.34, P = 0.15). Species associated with control stands (Indicator Value [IV]>24.6, P<0.09) included Acadian flycatcher (*Empidonax virescens*), eastern towhee (*Pipilo erythrophthalmus*), and red-eyed vireo (*Vireo olivaceus*). Two species, Carolina wren (*Thryothorus ludovicianus*) and summer tanager (*Piranga rubra*), were associated with row thinning (IV>36.1, P<0.06). Species commonly associated with early successional habitat were associated with the other 2 treatments: brown-headed cowbird (*Molothrus ater*), indigo bunting (*Passerina cyanea*), and white-eyed vireo (*Vireo griseus*) were associated with patch clearcuts (IV>33.7, P<0.09), whereas yellow-breasted chat (*Icteria virens*) was associated with diagonal corridor thinning (IV = 42.0, P<0.01).

Table 3—Mean forest metrics within ≥20 year-old oak plantations in Arkansas, Louisiana, and Mississippi during 2012 post-treatment assessment of stands treated to enhance wildlife habitat during 2010. Prescribed treatments included untreated control, diagonal corridor thinning, variable patch clearcuts, and single (1 of 4) row thinning. P-values are for $F_{3,6}$ from analysis of variance comparing treatments

Metric	Control	Diagonal	Patch	Row	P
Canopy Closure (%; densiometer)	91.8 ± 1.3	67.6 ± 3.5	55.9 ± 3.8	78.6 ± 2.2	0.019
Ground Cover (%)	28.6 ± 2.8	49.8 ± 3.5	55.7 ± 3.5	33.4 ± 2.6	0.172
Vegetation volume 0-1 m	18.6 ± 1.4	36.3 ± 2.5	41.9 ± 2.8	22.4 ± 1.5	0.100
Vegetation volume 1-2 m	10.0 ± 0.8	14.6 ± 1.3	16.2 ± 1.3	11.3 ± 0.9	0.342
Basal Area (m^2/ha)	11.1 ± 0.8	8.3 ± 1.2	6.4 ± 0.9	9.6 ± 0.9	0.254
Density (stems dbh ≥ 6 cm/ha)	588 ± 45	323 ± 34	282 ± 42	467 ± 48	0.071
QMD (cm)	15.9 ± 0.7	17.4 ± 0.9	16.3 ± 1.2	16.8 ± 1.0	0.793
Tree height (m)	11.3 ± 0.5	12.7 ± 0.8	11.0 ± 0.6	13.1 ± 0.7	0.256
Snags (stems/ha)	27.1 ± 9.7	40.0 ± 12.7	19.3 ± 7.7	68.0 ± 22.6	0.431
Dead wood (m^3/ha)	1.4 ± 0.5	11.6 ± 3.1	12.8 ± 3.0	13.0 ± 2.8	0.093
Regeneration (stems ≥0.5 m/ha)	1732 ± 449	1832 ± 354	1942 ± 485	2596 ± 440	0.171
Shade intolerant regeneration	924 ± 311	797 ± 216	1071 ± 262	1312 ± 294	0.222
Shade tolerant regeneration	808 ± 168	1034 ± 182	871 ± 320	1283 ± 228	0.267

Table 4—Mean forest metrics within ≥20 year-old oak plantations on National Wildlife Refuges (NWR) in Arkansas, Louisiana, and Mississippi during 2012 post-treatment assessment of stands treated to enhance wildlife habitat during 2010. P-values are for $F_{2,6}$ from analysis of variance comparing study site locations

Metric	Cache River NWR	Grand Cote NWR	Yazoo NWR	P
Canopy Closure (%; densiometer)	68.7 ± 3.1	79.2 ± 2.1	72.4 ± 2.7	0.368
Ground Cover	48.5 ± 3.2	35.9 ± 2.3	41.3 ± 2.7	0.502
Vegetation volume 0-1 m	37.7 ± 2.5	24.2 ± 1.4	27.7 ± 1.8	0.249
Vegetation volume 1-2 m	19.6 ± 1.1	12.0 ± 0.8	7.8 ± 0.7	0.021
Basal Area (m^2/ha)	7.6 ± 0.8	7.2 ± 0.5	11.7 ± 1.0	0.089
Density (stems dbh ≥ 6 cm/ha)	479 ± 49	419 ± 37	350 ± 35	0.424
QMD (cm)	13.5 ± 0.6	15.0 ± 0.4	21.2 ± 0.8	0.003
Tree height (m)	10.5 ± 0.4	9.8 ± 0.1	15.8 ± 0.5	0.002
Snags (stems/ha)	3.8 ± 2.7	96.9 ± 18.2	14.8 ± 4.6	0.021
Dead wood (m^3/ha)	6.8 ± 1.1	4.8 ± 0.7	17.4 ± 3.5	0.030
Regeneration (stems ≥0.5 m/ha)	4463 ± 413	865 ± 144	897 ± 151	<0.001
Shade intolerant regeneration	2470 ± 272	61 ± 15	631 ± 127	<0.001
Shade tolerant regeneration	1993 ± 246	804 ± 140	266 ± 63	<0.001

Table 5—Mean stem density of trees with diameter at breast height (dbh) ≥6 cm, their quadratic mean diameter, and average height for presumably planted oaks (*Quercus* spp.) and presumed colonizing non-oak canopy tree species in ≥20 year-old oak plantations in Arkansas, Louisiana, and Mississippi in 2009 before application, and in 2012 after application of silvicultural treatments to enhance wildlife habitat during 2010

	2009		2012					
	Density (stems >5 cm/ha)		Density (stems ≥6 cm/ha)		QMD (cm)		Height (m)	
Treatment	Oaks	Non-oaks	Oaks	Non-oaks	Oaks	Non-oaks	Oaks	Non-oaks
Control	448 ± 36	400 ± 39	327 ± 36	250 ± 39	19.3 ± 0.8	9.5 ± 0.8	12.4 ± 0.5	8.6 ± 0.7
Diagonal	355 ± 33	244 ± 16	237 ± 33	84 ± 16	19.5 ± 0.8	6.7 ± 0.9	13.4 ± 0.7	6.0 ± 0.8
Patch	288 ± 33	292 ± 17	201 ± 33	80 ± 17	16.5 ± 1.5	9.0 ± 1.2	10.3 ± 0.9	7.2 ± 0.8
Row	369 ± 27	440 ± 52	219 ± 27	243 ± 52	16.1 ± 1.5	9.9 ± 1.3	16.1 ± 1.5	8.5 ± 0.9

DISCUSSION

Pre-treatment tree densities on control stands and row thinning stands were greater than on these stands post-treatment (tables 1 and 3). We attribute this apparent discrepancy to differences in minimum stem diameter included in assessments (≥2 inch dbh vs. ≥6 cm dbh), but differences in plot size (0.05 acre vs. 0.05 ha) and tree mortality between assessments may have also influenced live tree density.

The greater age of plantings at Yazoo NWR likely contributed to taller trees with larger diameters and greater basal area per ha at this location. Felling of larger trees likely contributed to greater volume of coarse woody debris post-treatment at Yazoo NWR. Apparent increases in understory vegetation at Cache River NWR may be a response to reduced canopy closure at Cache River NWR (~69 percent) compared to 72 percent and 79 percent canopy closure at Yazoo and Grand Cote NWR, respectively.

Despite a prescription to remove the most canopy using diagonal corridor treatments, less post-treatment canopy was found in variable patch cut treatments (56 percent) than in diagonal corridor treatments (68 percent). Relative to canopy closure in post-treatment

Table 6—Number of birds detected during breeding bird point counts during 2012 – 2014 post-treatment assessment of stand treatments to enhance wildlife habitat during 2010. Prescribed treatments included untreated control, diagonal corridor thinning, variable patch clearcuts, and single (1 of 4) row thinning

Species	Control (n = 19)	Diagonal (n = 20)	Patch (n = 18)	Row (n = 20)
Acadian flycatcher (*Empidonax virescens*)	11	1	4	8
American crow (*Corvus brachyrhynchos*)	3	7	6	9
Baltimore oriole (*Icterus galbula*)	0	3	1	5
Blue-gray gnatcatcher (*Polioptila caerulea*)	1	1	6	3
Blue jay (*Cyanocitta cristata*)	2	3	4	1
Brown-headed cowbird (Molothrus ater)	8	13	15	6
Brown thrasher (*Toxostoma rufum*)	0	1	3	0
Carolina chickadee (*Poecile carolinensi*s)	4	4	3	0
Carolina wren (*Thryothorus ludovicianus*)	4	14	9	14
Common grackle (*Quiscalus quiscula*)	0	14	0	5
Downy woodpecker (*Picoides pubescens*)	3	3	4	4
Eastern towhee (*Pipilo erythrophthalmus*)	9	1	5	2
Eastern wood pewee (*Contopus virens*)	1	2	1	2
Great-crested flycatcher (*Myiarchus crinitus*)	1	1	2	3
Indigo Bunting (*Passerina cyanea*)	9	24	25	16
Kentucky warbler (*Geothlypis formosa*)	2	3	2	2
Mourning dove (*Zenaida macroura*)	3	11	10	7
Northern cardinal (*Cardinalis cardinalis*)	60	65	46	66
Prothonotary warbler (*Protonotaria citrea*)	8	4	3	6
Red-bellied woodpecker (*Melanerpes carolinus*)	4	4	3	7
Red-eyed vireo (*Vireo olivaceus*)	4	0	0	1
Ruby-throated hummingbird (*Archilochus colubris*)	1	2	1	1
Summer tanager (*Piranga rubra*)	3	0	1	7
Tufted Titmouse (*Baeolophus bicolor*)	17	7	8	5
White-eyed Vireo (*Vireo griseus*)	11	19	21	14
Wood thrush (*Hylocichla mustelina*)	1	2	4	5
Yellow-billed Cuckoo (*Coccyzus americanus*)	25	28	24	28
Yellow-breasted Chat (*Icteria virens*)	13	67	55	11
Birds detected / count	11.1 ± 1.5	15.4 ± 2.2	15.1 ± 1.9	12.2 ± 1.6
Species detected / count	7.0 ± 0.6	7.8 ± 0.5	8.7 ± 0.6	8.0 ± 0.7

Species with ≤2 detections included: blue grosbeak (*Passerina caerulea*), chimney swift (*Chaetura pelagica*), common yellowthroat (*Geothlypis trichas*), fish crow (*Corvus ossifragus*), great-horned owl (*Bubo virginianus*), hairy woodpecker (*Leuconotopicus villosus*), northern mockingbird (*Mimus polyglottos*), northern parula (*Setophaga americana*), pileated woodpecker (*Hylatomus pileatus*), prairie warbler (*Setophaga discolor*), red-shouldered hawk (*Buteo lineatus*), Swainson's warbler (*Limnothlypis swainsonii*), white-breasted nuthatch (*Sitta carolinensis*), and yellow-throated vireo (*Vireo flavifrons*).

control stands, application of treatments were conservative compared to prescription intentions for row and diagonal corridor thinning: Canopy cover was reduced ~13 percent by row thinning (prescription of 25 percent reduction), ~24 percent by diagonal corridor thinning (prescription of ~56 percent reduction), and ~36 percent by variable patch clearcuts thinning (prescription of 30-40 percent reduction). Even so, several desired changes in forest structure likely to promote desired forest conditions for wildlife were associated with diagonal corridor thinning and variable patch clearcut treatments including: Greater canopy heterogeneity as denoted by increased SE of canopy closure, reduced canopy cover and basal area, increased understory vegetation, and increased volume of coarse woody debris. We note however, increased woody debris was a byproduct of these non-commercial treatments where no biomass was removed from stands.

Treated stands were relatively small for evaluation of bird use of these areas. Thus, edge effects due to bird movements among stands and the surrounding landscape were likely. Even so, marked increases in the abundance of birds associated with early successional forest habitats (Thompson and DeGraaf 2001) were noted within stands subjected to variable patch clearcuts and diagonal corridor thinning. Lack of an increase of bird species that are typically associated with shrub-scrub habitats within stands subjected to row thinning suggests that openings within this treatment were insufficient in area to create habitat conditions that attract colonization by these bird species. Indeed, 4 years after treatment (2014), our subjective visual observation of the canopy openings created by row thinning suggests that these single row openings are being captured by lateral crown growth of residual canopy trees such that little understory vegetation is being stimulated.

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LITERATURE CITED

Anderson, M.J. 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology. 26: 32-46.

Chianucci, F.; Cutini, A. 2012. Digital hemispherical photography for estimating forest canopy properties: current controversies and opportunities. Journal of Biogeosciences and Forestry. 5: 290-295. http://www.sisef.it/iforest/contents/?id=ifor0775-005. [Date accessed: 25 August 2015].

Dufréne, M.; Legendre, P. 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecological Monographs. 67: 345-366.

Farnsworth, G.L.; Nichols, J.D.; Sauer, J.R. [and others]. 2005. Statistical approaches to the analysis of point count data: A little extra information can go a long way. In: Ralph, C.J.; Rich, T.D., eds. Bird conservation implementation and integration in the Americas: proceedings of the 3rd international partners in flight conference. Gen. Tech. Rep. PSW-GTR-191. Albany, CA: U.S. Department of Agriculture Forest Service, Pacific Southwest Research Station: 736–743.

Johnson, P.S. 2004. Thinking about oak forests as responsive ecosystems. In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 13-18.

LMVJV Forest Resource Conservation Working Group. 2007. Restoration, management, and monitoring of forest resources in the Mississippi Alluvial Valley: recommendations for enhancing wildlife habitat. Wilson, R.; Ribbeck, K; King, S.; Twedt, D., eds. Vicksburg, MS: Lower Mississippi Valley Joint Venture. 88 p.

Lorimer, C.G. 1989. Relative effects of small and large disturbances on temperate hardwood forest structure. Ecology. 70: 565-567.

Meadows, J.S. 1996. Thinning guidelines for southern bottomland hardwood forests. In: Flynn, K.M., ed. Proceedings of the southern forested wetlands ecology and management conference. Clemson, SC: Consortium for Research on Southern Forested Wetlands: 98-101.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. New York: John Wiley & Sons. 520 p.

Plummer, J.F.; Keyes, C.R.; Varner III, J.M. 2012. Early-stage thinning for the restoration of young redwood – Douglas-fir forests in northern coastal California, USA. ISRN Ecology: Article 725827:1-9. http://dx.doi.org/10.5402/2012/725827. [Date accessed: August 25, 2015].

Thompson III, F.R.; DeGraaf, R.M. 2001. Conservation approaches for woody, early successional communities in the eastern United States. Wildlife Society Bulletin. 29: 483-494.

Twedt, D.J. 2012. Wildlife forestry. In Okia, C.A., ed. Global perspectives on sustainable forest management, ISBN: 978-953-51-0569-5. Rijeka, Croatia: INTECH: 161-190. http://www.intechopen.com/books/global-perspectives-on-sustainable-forest-management/wildlife-forestry. [Date accessed: August 25, 2015].

Twedt, D.J.; Somershoe, S.G. 2009. Bird response to prescribed silvicultural treatments in bottomland hardwood forests. Journal of Wildlife Management. 73: 1140-1150.

Wigley, T.B.; Roberts, T.H. 1994. A review of wildlife changes in southern bottomland hardwoods due to forest management practices. Wetlands. 14: 41-48.

SILVICULTURE FOR A DECLINING SPECIES, CERULEAN WARBLER: 10-YEAR RESULTS OF A PILOT STUDY IN THE MISSISSIPPI ALLUVIAL VALLEY

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Abstract—We report on the current status of a long-term study of Cerulean Warbler (*Setophaga cerulea*) response to silviculture on a 58-ha tract in Desha County, Arkansas. The work involved a 10 year pre-manipulation monitoring of the birds on the tract, followed by implementation of a split plot comparison of alternative treatments, each applied to a randomly selected half of the original plot. We present results of the alternative treatments on warbler species occurrence, distribution and standing crop of canopy and subcanopy trees, and composition and abundance of advanced regeneration resulting from the manipulation applied in 2002-2004. Basal areas in the area managed by the Cerulean Warbler prescription were higher than those in area managed by a standard prescription. Advanced regeneration did not meet company standards for favored species on nearly 2/3 of the company treated area and almost 90 percent of the area treated with the Cerulean Warbler prescription. Interpretation of these values was complicated by lingering effects of a devastating 1994 ice storm. This necessarily unreplicated study serves as a pilot for later evaluation of a Cerulean Warbler silvicultural treatment. It is a cautionary tale because continuing decline of the Cerulean Warbler population introduces an unknown amount of uncertainty into interpretation of the response of the birds to the experimental manipulation; no Cerulean Warblers were recorded on the plot in the most recent survey. The study provides a basis to consider trade-offs in management for a resource that may be dominated by extra-ownership effects beyond the manager's control or manipulation.

INTRODUCTION

Silviculture for Songbirds, the practice of managing forests to produce specific habitats for bird communities of interest, is usually practiced as a process in which a prescription designed to produce specific forest products is applied to stands of interest and the response of birds to the habitats resulting from the manipulations is monitored as a design output. The process is a well-respected one, a reasonable basis for adaptive management applications (Powell and others 2000), and one that works especially well for targeting community or guild responses. In the case of particularly vulnerable species, of which the Cerulean Warbler (*Setophaga cerulea*) is certainly one (Buehler and others 2013, Hamel 2000, US Fish and Wildlife Service 2006), the importance of silviculture to the continued existence of the species is well-recognized (Boves and others 2013, Buehler and others 2008, Hamel 2005) among other actions (Buehler and others 2006). For such vulnerable species as the Cerulean Warbler, a management approach directly tailored to the biology of the species may be more useful than one based simply on monitoring avian response to standard silvicultural prescriptions.

The current report is the third in a series depicting an experimental silviculture prescription designed to produce habitat conditions identified from observations of the behavioral ecology of Cerulean Warbler (Hamel and others 2006, Hamel and others 2010). Our objective is to evaluate the response of the birds to the treatment over a 10-year entry cycle, and to compare the standing crop of trees and advanced regeneration at the end of the cycle. Our work has been a necessarily unreplicated study as no other suitable study sites were available in the physiographic province in which we work.

METHODS

Study Area Location and Recent History

The study area is a 58-ha (143 acres) portion of a larger 130-ha (320 acres) management unit on properties managed by Anderson Tully Co. (ATCO), a division of The Forestland Group, in Desha County., Arkansas (33° 44' 42" N, 91° 9' 30" W, fig. 1). Primary management of

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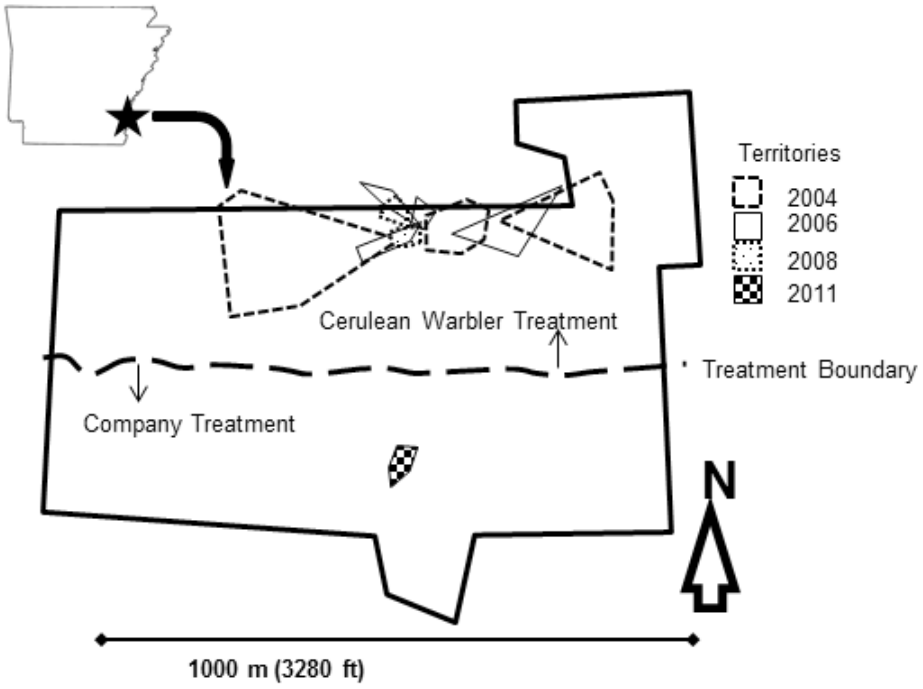


Figure 1—Cerulean Warbler study site in Desha County, Arkansas, USA.

the unit is for large sawtimber products of bottomland hardwoods. After the harvest entry into the unit in 1991, which was conducted according to standard company prescription, the study area was removed from the management portfolio of the company to enable study of behavioral ecology of Cerulean Warbler on it. As such, the study area was not available for company salvage logging activities subsequent to a devastating 1994 ice storm that affected much of a 20 county area in Arkansas and adjacent Mississippi. The storm created extensive damage to the forest canopy of the study area with heavier damage to the midstory than overstory trees of the stand (Christopher A. Woodson, U.S. Fish and Wildlife Service, 101 Park DeVille Drive, Suite B, Columbia, MO 65203, unpublished analyses).

Forest vegetation of the study area is bottomland hardwoods of the riverfront hardwoods subtype (Hodges 1997, Johnson 1973), characterized by American sycamore (*Platanus occidentalis*), eastern cottonwood (*Populus deltoides*), American elm (*Ulmus americana*), sweet pecan (*Carya illinoiensis*), green ash (*Fraxinus pennsylvanica*). Soils in the study area are of Sharkey-Commerce-Coushatta association (Gill and others 1972), which are young silt loam soils in new meanders of the Mississippi River. The study area and adjacent company lands in Desha County are within the Mississippi River batture lands.

The study area, one of three in a larger study of Cerulean Warbler behavioral ecology (Hamel 2005), is near the extreme southern end of the breeding range of the species. The others are in Shelby and Lauderdale Counties. in Tennessee.

Treatments

For a complete description of treatments and site conditions, please see Hamel and others (2006, Hamel and others 2010). Treatments applied in the study, defined in Hamel and others (2006), are abstracted as follows.

Company Treatment—This partial cutting prescription involved elements of improvement cutting, thinning, and regeneration cutting. It involved harvesting overstory trees to reduce mortality, improve species composition and spacing, and increase growth of the residual stand. It further involved cutting midstory trees to remove poorly formed shade tolerant species in order to release advanced regeneration and encourage the establishment and growth of additional shade intolerant regeneration of desirable species.

Cerulean Warbler Treatment—This partial cutting prescription was a modification of the standard prescription, involving elements of improvement cutting, thinning, and regeneration cutting. The prescription differed from the standard prescription in that fewer trees were removed from the shade tolerant midstory.

Avian Sampling

We conducted territory mapping of Cerulean Warbler, Northern Parula (*Setophaga americana*), American Redstart (*S. ruticilla*), Yellow-throated Warbler (*S. dominica*), Hooded Warbler (*S. citrina*), Swainson's Warbler (*Limnothlypis swainsonii*), as well as daily tallies of all bird species observed on the study area during breeding season visits. Sample sizes by year were as follows: 2004 (5 samples), 2005 (1), 2006 (8), 2007 (1), 2008 (4), 2011 (3), 2012 (8), 2013 (8), 2014 (1). Anticipated fieldwork in 2010, as well as half of that in 2011, and portions in other years could not be conducted because flooding precluded access to the study site. Sampling consisted of walking surveys by observers (authors Smith or Hamel) on predetermined routes through the study area, during which encountered individuals were marked on maps. Territory maps were prepared from the field maps; a territory was assigned to the area where a male bird was recorded on at least three visits in a year. Results from these territory maps depict the composite response of the Cerulean Warbler to the treatments. In addition to the mapped locations, associated observations of behavior of the birds were used to interpret Cerulean Warbler use of the treatments. The composite map of the response of the birds to the treatments (fig. 1) was evaluated by visual inspection.

Vegetation Sampling

We measured the forest vegetation of the study area after the 2013 growing season at two scales. Measurements of canopy vegetation were made at regularly spaced intersections of a 50x50m grid of the study area (N = 260). Grid intersections on the border between the two treatments were assigned alternately to Company Treatment or Cerulean Warbler Treatment systematically from a random starting point. At each such intersection, a variable-radius sample was taken using a 6.9-m^2/ha (30 ft^2 per acre) basal area factor sampling device (JIM-GEM® Cruz-All, Forestry Suppliers, Inc., Jackson, MS). Each apparent in-tree was identified to species, and its crown class dominant, co-dominant, intermediate, suppressed), diameter at 1.4 m (4.5 ft) above ground level, height using a hypsometer (Haglöf, Långsele, Sweden) , presence of vines in the canopy, and distance and azimuth from the grid point were recorded. After fieldwork was completed, the record of diameters was screened to remove trees from the sample whose center was farther from the grid point than the limiting distance of the basal area factor for the measured diameter. The sample measurements were summarized into density and basal area data for individual species, for different shade tolerance classes (Burns and Honkala 1990), and by ATCO Market Classes (table 1). Resulting summaries for the two treatments were compared using t-tests, with significance accepted at experiment-wide Bonferroni-corrected $\alpha = 0.05$.

Advanced regeneration developed or promoted as a result of the treatments in this study was measured at 0.0004 ha (0.001 acre, "milacre") fixed radius plots. All stems less than 10.2 cm (4 in) diameter in these plots were tallied to species and recorded in one of two categories, those (a) shorter and (b) taller than 1.4 m (4.5 ft). These plots were located as follows: one plot was superimposed on each grid intersection and the other two plots were sited 12.5 m (41 ft) away from that center plot. In grid locations within the two treatments these plots were located east and west of the intersection. For gridpoints on the border of the treatments, the two plots were placed north and south of the intersection. In addition to these tallies, stems 10-15 cm (4-6 in), and 15-25 cm (6-10 in) dbh were tallied in 0.008 ha (0.02 acre) fixed radius plots centered on the grid intersections.

A variety of locations and plot sizes was necessary to characterize the advanced regeneration in three ways. The first of these ways dealt only with the stems tallied in the milacre plots, and involved stem densities summarized to species, to shade tolerance category, and to ATCO Market Class (table 1). The second set of characterizations involved calculation of advanced regeneration score (Johnson 1980) for each grid intersection. The regeneration score was determined via averaging the tally of stems measured on the three milacre plots and stems 10-25 cm were summarized from the single 0.008 ha plot at the intersection. These tallies all were standardized to stems per 0.01 acre plot required by the Johnson (1980) method, and scaled by the factors published in the Johnson (1980) protocol into a composite regeneration-points-per-plot basis. The third set of characterizations was applied to tally of stems as in the Johnson (1980) method, applied only to the ATCO Market Class A species. A composite density estimation was calculated from these data as follows,

$$\text{Johnson Points} = (\text{Stems} < 4.5 \text{ ft tall/ac})/6 + (\text{stems} > 4.5 \text{ ft tall/ac})/2 + (\text{stems} > 4\text{" dbh/ac}). \quad (1)$$

Statistical Treatment

Regeneration data compiled in these categories were analyzed as follows. The milacre data were analyzed with SAS/STAT software, Version 9.2 of the SAS System for Windows using Proc GLIMMX generalized linear mixed models for Poisson distributed data (© SAS Institute Inc. 2009) and treatments tested

Table 1—Basal area of canopy trees recorded on Cerulean Warbler study area, Desha Co., AR, after 2013 growing season. Asterisks indicate species whose abundance differed between the treatments at P = 0.05 after Bonferroni correction for simultaneous multiple species comparisons (ATCO – Company Treatment, CERW – Cerulean Warbler Treatment)

Tree Species	Shade tolerance	ATCO Market Class	Treatment		
			ATCO		CERW
			N = 123		N = 137
			mean basal area ± s.e., m^2/ha (ft^2/ac)		
boxelder, *Acer negundo*	Tolerant	C	3.1 ± 0.5		3.6 ± 0.5
			(13.5 ± 2.2)		(15.7 ± 2.2)
red maple, *Acer rubrum*	Moderate	B	0.1 ± 0.1		0.1 ± 0.1
			(0.4 ± 0.4)		(0.4 ± 0.4)
sweet pecan, *Carya illinoiensis*	Intolerant	A	0.9 ± 0.2		0.9 ± 0.2
			(3.9 ± 0.9)		(3.9 ± 0.9)
sugarberry, *Celtis laevigata*	Moderate	B	5.4 ± 0.6	**	11.1 ± 0.8
			(23.5 ± 2.6)	**	(48.3 ± 3.5)
dogwood, *Cornus drummondi*	—	D	0.1 ± 0.1		-
			(0.4 ± 0.4)		-
persimmon, *Diospyros virginiana*	Tolerant	B	0.1 ± 0.1		-
			(0.4 ± 0.4)		-
green ash, *Fraxinus pennsylvanica*	Moderate	A	2.5 ± 0.4		4 ± 0.6
			(10.9 ± 1.7)		(17.4 ± 2.6)
sweetgum, *Liquidambar styraciflua*	Intolerant	A	2 ± 0.4	**	0.2 ± 0.1
			(8.7 ± 1.7)	**	(0.9 ± 0.4)
sycamore, *Platanus occidentalis*	Intolerant	A	1.6 ± 0.4		1.9 ± 0.3
			(7 ± 1.7)		(8.3 ± 1.3)
cottonwood, *Populus deltoides*	Intolerant	A	0.2 ± 0.1		0.1 ± 0.1
			(0.9 ± 0.4)		(0.4 ± 0.4)
overcup oak, *Quercus lyrata*	Intolerant	B	0.1 ± 0.1		-
			(0.4 ± 0.4)		-
Nuttall oak, *Quercus nuttallii*	Intolerant	A	0.1 ± 0.1		-
			(0.4 ± 0.4)		-
baldcypress, *Taxodium distichum*	Moderate	C	2 ± 0.5		4.1 ± 0.8
			(8.7 ± 2.2)		(17.9 ± 3.5)
American elm, *Ulmus americana*	Moderate	B	1.7 ± 0.3		2.4 ± 0.4
			(7.4 ± 1.3)		(10.5 ± 1.7)
Total			**19.7 ± 1.1**	**	**29.1 ± 1.1**
			(371 ± 5)	**	**(479 ± 5)**

for significance at α = 0.05. Data from the Johnson (1980) characterization as well as the composite ATCO Market Class A trees were modeled with the method of Krishnamoorthy and others (2011) to assess of the adequacy of advanced regeneration.

RESULTS

Avian Sampling

Surveys were conducted frequently enough in 2004, 2006, 2008, 2011, 2012, and 2013 to map territories on and adjacent to the study area. Female Cerulean Warblers were associated with the males in the surveys in 2004, 2006, and 2008, suggesting that breeding occurred; however, no nests were discovered. In all years save 2011, territories were located in the Cerulean Warbler Treatment area (fig. 1). During the survey in 2011, cut short by flooding, one singing male was found three times during April in the Company Treatment area. In 2012-2014, no Cerulean Warblers were found in the study area.

Vegetation Sampling

Canopy Vegetation—Mean canopy basal area ten years after treatment was significantly greater ($t = 6$, 258 df, $P < 0.0001$; fig. 2) on the Cerulean Warbler Treatment area (29 ± 1.1 m^2/ha; 479 ± 5 ft^2per acre, n=137 plots; quadratic mean diameter 27.7 cm, 10.9 in) than that on the Company Treatment area (19.7 ± 1.1 m^2/ha; 371 ± 5 ft^2 per acre, n=123 plots; quadratic mean diameter 26.0 cm, 10.2 in). Individuals of 14 tree species were recorded in the canopy (table 1). Basal area of two of these species differed significantly at the comparison-wide α = 0.05. Sugarberry (*Celtis laevigata*) was more abundant on the Cerulean Warbler Treatment ($t = 5.4$, df= 258, $P < 0.0001$); and sweetgum (*Liquidambar styraciflua*) was more abundant on the Company Treatment ($t = 4.9$, df= 258, $P < 0.0001$). These two species were responsible for the significant differences between treatments by shade tolerance class as well (fig. 3); sweetgum is shade intolerant, while sugarberry is moderately tolerant of shade. The treatments were different in basal area by ATCO Market Class only for Class B species, also because of the large difference in sugarberry basal area (fig. 4).

Advanced Regeneration—Ten years after the application of the treatments, estimated total advanced regeneration measured on milacre plots was abundant; Company Treatment 3537 ± 155 stems/ha (1432 ± 62 stems per acre), n = 368; Cerulean Warbler Treatment 3507 ± 148 stems/ha (1420 ± 59 stems per acre), n = 412. The treatments did not differ significantly by generalized linear mixed model with Poisson distributed data ($F_{1, 778} = 0.02$, $P = 0.89$). Seedlings and saplings of 14 tree species were identified on these plots (table 2), in which mean abundance of three species, boxelder (*Acer negundo*; $t = 6.2$, df=778, $P < 0.0001$), sugarberry ($t = 7.3$, df=778, $P < 0.0001$), and sweetgum ($t = 6.3$, df=778, $P < 0.0001$) differed between the treatments by

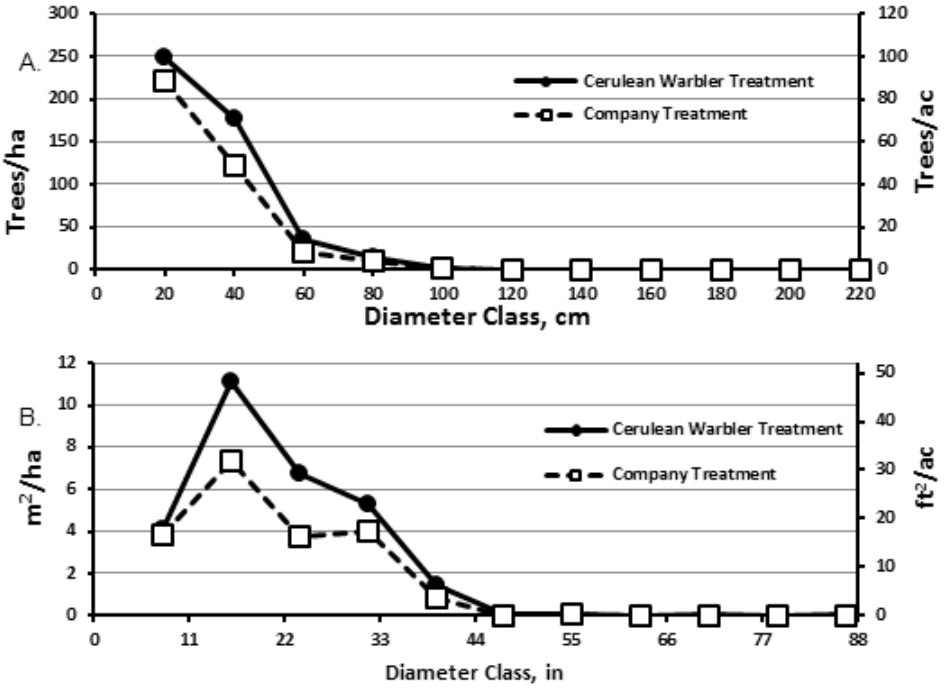


Figure 2—Diameter class distributions of tree density (upper) and basal area (lower) on the Cerulean Warbler study site measured after 2013 growing season.

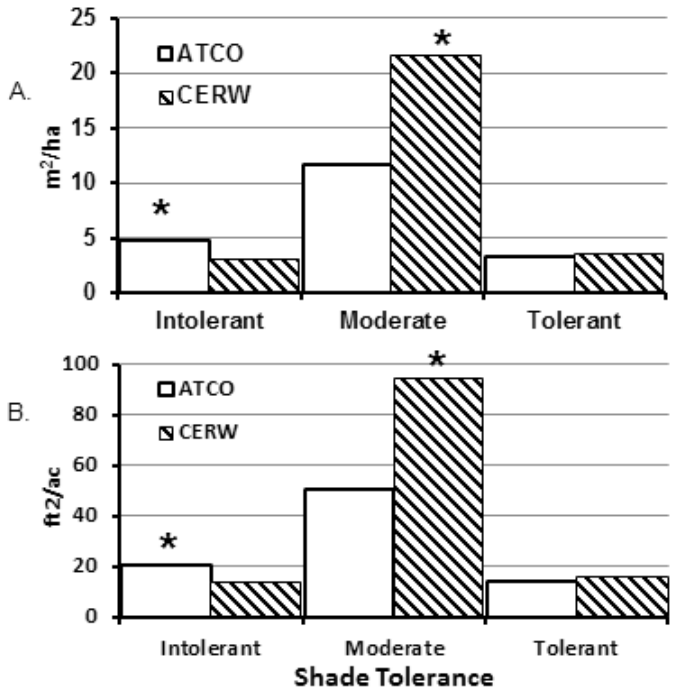


Figure 3—Forest canopy basal area distribution by shade tolerance classes, Cerulean Warbler study area, Desha Co., AR, after 2013 growing season. A. Basal area expressed in m^2/ha; B. Basal area expressed in ft^2/ac. Asterisks indicate that treatments differ at P = 0.05. ATCO – Company Treatment; CERW – Cerulean Warbler Treatment.

t-test after Bonferroni correction to comparison-wide α = 0.05.

The distribution of advanced regeneration between the treatments differed for each shade tolerance class modeled by the generalized linear mixed models approach; the Company Treatment produced more shade intolerants [709 ± 72 stems/ha (287 ± 29 stems per acre), n = 368; $F_{1, 760} = 21.16$, $P < 0.0001$] and more shade tolerants [1383 ± 99 stems/ha (560 ± 40 stems per acre), n=368; $F_{1, 760} = 37.23$, $P < 0.0001$] than the Cerulean Warbler Treatment [intolerants 331 ± 44 stems/ha (134 ± 18 stems per acre), tolerants (672 ± 62 stems/ha (272 ± 25 stems per acre), n=412]. The Cerulean Warbler Treatment produced more advanced regeneration of moderate shade tolerance [2507 ± 124 stems/ha (1015 ± 50 stems per acre), n= 412; $F_{1, 760} = 36.3$, $P < 0.0001$] than did the Company treatment [1509 ± 104 stems/ha (611 ± 42 stems per acre), n = 368]. Advanced regeneration of the desired species in the ATCO Market Class A was more abundant in the Company Treatment [855 ± 77 stems/ha (346 ± 31 stems per acre), n = 368; $F_{1, 760} = 18.42$, $P < 0.0001$] than in the Cerulean Warbler Treatment [454 ± 52 stems/ha (184 ± 21 stems per acre), n = 412].

In comparison to the tolerance intervals modeled from milacre samples using methods in Krishnamoorthy and others (2011) for ATCO Market Class A species, the Company Treatment produced at least 500 stems/ha (200 stems per acre) throughout the treatment, while barely 10 percent of the Cerulean Warbler Treatment produced the desired abundance of advanced regeneration (fig. 5). A further comparison using the same modeling approach to the composite of data from milacre and 0.008 ha plots yielded a similar result (fig. 6) relative to a 250 stems/ha (100 stems per acre) criterion density of Market Class A stems.

Results of the examination of the advanced regeneration on the two treatments using the Johnson (1980) points system indicated that 72 percent of the plots on the Company Treatment, and 56 percent of the plots on the Cerulean Warbler Treatment, contained sufficient advanced regeneration to be considered stocked. By comparison, 21 percent of plots on the Company Treatment, and 8 percent of plots on the Cerulean Warbler Treatment, were stocked with sufficient advanced regeneration of desirable Market Class A species.

DISCUSSION

Several reservations must be raised about this study. It was unreplicated, so uncontrolled differences between the treatments may have produced the observed

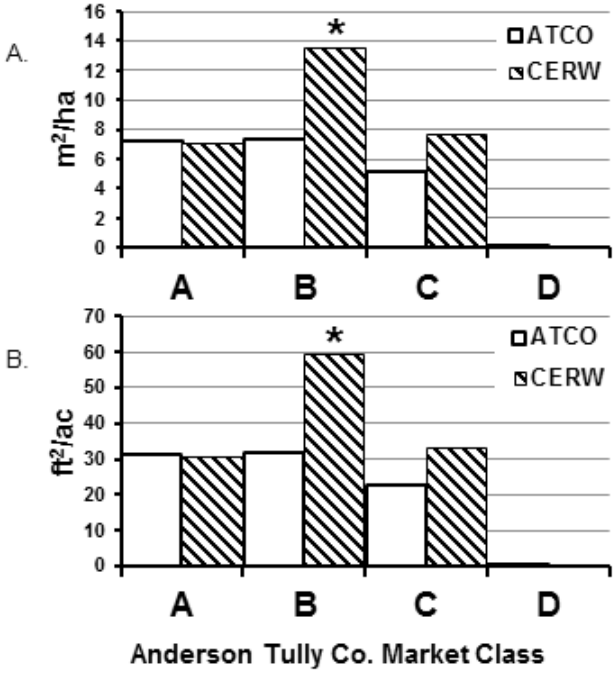


Figure 4—Forest canopy basal area distribution by Anderson Tully Market Class, Cerulean Warbler study area, Desha Co., AR, after 2013 growing season. A. Basal area expressed in m^2/ha; B. Basal area expressed in ft^2/ac. Asterisks indicate treatments differ at P = 0.05. ATCO – Company Treatment; CERW – Cerulean Warbler Treatment.

Table 2—Advanced regeneration recorded on Cerulean Warbler study area, Desha Co., AR, after 2013 growing season. Asterisks indicate species whose abundance differed between the treatments at P = 0.05 after Bonferroni correction for simultaneous multiple species comparisons (ATCO – Company Treatment, CERW – Cerulean Warbler Treatment)

Tree Species	Shade tolerance	ATCO Market Class	Treatment		
			ATCO		CERW
			N = 368		N = 412
			mean stems/ha, (stems/ac)		
boxelder, *Acer negundo*	Tolerant	C	1235 (500)	**	576 (233)
red maple, *Acer rubrum*	Moderate	B	7 (2.7)		12 (5)
sweet pecan, *Carya illinoiensis*	Intolerant	A	148 (60)		156 (63)
sugarberry, *Celtis laevigata*	Moderate	B	846 (342)	**	1805 (731)
persimmon, *Diospyros virginiana*	Tolerant	B	60 (24)		30 (12)
green ash, *Fraxinus pennsylvanica*	Moderate	A	181 (73)		144 (58)
honeylocust, *Gleditsia triacanthos*	Intolerant	D	-		6 (2)
sweetgum, *Liquidambar styraciflua*	Intolerant	A	271 (111)	**	12 (5)
red mulberry, *Morus rubra*	Tolerant	C	40 (16)		66 (27)
sycamore, *Platanus occidentalis*	Intolerant	A	161 (65)		138 (56)
water oak, *Quercus nigra*	Intolerant	B	47 (19)		12 (5)
Nuttall oak, *Quercus nuttallii*	Intolerant	A	74 (30)		6 (2)
baldcypress, *Taxodium distichum*	Moderate	C	34 (14)		30 (12)
American elm, *Ulmus americana*	Moderate	B	430 (174)		516 209)
Total			**3537 (1432)**		**3507 (1420)**

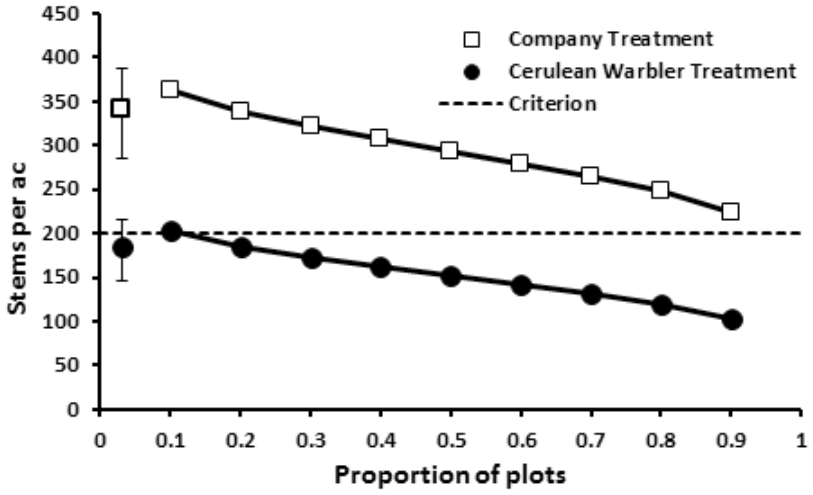


Figure 5—Proportion of milacre advanced regeneration measurement plots expected to meet 200 stems/ac (500 stems/ha) criterion abundance of Anderson Tully Co. Market Class A species in treatments on Cerulean Warbler study area in Desha Co., AR, measured during 2013 growing season (Company Treatment n = 368, Cerulean Warbler Treatment n = 412). Connected symbols reflect model estimates (Krishnamoorthy and others 2011) based upon actual values, whose means ± 95 percent confidence intervals are shown in the separate symbols

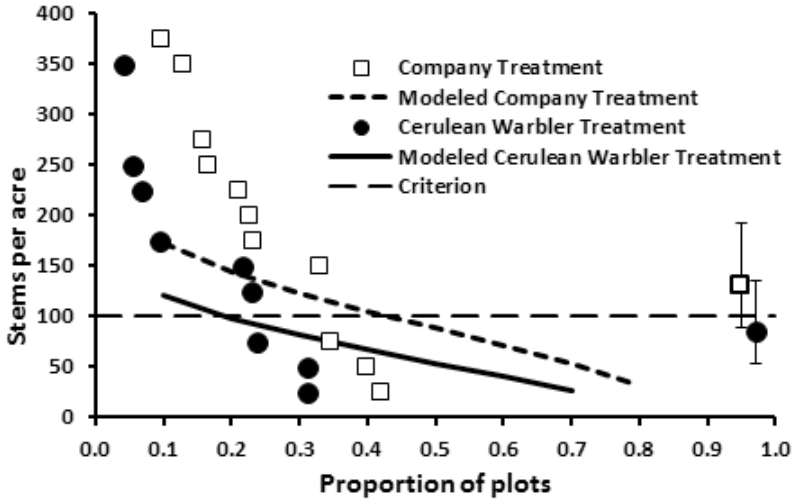


Figure 6—Proportion of 0.02 ac advanced regeneration composite measurement plots expected to meet 100 stems/ac (247 stems/ha) criterion abundance of Anderson Tully Co. Market Class A species in treatments on Cerulean Warbler study area in Desha Co., AR, measured during 2013 growing season (Company Treatment n = 123, Cerulean Warbler Treatment n = 137). Symbols are actual measurements. Lines reflect model estimates (Krishnamoorthy and others 2011) based upon actual values, whose means ± 95 percent confidence intervals are shown in the separate symbols on the right edge of the graph. Note that only 19 percent of plots in the Cerulean Warbler Treatment area and 45 percent of plots in the Company Treatment area are expected to meet criterion abundance, despite the fact that the mean values do not differ significantly from criterion.

results. Also, global Cerulean Warbler populations declined during the study, suggesting that factors external to the study may have affected response by the birds to an unknown degree. The duration of the study itself exposed the plot to effects of a severe ice-storm that impeded management actions carried out on other affected company lands. Severe flooding prevented visitation to the area in more than one year of the study.

Having raised these cautions, several encouraging results were observed. First, as Cerulean Warbler pairs were present on the Cerulean Warbler Treatment area in three sampling sessions, 2004-2008, the prescription designed for the Warbler is provisionally successful. Our work lasted longer than that of Boves and others (2013) and may have reflected some of the source-sink and ecological trap effects they postulated. The concentration of territories in the same area, and the positive association of territory size with distance from a common center point during the study begs further study.

Forest canopy and advanced regeneration resulting from the treatments were consistent with the prescriptions applied, and provide some indication of the trade-offs required in this habitat production between suitable habitat for the Cerulean Warbler and maintenance of sawtimber production into the future on the same lands.

CONCLUSIONS

1. The treatments produced forest stand conditions consistent with the prescriptions applied.
2. The Cerulean Warbler Treatment intended to produce habitat for Cerulean Warbler was used consistently by the birds for a portion of the study; the Company Treatment was used only incidentally by the birds.
3. This unreplicated pilot study offers guidance for more substantive later work in Mississippi River Alluvial Valley forests.
4. Behavioral ecology of Cerulean Warbler merits continued examination.

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LITERATURE CITED

Boves, T.J.; Buehler, D.A.; Sheehan, J. [and others]. 2013. Emulating natural disturbances for declining late-successional species: a case study of the consequences for Cerulean Warblers (*Setophaga cerulea*). PLoS ONE. 8(1): e52107. http://journals.plos.org/plosone/article?id=10.1371/journal.pone.0052107. [Date accessed: April 28, 2015].

Buehler, D.A.; Giocomo, J.J.; Jones, J. [and others]. 2008. Cerulean Warbler reproduction, survival, and models of population decline. Journal of Wildlife Management. 72: 646-653.

Buehler, D.A.; Hamel, P.B.; Boves, T.J. 2013. Cerulean Warbler *Setophaga cerulea*. Revised. In: Birds of North America online. Poole, A., ed. Ithaca, NY: Cornell Lab of Ornithology. http://bna.birds.cornell.edu/bna/species/511/articles/introduction. [Date accessed: April 28, 2015].

Buehler, D.A.; Welton, M.J.; Beachy, T.A. 2006. Predicting Cerulean Warbler habitat use in the Cumberland Mountains of Tennessee. Journal of Wildlife Management. 70: 1763-1769.

Burns, R.M.; Honkala, B.H., tech. cords. 1990. Silvics of North America: Vol. 1. Conifers; Vol. 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture. 877 p.

Gill, H.V.; Larance, F.C.; Cloutier, H.; Long, H.W. 1972. Soil survey of Desha County, Arkansas. Washington, DC: U. S. Department of Agriculture Soil Conservation Service. In cooperation with: Arkansas Agricultural Experiment Station. 50 p. http://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx. [Date accessed: April 24, 2015].

Hamel, P.B. 2000. Cerulean Warbler status assessment. St. Paul, MN: U.S. Department of Interior, Fish and Wildlife Service. 141 p. http://www.fws.gov/midwest/es/soc/birds/cerw/pdf/cerw-sa.pdf. [Date accessed: April 28, 2015].

Hamel, P.B. 2005. Suggestions for a silvicultural prescription for Cerulean Warblers in the Lower Mississippi Alluvial Valley. In: Ralph, C.J.; Rich, T.D., eds. Bird conservation implementation and integration in the Americas: proceedings of the third international partners in flight conference, 20-24 March 2002, Asilomar Conference Grounds, CA. Gen. Tech Rep. PSW 191. Albany, CA: U.S. Department of Agriculture Forest Service, Pacific Southwest Research Station: 567-575.

Hamel, P.B.; Staten, M.; Wishard, R. 2006. Initial Cerulean Warbler (*Dendroica cerulea*) response to experimental silvicultural manipulations, Desha County, Arkansas. In: Connor, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS 92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 3-9.

Hamel, P.B.; Staten, M.; Wishard, R.; Smith, C.G., III. 2010. Cerulean Warbler response to silvicultural manipulations on managed forestland in Desha County, Arkansas, third year results. In: Stanturf, J.A., ed. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS 121. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 475-479.

Hodges, J.D. 1997. Development and ecology of bottomland hardwood sites. Forest Ecology and Management. 90: 117-125.

Johnson, R.L. 1973. Oak-gum-cypress in the Midsouth. In: Silvicultural systems for the major forest types of the United States. Agric. Handb. 445. Washington, DC: U.S. Department of Agriculture Forest Service, Division of Timber Management Research: 98-101.

Johnson, R.L. 1980. New ideas about regeneration of hardwoods. In: Proceedings of Hardwood Regeneration Symposium. Atlanta, Ga., Jan 29, 1980. Forest Park, GA: Southeastern Lumber Manufacturers Association: 17-19.

Krishnamoorthy, K.; Yanping Xia; Fang Xie. 2011. A simple approximate procedure for constructing binomial and poisson tolerance intervals. Communications in Statistics. 40: 2243-2258.

Powell, L.A.; Lang, J.D.; Conroy, M.J.; Krementz, D.G. 2000. Effects of forest management on density, survival, and population growth of wood thrushes. Journal of Wildlife Management. 64: 11-23.

SAS Institute Inc. 2009. SAS/STAT ® 9.2 User's Guide, Second Edition. Cary, NC: SAS Institute Inc. 7869 p. http://support.sas.com/documentation/cdl/en/statug/63033/HTML/default/viewer.htm#statug_glimmix_a0000001405.htm. [Date accessed: May 13, 2015].

U.S. Department of the Interior, Fish and Wildlife Service. 2006. "Endangered and threatened wildlife and plants; 12-month finding on a petition to list the Cerulean Warbler (*Dendroica cerulea*) as threatened with critical habitat." Federal Register. 71.234: 70717-70733.

BIRD-FRIENDLY RECOMMENDATIONS FOR BOTTOMLAND FORESTS IN THE CAROLINAS: BIRDS AND PEOPLE ON COMMON GROUND

Norman Brunswig, Sharon Richardson, Matthew Johnson, and Brandon Heitkamp[1]

Abstract—Bottomland hardwood forests have suffered tremendous losses in the United States. Yet they support some of the densest breeding populations of imperiled migratory song birds in the eastern US, providing nesting habitat for 49 species, 32 of which share some Conservation Status. Traditional management for bottomland hardwood forests in the southeast has tended to rely on one of two strategies- large scale clearcuts >50 acres, or no-cutting at all. As a conservation organization, Audubon and others have encouraged landowners, land trusts and conservation groups to seek the protection of bottomland hardwood forests and prescribe a "no-cut" policy to allow the forest to mature to an old growth climax community. Bottomland hardwood forests provide habitat for 140-200 species that use different niches in the forest structure. There are priority birds that require small scale openings (Swainson's warblers), some that that can tolerate thinning within the canopy (Prothonotary warblers), and some that cannot tolerate any disturbance (Red-eyed vireo). Audubon South Carolina set out to review available research in order to identify tolerance thresholds that could be incorporated into a forest management plan or conservation easement that would allow an alternative management regime, other than cut it all or cut nothing. A set of bird-friendly best management practices were identified to address the needs of the disturbance-dependent birds while not compromising the needs of the disturbance-tolerant species, and that allow some modest harvesting revenue. When these management practices are embedded at a landscape-scale where there are large tracts of no-disturbance, then the habitat needs of the entire suite of species can be addressed. Initial demonstration sites at Silver Bluff Audubon Center and Sanctuary in Jackson, SC successfully attracted the birds of interest. One year after three small clear cuts (1, 3, 5 acres) were logged within a 90-acre stand, the habitat now has shrubby thickets and sightings of Swainson's Warblers have been confirmed.

INTRODUCTION

Project Need

Bottomland hardwood forests have suffered tremendous losses in the United States, with as much as an 80 percent reduction in area (Abernethy 1987, Gosselink 1989, Twedt and others 1999). These very forests provided much of the best habitat for many forest interior bird species whose numbers have declined dramatically.

In the South Atlantic Coastal Plain of North and South Carolina today, bottomland hardwood forests associated with large river floodplains and blackwater swamps still provide much of the best remaining habitat for many of those birds, including several of conservation concern such as Swainson's Warbler, Prothonotary Warbler, Swallow-tailed Kite, and Chimney Swift. All four of these bird species have been assigned High Priority by the US Fish and Wildlife Service in its South Atlantic Migratory Bird Initiative (SAMBI) plan. The SAMBI is the mechanism within the Atlantic Coast Joint Venture (ACJV) to integrate and coordinate bird conservation efforts to meet habitat conservation goals and to address the regional priority suite of bird species identified for the South Atlantic Coastal Plain Region of the four national bird plans (North American Waterfowl Management Plan, Partners in Flight Southeast Regional Plan, US Shorebird Conservation Plan and the North American Waterbird Conservation Plan). Only two birds dependent on bottomland forests received the Highest Priority, Ivory-billed Woodpecker and Bachman's Warbler, and one or both of them are either extinct or teetering on the edge of extinction.

In South and North Carolina, there are two primary mechanisms for protecting large tracts of forest: through acquisition and management by public (state and federal) or nonprofit entities (Audubon, Land Trusts and The Nature Conservancy), and voluntary conservation easements by private landowners subject to Forest Management Plans. For example, large blocks of protected, mature and maturing bottomland

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hardwood forests include the Savannah National Wildlife Refuge, Congaree National Park, Francis Beidler Forest, Lumber River, Waccamaw State Parks, and Roanoke River National Wildlife Refuge. There are also thousands of privately owned forest acres held under conservation easement. Collectively, these protected lands provide secure habitat for the species mentioned above and their habitat associates; however, much more bottomland habitat must be protected and compatibly managed to stabilize and increase their populations.

While managers and owners of such protected forests may choose to manage for maximum benefit of forest birds, forgoing most or all potential revenue from timber, many private bottomland forest owners cannot. Audubon South Carolina, and Audubon North Carolina and their conservation partners want to provide information to private landowners and managers describing how they can manage their forests to create and sustain habitat vital to some of the forest birds in greatest need, while also generating modest timber revenue.

Species Background – Clearcuts and Canopy Gaps

Swainson's Warbler is considered by many to be one of the most endangered songbirds in the Southeast (Savage and others 2010, Thompson 2005). In the southeastern coastal plain, this species is strongly associated with large blocks of bottomland hardwood forest, specifically dense stands of cane and palmetto and/or vine tangles which develop in natural or created canopy gaps (Wright 2002, Savage and others 2010).

Small scale canopy gaps are essential for many forest birds (Pashley and Barrow 1993, Hunter 2001, Moorman & Guynn 2001, Thompson 2005, Bowen 2007). Thompson (2005) reported that understory thickets (often in canopy gaps) provide important habitat for Swainson's Warbler, one of our surrogate species, and its associates. Artificially creating these gaps mimics a natural tree-fall event (Zimmerman 2010), similar to tornado touchdowns, lightning strikes, and hurricanes.

Thompson (2005) further stated that larger understory thickets provide better habitat than smaller ones. Thickets in Thompson's core Swainson's Warbler habitats at the Woodbury Tract study site (Britton's Neck, South Carolina) were larger (784 square meters/2.1 acres) than in the non-core areas (518 square meters/1.38 acres). That said, the best evidence available suggests that patch clear-cuts of at least two acres, located on the best high bottomland hardwood sites, should develop into productive habitat for Swainson's Warbler and its associates. Furthermore, Thomas (1996) states that creating small canopy openings, less than 4 ha (10 ac), will promote and sustain the growth of understory plant species, including cane (*Arundinaria* spp.).

To drive home the dense tangles point, Thompson (2005) describes Swainson's core breeding habitat as understory thickets "requiring a machete to traverse". She further states that thickets providing the best and most productive habitat occur on the highest bottomland hardwood sites, frequently dominated by cane, dwarf palmetto, greenbriar, peppervine, grape, poison ivy, and blackberry. Graves (2001) and Peters (1999) paint a similar picture of the structure and plant composition of Swainson's Warbler habitat. Stems and limbs of seedlings and shrubs frequently contribute to the jungle of vines, briars and other plant parts which collectively compose the nesting habitat for Swainson's Warbler (Graves 2002). The trees most commonly associated with the core breeding areas on Thompson's study area were sweetgum, red maple, ironwood, green ash and hawthorn. Those same trees species and others including laurel oak, water tupelo, elm species and persimmon describe Swainson's habitat at Audubon's Francis Beidler Forest (Brunswig 2012, personal communication). Thompson (2005) further described the most productive Swainson's understory thickets as being close to swamp sloughs but not cypress-tupelo flats and streams, with lots of dry leaves on the ground and little or no grass. They should also be as far as possible from a forest edge, ideally 1000 m or more, to reduce Brown-headed Cowbird parasitism (Twedt and others 1999). These high bottomland hardwood forest sites, where the best thickets develop, not surprisingly experience infrequent and short duration flooding, leaving the dry leaves on the forest floor loose, fluffy and easy to turn for the foraging warblers. In fact, a dense leaf litter layer seems to be an essential component of good Swainson's Warbler habitat (Savage and others 2010).

There is a real premium associated with locating these understory thickets in the best possible places. The best thickets allowed Swainson's to establish and defend smaller territories (1.19 ha vs 1.56 ha on the site at large), preserving energy for the males and allowing more pairs to utilize a portion of each thicket, resulting in clusters of birds in and near the best thickets (Meanley 1969, Thompson 2005). In such situations, Thompson (2005) describes them as semi-colonial. In the very best thickets, Swainson's sometimes produced two and rarely three successful broods, greatly increasing seasonal fecundity (Meanley 1969, Thompson 2005). According to Thompson, 90 percent of the Swainson's Warblers nesting in the high density area of her study site were located less than 5 meters from an understory thicket, with 2/3 nesting inside thickets. According to Graves (2002), scattered understory thickets were the most conspicuous characteristic of Swainson's Warbler breeding territories.

Thickets developed for Swainson's Warbler will provide good habitat for several other bottomland birds in need of conservation action, such as Hooded Warbler, Kentucky Warbler, White-eyed Vireo and Eastern Towhee. Kentucky Warbler and Eastern Towhee are rated High Priority and Hooded Warbler and White-eyed Vireo are rated Moderate Priority on the SAMBI list, providing the opportunity to assist a suite of birds with one management prescription.

While not used for nesting, understory thickets can provide good habitat for post-fledging Prothonotary Warbler families. At Francis Beidler Forest, these birds were frequently seen in late summer in a 10-acre thicket located more than 2000 feet from the nearest good Prothonotary Warbler nesting habitat (Personal Communication, Norman Brunswig). When planning to create new understory thickets, proximity to Prothonotary Warbler nesting habitat should be considered in the decision making process. Additionally, any bottomland hardwood stand located on the upland/wetland ecotone being considered for harvest by clearcut and natural regeneration, should be considered a likely future Swainson's Warbler nesting area and Prothonotary Warbler post-fledging habitat area (i.e. natural regeneration thickets).

Species Background – Bottomland Hardwood Thinning

In their studies of bird responses to bottomland hardwood forest management in 124 timber stands on 12 Louisiana wildlife management areas, Norris and others, (2009) reported that some interior species such as Red-eyed Vireo and Yellow-throated Vireo, which are most abundant in mature bottomland hardwood forests, experienced population declines in response to any forest harvest regime (individual selection, group selection or extensive harvest with more than a 40 percent reduction of canopy closure). For these species, maintaining large areas of closed canopy, interior forest is critically important. However, they also found that the densities of other species that prefer mature bottomlands with closed or near closed canopies, such as Prothonotary Warbler, Acadian Flycatcher and Yellow-billed Cuckoo, did not decline dramatically in response to selective timber harvest which did not reduce canopy closure to below 60-70 percent. Additionally, they found that eight other species, including Eastern Wood Peewee, Carolina Wren, Red-bellied Woodpecker and Wood Thrush, achieved their greatest densities in stands which had been harvested using single tree selection and maintaining the same 60-70 precent canopy closure limit.

It is important to note that three of the species mentioned above as not experiencing dramatic population declines in response to single tree selection (Prothonotary Warbler, Acadian Flycatcher and Yellow-billed Cuckoo), and two of the species that reached their greatest densities in stands which had been harvested using single tree selection, both with the minimum canopy closure limit of 60-70 percent, (Eastern Wood Peewee and Red-bellied Woodpecker) appear on the South Atlantic Migratory Bird Initiative (SAMBI) list of bottomland hardwood bird species needing conservation attention. Prothonotary Warbler, Yellow-billed Cuckoo and Eastern Wood Peewee are listed as High Priority and Red-bellied Woodpecker and Acadian Flycatcher as Moderate Priority.

To add additional weight to the very high conservation priority we assign to Prothonotary Warbler (US Fish and Wildlife Service Species of Conservation Concern), we should also acknowledge the special Responsibility Species status enjoyed by Prothonotary Warbler, Northern Parula, Yellow-throated Warbler and Yellow-throated Vireo, which are all bottomland hardwood species. All four species are also ranked as High Priority for conservation in the Partners in Flight Physiographic Plans and will respond well to no harvest and/or single tree selection harvest in bottomland hardwoods. The following is a list of the four Responsibility Species with an estimate of the total global population of each thought to nest in North and South Carolina (Partners in Flight Science Committee 2013): Northern Parula (11.5 percent), Yellow-throated Warbler (20 percent), Prothonotary Warbler (24.4 percent), and Yellow-throated Vireo (10.4 percent).

Results reported by Norris and others (1999) suggest that thinning established bottomland hardwood stands, using single tree selection which does not reduce canopy closure below 60-70 percent, improves habitat for several birds that need conservation action, while not dramatically degrading the habitat for several others that also need such help. Additionally, while small clearcuts can create habitat for Swainson's Warbler, thinning may improve their habitat too (Somershoe and others 2003).

For Red-eyed Vireo, Yellow-throated Vireo and other species which are very sensitive to any opening of the canopy, large areas of no-harvest forest should be retained where possible. This outcome might be achieved by thinking and planning across ownership boundaries, such that nearby stands and forests known to be managed with a mature, old growth, no-cut philosophy and likely to remain so can meet the needs of the Red-eyed Vireo and its associates, freeing other landowners and managers to practice other bird-friendly forestry activities.

RESULTS AND DISCUSSION

Bird Friendly Best Management Practices Recommendations

In this program, the Prothonotary Warbler and Swainson's Warbler will serve as surrogate species for the several other birds which share similar structural habitat requirements. It is estimated that North and South Carolina support 22 percent of the global population of nesting Prothonotary Warblers and 12 percent of Swainson's Warblers (Partners in Flight Science Committee 2013), so intentional management for these species is particularly important. Because we have such a high percentage of their global populations, we have both great responsibility and great potential to help them.

Given the dual and equally important goals of optimizing both understory thickets for Swainson's Warbler and its allies, and mature interior forest habitat for Prothonotary Warbler and its allies, the prevailing criteria that we suggest for bird-friendly bottomland hardwood management will be staying above 60-70 percent canopy closure across large bottomland hardwood forest landscapes, and limiting the total area of forest which falls below 40 percent canopy closure to 10 percent of any given bottomland hardwood forest that is greater than 2,000 acres. Means for achieving these management suggestions are outlined below. Note that the below suggestions are adapted from, at least in part, the Lower Mississippi Valley Joint Venture's recommendations for forest types in that area (LMVJV Forest Resource Conservation Working Group 2007), as well as the available literature in the species review to follow.

Desired Habitat Condition Options: No Harvest

Landowners that desire to create and/or maintain old growth on part or all of their bottomland hardwood forest, and timber revenue is not a major concern, should certainly do so. In such cases, obviously, no harvest is needed. Many interior forest bird species, such as Prothonotary Warbler, Red-eyed Vireo and Yellow-throated Vireo, find their very best habitat in such undisturbed forest stands. These lands can function as the old-growth cores, around and adjacent to which other owners in the landscape can provide the other forest habitat types and successional stages.

Desired Habitat Condition Options: Modified Approach

When some revenue is desired, multiple clear-cuts of up to 10 acres in size, up to 10 percent of the forest stand, could be recommended to create regenerating, dense understory habitat (aka "thickets"). Thickets can be isolated or near one another. In the latter situation, two or more thickets might be connected by a single skid trail, to minimize the area of disturbance and fragmentation resulting from a larger single cut, and avoid multiple loading decks. The result would be a "string of pearls" harvest design (fig. 1a). An alternate design would be to locate the loading deck in the middle and cut patches like spokes of a wheel (fig. 1b). Foresters should also consider the surrounding forest landscape when deciding upon the location of clearcuts. If possible, avoid placing clearcuts adjacent to existing early-successional habitat, such as agricultural land, even if such land is on a different stand or belongs to a different landowner.

It is also a recommendation to "feather" the edges of any clearcuts to soften the edge effect (fig. 2). This will help prevent nest parasitism by the Brown-headed Cowbird. Brown-headed Cowbirds are brood parasites, meaning that they lays their eggs in nests of other species. The foster parents then unknowingly raise the young cowbirds, usually at the expense of their own offspring. Cowbird eggs require a shorter incubation period than most other songbirds and thus usually hatch first. Cowbird nestlings also grow large very quickly. These advantages allow them to command the most food from their foster parents, usually resulting in reduced nesting success of the host species.

Feathering edges—If clearcuts alone do not provide a sufficient economic opportunity to justify a harvest, landowners can thin bottomland hardwood forest stands using single tree selection, so long as they do not reduce the canopy closure from 90 percent (fig. 3a) to below 60 percent-70 percent (fig. 3b). Foresters can select trees for removal based on whatever silvicultural prescriptions or other goals they have, such as but not limited to improved species composition, release of residual trees, maximizing mast trees, or maximizing revenue. Using a variety of techniques, foresters can estimate the before-treatment canopy closure and the impact on canopy closure of the removal of each tree as it is being marked for harvest.
If understory thickets are to be created by multiple clearcuts and a stand-wide thinning is to take place as parts of the same harvest operation in the same stand, it is important to mark the thickets (clearcuts) first, to know how many acres and how much canopy opening will result from them before marking the thinning.

When determining the number and size of understory thickets to be created in a particular stand, the canopy openings generated from the combined harvest (clearcuts and thinning) must be included in the total canopy opening estimate. To reiterate, for the best possible bottomland hardwood bird habitat conservation outcome, clearcuts should occupy no more than 10 percent of a given stand and all accompanying thinning should not reduce canopy closure below 60-70 percent. We feel that these

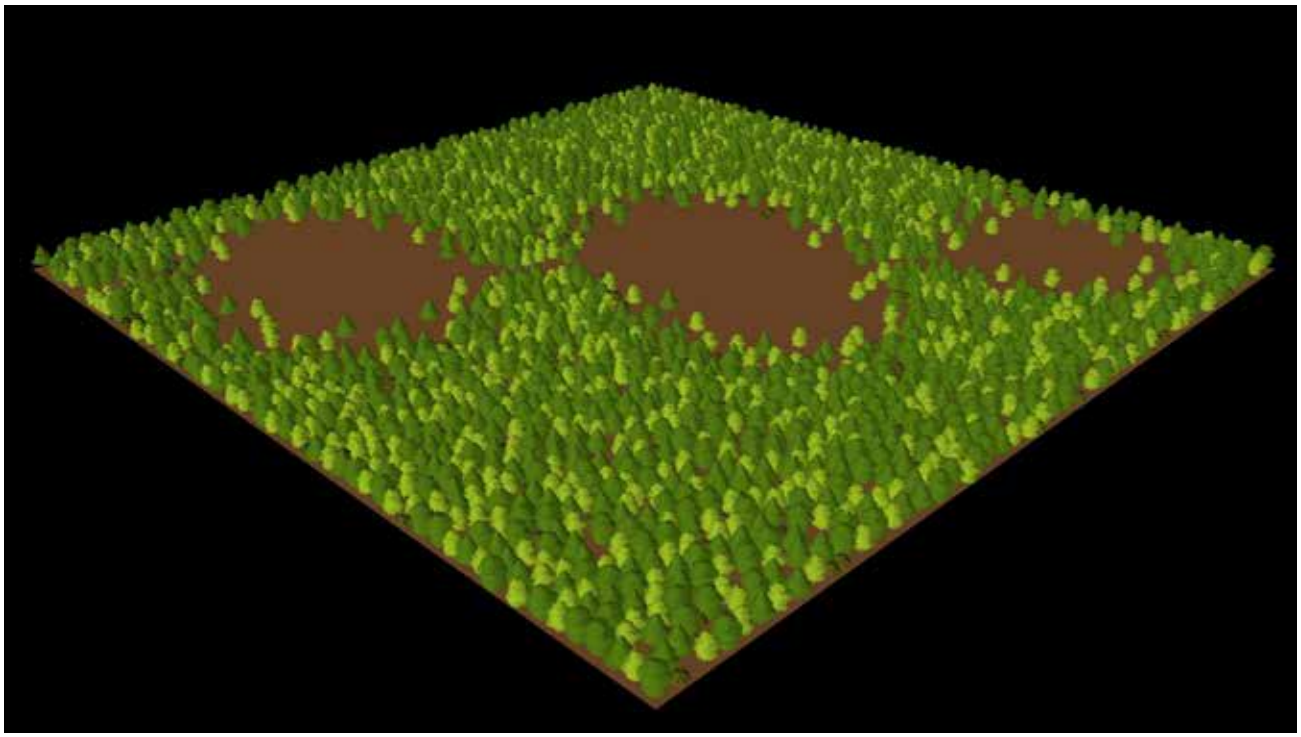


Figure 1a—String of Pearls.

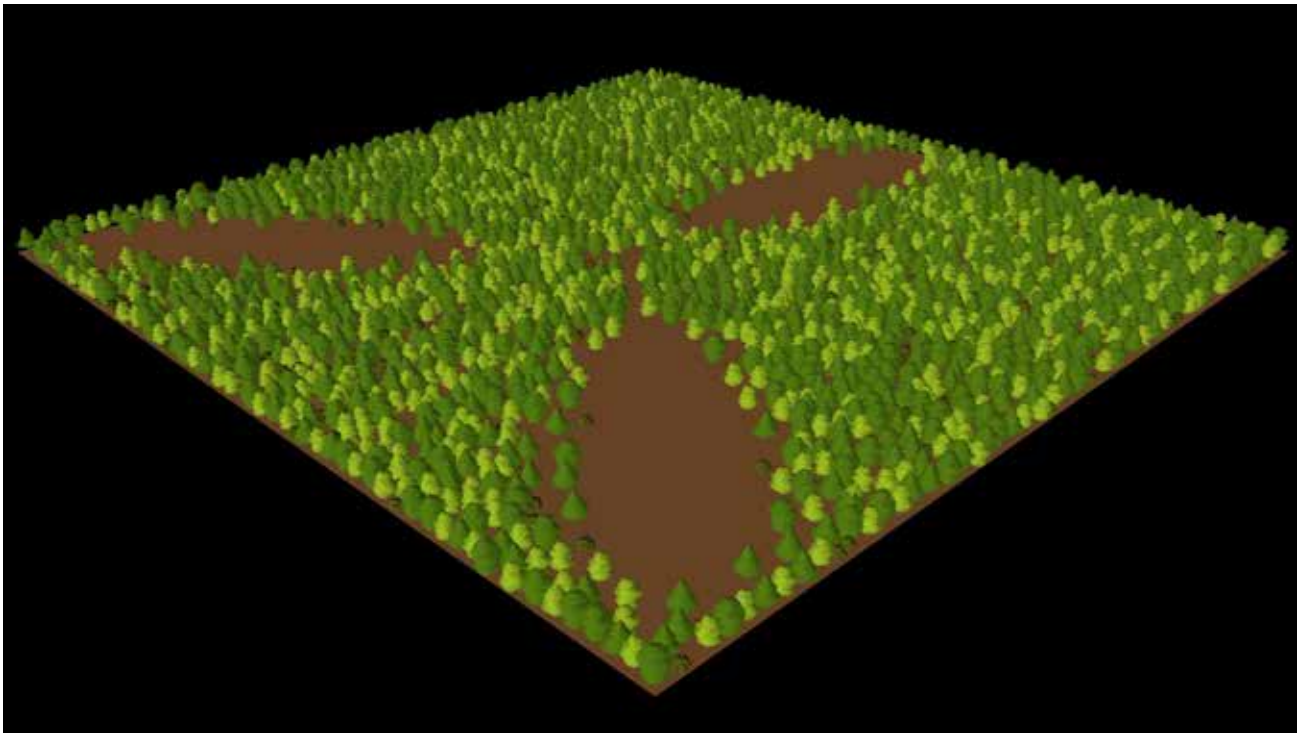


Figure 1b— Spokes of Wheel.



Figure 2—Feathered Edges of Clearcut.



Figure 3a—95 percent canopy closure, looking up from within stand.

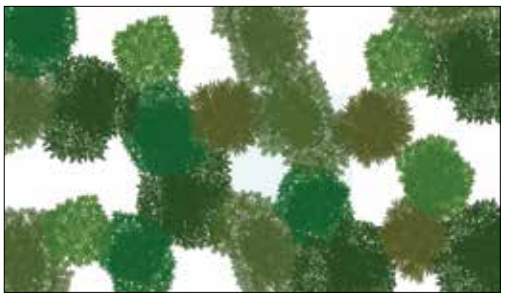


Figure 3b—65 percent canopy closure.

recommendations do the best possible job of benefiting both birds and landowners in this forest type.

Additional Species to Consider

Swallow-tailed Kites represent another priority species of southeastern bottomland forests (listed as High Priority on the SAMBI plan). These kites forage over bottomland hardwood forests and cattle pastures proximal to them, and nest in super-emergent trees such as bald cypress in floodplains and loblolly pines in or near swamps or on swamp islands. Simple steps such as retaining very tall cypresses or other trees in swamp Streamside Management Zones and elsewhere in and near bottomlands can create and sustain habitat for this elegant, charismatic bird. Similarly, electing to retain very tall swamp loblolly pines wherever they exist can also provide this important nesting habitat. Yellow-throated Warbler, which has an affinity for bald cypress in bottomland forests, may also be benefitted by retaining tall cypresses (Graber 1983, Hall 1996, Gabbe 2002).

Chimney Swift nest in hollow trees, in addition to the chimneys from which they derived their name. Retaining large, hollow trees of any species in bottomland hardwood stands can help support this remarkable insect eating bird. In the case of cypress, the needs of both Swallow-tailed Kite and Chimney Swift might be met with the same conservation action.

CONCLUSIONS

Audubon has demonstrated bird friendly forest management practices at their 3,250 acre Silver Bluff Audubon Center and Sanctuary to improve diversity of habitat in bottomland hardwood forests by adjusting the management. Staff logged three small clearcuts totaling 9 acres within a 90 acre even-aged bottomland hardwood stand and thinned an additional 70 acres, keeping the overall canopy closure to >60 percent. (fig. 4). An additional 10 acres within the stand were left undisturbed as a buffer along the Savannah River. The management followed the recommended thresholds that can be tolerated by most interior species, while allowing for new openings to develop into early successional habitat to support additional species such as Swanson's Warblers. Given that the 90 acre stand is embedded in an expansive landscape >7,000 acres of undisturbed bottomland hardwoods, the silvacultural activities address all the suites of species. The demonstration site has been a focal point at multiple outreach workshops for foresters and landowners. It has been one year since the clearcuts were completed. In May 2015, Swainson's warblers were sighted and heard in the recent clearcuts, and the majority of the disturbance tolerant species have also returned to the portions of this forest stand that were thinned. The canopy that was thinned to 60 percent has been restored to almost 90 percent. When the canopy closure of the clear cuts reaches 95 percent,

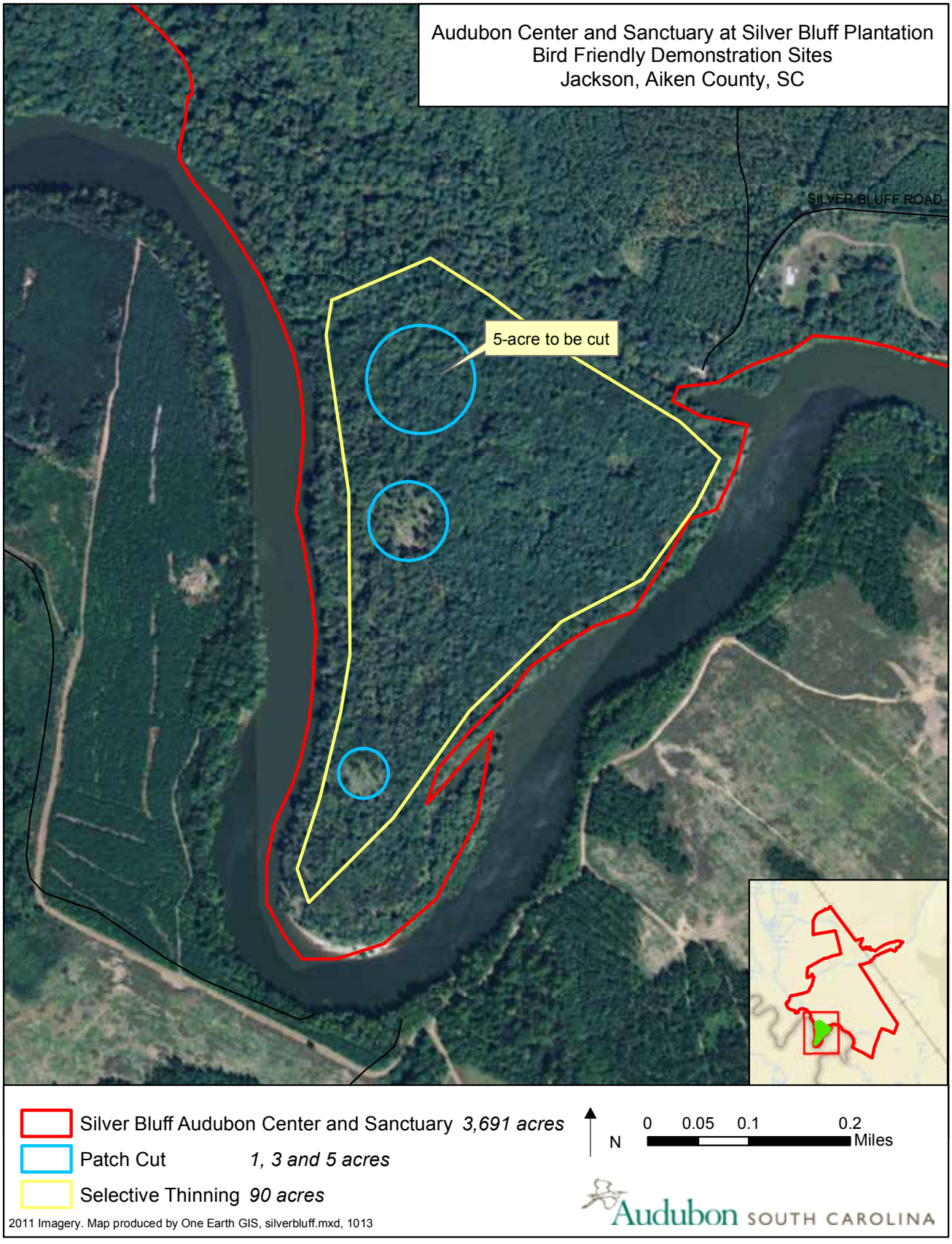


Figure 4—Thinning and Clearcut at Silver Bluff 90 acre Island.

expected in 5-7 years, Audubon will identify and cut an additional 10percent of the mature forest. Forty-five acres along the east side of the stand will not be disturbed and instead, allowed to revert to old growth bottomland forest. Thinning, as needed to continue to create new habitat, will be allowed in this part of the stand. Following the successful implementation at the Silver Bluff demonstration site, Audubon South Carolina is seeking to replicate their success at a similar demonstration site on the 17,000 acre Audubon Center and Sanctuary at Francis Beidler Forest in Dorchester County, SC.

ACKNOWLEDGMENTS

Audubon South Carolina appreciates the efforts and guidance of its partners, including American Forest Management, Audubon North Carolina, Audubon Vermont, and Lowcountry Open Land Trust. In addition, Audubon appreciates Together Green, an Audubon program funded by Toyota, for providing the startup funds for this project, and Sustainable Forestry Initiatives, Inc. for continuing to fund the research and outreach efforts.

LITERATURE CITED

Abernethy, Y. A. 1987. U.S. forested wetlands:status and changes 1940-1980. Biosciences. 37: 721-727.

Bowen, L. C. 2007. Seasonal bird use of canopy gaps in bottomland forest. Wilson Journal of Ornithology. 119: 77-88.

Gabbe, A. P. 2002. Tree-species preferences of foraging insectivorous birds: implications for floodplain forest restoration. Conservation Biology. 16: 462-470.

Gosselink, J. A. 1989. Cumulative impact assessment in bottomland hardwood forests. Wetlands. 9: 89-174.

Graber, J. R. 1983. Illinois birds: wood warblers. Champaign, IL: State of Illinois, Department of Energy and Natural Resources, Natural History Survey Division. Biological Notes. 118: 144 p.

Graves, G. 2001. Factors governing the distribution of Swainson's Warbler along a hydrological gradient in the Great Dismal Swamp. Auk. 118(3): 650-664.

Graves, G. 2002. Habitat characteristics in the core breeding range of the Swainson's Warbler. Wilson Bulletin. 114(2): 210-220.

Hall, G. 1996. Yellow-throated warbler (*Dendrocia dominica*). In: A. Poole and F. Gill (eds.), The Birds of North America, No. 223. Philadelphia, PA: Academy of Natural Sciences, and The American Ornithologists' Union, Washington, D.C. Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/223 doi:10.2173/bna.223. [Date accessed unknown].

Hunter, W. D. 2001. Conservation of disturbance-dependent birds in eastern North America. Wildlife Society Bulletin. 29: 440-455.

Meanley, B. 1969. Pre-nesting and nesting behavior of the Swainson's Warbler. Wilson Bulletin. 81(1): 246-257.

Moorman, C. E.; Guynn, D.C. 2001. Effects of group-selection opening size on breeding bird habitat use in a bottomland forest. Ecological Applications. 11: 1680-1691.

Norris, J. M.; Chamberlain, M.J.; Twedt, D.J. 2009. Effects of wildlife forestry on abundance of breeding birds in bottomland hardwood forests of Louisiana. Journal of Wildlife Management. 73(8): 1368-1379.

Partners in Flight Science Committee. 2013. Population Estimates Database, version 2013. http://rmbo.org/pifpopestimates. [Date accessed: December 11, 2014].

Pashley, D.N.; Barrow, W.C. 1993. Effects of land use practices on neotropical migratory birds in bottomland hardwood forests. In: Finch, D; Stangel, P., eds. Status and management of neotropical migratory birds. Gen. Tech. Rep. RMRS-229. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 315-320.

Peters, K. 1999. Swainson's Warbler (*Limnothlypis swainsonii*) habitat selection in a managed bottomland hardwood forest in South Carolina. Raleigh, NC: North Carolina State University. 91 p. M.S. thesis.

Savage, A. L.; Moorman, C.E.; Gerwin, J.A.; Sorenson, C. 2010. Prey selection by Swainson's warblers on the breeding grounds. The Condor. 112: 605-614.

Somershoe, S.G.; Hudman, S.P.; Chandler, C.R. 2003. Habitat use by Swainson's Warblers in a managed bottomland forest. Wilson Bulletin. 115(2): 148-154.

Thomas, B. W. 1996. Habitat selection and breeding status of Swainson's Warblers in southern Missouri. Journal Wildlife Management. 60(3): 611-616.

Thompson, J. 2005. Breeding biology of Swainson's Warblers in a managed South Carolina bottomland forest. Raleigh, NC: North Carolina State University. 177 p. Ph.D. dissertation.

Twedt, D.J., Wilson, R.R.; Henne-Kerr, J.L.; Hamilton, R.B. 1999. Impact of forest type and management strategy on avian densities in the Mississippi Alluvial Valley, USA. Forest Ecology and Management. 123(2-3): 261-274.

Twedt, D.J.; Loesch, C.R. 1999. Forest areas and distribution in the Mississippi Alluvial Valley: implications for breeding bird conservation. Journal of Biogeography. 26: 1215-1224.

Wilson, R.; Ribbeck, K.; King, S.; Twedt. eds. 2007. LMVJV forest resource conservation working group. Restoration, management, and monitoring of forest resources in the Mississippi Alluvial Valley: recommendations for enhancing wildlife habitat. Vicksburg, MS: Lower Mississippi Valley Joint Venture Office. 137 p.

Wright, E.A. 2002. Breeding population density and habitat use of Swainson's Warbler in a Georgia floodplain forest. Athens, GA: The University of Georgia. 99 p. M.S. thesis.

Zimmerman, J.A. 2010. Habitat relationships of breeding birds in bottomland hardwood forests of the Altamaha River, Georgia. Statesboro, GA: Georgia Southern University. 73 p. M.S. thesis.

SLASH PINE REGENERATION AND GROUNDCOVER RESPONSES FOLLOWING HARVESTING IN HYDRIC FLATWOODS

Kimberly Bohn and Ajay Sharma[1]

Abstract—As part of a long term project assessing different strategies for converting slash pine (*Pinus elliottii*) plantations to uneven-aged, multifunctional forests, we evaluated the effects of five different harvest methods (group selection, shelterwood, third row thin, 'cut 2 leave 3' row thin, and staggered third row thin) on natural regeneration of slash pine and groundcover one growing season following harvesting. While shelterwood and group selection harvests resulted in highest total number of slash pine seedlings of any size (9708 and 9132 seedlings/ha), group selection and 'cut 2 leave 3' row thinning had the highest number of large sized seedlings (408 and 340 seedlings greater than 60 cm). Quick height growth of slash pine regeneration will be a critical component of the stand conversion process, particularly where prescribed fire may be used for vegetation control. Although, the harvest treatments resulted in considerable decreases in shrub cover as a result of the mechanical operations, the shrub component still dominated ground cover in all of the treatments after one growing season. Shelterwood treatment had the least shrubs (approximately 12 percent) and had the highest proportion of graminoids (approximately 25 percent of total groundcover) compared to other harvest treatments. Total species richness varied from 21 genera in uncut control to 40 genera in staggered third row thin treatment. The most appropriate harvesting regime for converting plantations may be dependent on prioritizing objectives for restoring both forest structure in addition to groundcover diversity.

INTRODUCTION

In the last few decades, concerted efforts have been made to restore southern pine ecosystems throughout the southeast, particularly in context to promoting biodiversity within longleaf pine (*Pinus palustris*) ecosystems. Across the landscape, the natural, pre European settlement, longleaf pine ecosystems were typically variable in density, tree sizes and age classes (Pederson and others 2008, Platt and Rathbun 1993). This natural structural variability, along with a frequent regime of low intensity fires sparked by lightening, were key components that made this particular ecosystem rich in plant and wildlife diversity. Interestingly, natural characteristics of slash pine (*Pinus elliottii*) dominated ecosystems are similar to longleaf pine ecosystems in diversity, structure, and function (Doren and others 1993); and in fact, in wetter hydric ecosystems, longleaf pine may occur only as a co dominant or even subordinate species to slash pine.

As of 2007, slash pine including mixed stands with longleaf pine forests covered 5.3 million hectares (Smith and others 2009), with about 79 percent of the total area concentrated in Florida and Georgia (Barnett and Sheffield 2005); however most of that acreage was in plantation management. Interest in restoring and managing more structurally complex flatwoods with a component of slash pine in addition to longleaf pine has increased recently, yet less is currently known about appropriate management of slash pine forests or natural slash pine regeneration dynamics than longleaf pine. Two-aged and uneven-aged silvicultural methods have been proposed as techniques for restoring and maintaining these ecosystems. A basal area approaching 4 to 5 m^2/ha, as would be used with the irregular shelterwood method (Dickens and others 2004, Langdon and Bennett 1976) or up to 11.5 m^2/ha, typical of single and group selection methods, have been suggest to promote early seedling establishment or growth of slash pine (McMinn 1981). Complicating the matter, slash pine seedlings are not tolerant of fire, though saplings become more resistant as they mature, grow in tree height, and the bark thickens (Doren and others 1993). Thus, a main question for land managers remains regarding the feasibility of uneven-aged slash pine management and restoration of groundcover in areas where prescribed fire is intended for fuels reduction.

An operational scale project was designed to investigate the long term strategies for converting slash pine plantations to more 'natural' uneven-aged ecosystems. The overall objectives of that project are to determine the harvesting strategies that lead to sustainable uneven-aged forests in terms of forest structure, timber production and other ecological

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services such as ground cover diversity and carbon sequestration. The initial conversion harvesting treatments ranged from traditional thinning operations to uneven-aged methods. One growing season after harvesting, slash pine regeneration and groundcover responses were measured in order to quantify and compare tree regeneration densities and sizes and groundcover response within and between 5 conversion harvesting treatments in slash pine plantations.

METHODS

Conversion harvesting treatments were installed in mature slash pine plantations at Tate's Hell State Forest, Florida, in 2009. Tate's Hell State Forest (29.83N, 84.79W) consists of about 820 km^2 of poorly drained lowland mesic hydric flatwoods site between the Apalachicola and Ochlockonee Rivers in the panhandle Florida. The climate is humid subtropical with annual precipitation totaling about 147 cm, of which about 49 percent is received during June to September. Although more than 40 unique soil types occur within the forest, four groups account for the majority of the soils, namely, (a) Scranton Rutlege, (b) Plummer Surrency Pelham, (c) Meadowbrook Tooles Harbeson, and (d) Pamlico Pickney Maurepas. All are poorly drained hydric soils. The site was once a swampy mosaic of wet prairies, cypress (*Taxodium* spp.) sloughs, Atlantic White Cedar (*Chamaecyparis thyoides*) forests and other wetland and pine flatwoods communities, but large scale silvicultural operations and hydrological manipulations during 1960s through 1980s converted extensive areas of native habitats to slash pine plantation. Stands were established following intensive mechanical site preparation, bedding and planting at high densities.

Five conversion harvest treatments were completed in December 2011 in 30 yr old slash pine plantations with initial basal area 30 m^2/ha and a QMD of 18 cm. Treatments consisted of three intensities of thinning (3rd row thinning, "take 2 rows, leave 3" thinning, and "stutterstep across rows"), irregular shelterwood method to a residual basal area of 9 m^2/ha , and group selection method in which gap openings of 0.10, 0.20, 0.40 and 0.08 ha sizes were created, with the remaining matrix third row thinned. Each harvest treatment was 6 ha in size and was replicated across 3 unique, noncontiguous blocks across the forest.

Each of the harvest treatment plots had five permanent measurement plots of 25m x 25m, which were located at random grid points within each treatment plot. Within each 25m x 25m measurement plot, we established two 5m x 5m tree regeneration plots at diagonally opposite corners (south east and north west corners) and a 15m line transect oriented north to south from which to estimate understory and groundcover species (fig. 1).

Densities of tree seedlings < 1.4m in height were recorded on each of the ten regeneration plots located within the permanent measurement plots described above as well as on an additional ten 5m x 5m regeneration plots randomly established outside the permanent measurement plots to account for a more detailed response. Other attributes of the regeneration plots including their position within the harvest treatment (e.g. thinned vs. unthinned area, gap vs. matrix), vegetation condition (clear vs. shrubby) and litter status were also noted. Each recorded seedling was sorted by height/size category (<30cm, 30 to 60cm,

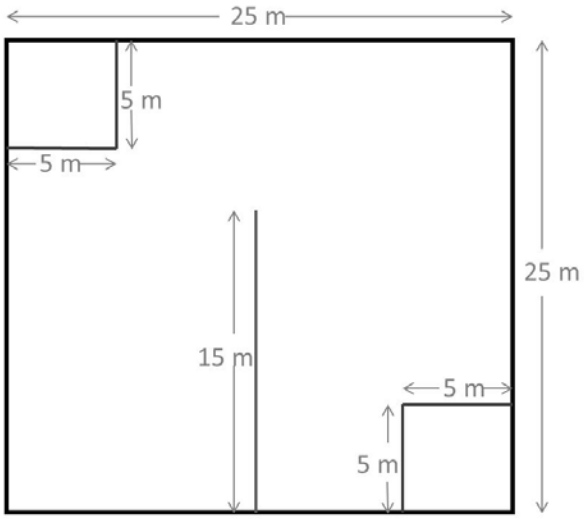


Figure 1—Layout of a 25m x 25m measurement plot with two 5m x 5m regeneration plots at SE and NW corners and a 15m line transect used to sample natural regeneration and groundcover.

>60cm). Analyses of variance were carried out to test the effects of harvest treatments on seedling densities. Tukey's HSD (Honestly Significant Differences) test was performed at α= 0.05 to test for significant differences.

Groundcover, including forbs/herbs, graminoids (grasses, sedges, and rushes), and shrubs and vines, were sampled along 15m line transects. We also included small tree seedlings along 15m line transect in our assessment of groundcover. Using the line intercept method, vertically projected foliar cover was recorded for each plant species along transect. These data were then transformed into mean percent cover values for each of the harvest treatments. The groundcover assessment was carried out during September to October 2012 when the groundcover species were in a phenological stage most suitable for identification. Tree regeneration responses to the harvest treatments were assessed in April 2013 after one year of harvesting and before the beginning of next growing season.

RESULTS AND DISCUSSION

Tree Regeneration

Seedlings of slash pine were observed in all of the harvested treatment plots and were significantly higher in number than on the uncut control plots (table 1). The maximum numbers of total seedlings (all heights combined) were observed in the shelterwood (9708 seedlings/ha) and group selection (9132 seedlings/ ha) treatments (table 1), although these were not were not significantly different from other treatments. In comparison, the uncut control plots averaged 412 seedlings/ha. The majority of seedlings recorded in the uncut control were in a single plot which had been accidentally burnt prior to installation of this project, which led to creation of small gaps where the regeneration occurred.

The size of regeneration is also of importance because the large sized seedlings may be more likely to survive prescribed burns. Group selection and 'cut 2 leave 3' row thin treatments had the highest number of the large sized seedlings > 60 cm (408 and 340 seedlings/ ha respectively), though none of the treatments were significantly different due to the high variability in densities of this size class (table 1). It is likely that the largest seedlings were actually advance regeneration released by the increased light in the open areas of these two treatments. The shelterwood treatment, which had resulted in the highest total number of seedlings, actually had the least number of seedlings > 60 cm (80 seedlings/ha). Plots in the shelterwood treatment were the most affected by harvest operations in terms of disturbance to soil structure due to the extensive movement of harvest equipment across the stands. It is possible that despite abundant seedling emergence in shelterwood treatment, a large proportion of seedlings could not establish and reach larger sizes under these disturbed soil conditions.

We also more specifically compared regeneration within thinned or harvested areas and unthinned rows (table 2). Within the 'cut 2 leave 3' row thin treatment, natural regeneration in the thinned portion where rows were cut out was considerably higher than in the areas of unthinned rows (9936 total seedlings/ha vs 3952 seedlings/ha respectively). Interestingly though, in the group selection treatments, the total number of seedlings were higher in the thinned matrix portion (11388 seedlings/ha) than in the gap portion (6000 seedlings/ha), though most of the seedlings in the

Table 1—Estimates of regeneration (mean + / - one standard deviation of number of seedlings) in mature slash pine stands one year following different harvest treatments at Tate's Hell State Forest, FL

Harvest Treatment	Density (seedlings/ha) by height class:			
	<30cm	30 to 60cm	>60cm	Total seedlings
3rd Row Thin	5180 ± 2424[b]	2160 ± 656[b]	260 ± 132[a]	7600 ± 2700[b]
Cut 2 Leave 3 Row Thin	4332 ± 548[b]	1980 ± 4.95[b]	340 ± 120[a]	6620 ± 668[b]
Group Selection	6000.15 ± 4136[b]	2668 ± 1472[b]	408 ± 232[a]	9132 ± 5460[b]
Shelterwood	6420 ± 3148[b]	3208 ± 2244[b]	80 ± 52[a]	9708 ± 5232[b]
Staggered 3rd Row Thin	5292 ± 2632[b]	2892 ± 1368[b]	128 ± 24[a]	8312 ± 3736[b]
Uncut Control	172 ± 120[a]	60 ± 68[a]	180 ± 312[a]	412 ± 492[a]

Table 2—Estimates of regeneration (mean number of seedlings per hectare) observed in cut and uncut portions within the harvest treatment in mature slash pine stands after one year following different harvest treatments at Tate's Hell State Forest, FL

Harvest Treatment	Density (seedlings/ha) by height class:			
	<30cm	30 to 60cm	>60cm	Total seedlings
Cut 2 Leave 3 Row Thin				
Within Thinned Stand Area	6272	3232	432	9936
Within Unthinned Stand Area	2600	900	452	3952
Group Selection				
Within Gap	2452	2680	868	6000
Within Matrix	8532	2732	124	11388

matrix portion were in the smallest size class while the gap portion had a substantially higher number of large sized seedlings (table 2). Similar observation were made in longleaf pine forests by McGuire and others (2001) and Gagnon and others (2004) where higher survival and density of longleaf pine seedlings after one growing season was observed in matrix portion than in the gaps. In these and other studies (Palik and others 1997), the seedlings also grew to a larger size in the gaps than in the matrix portion. Additionally, the greater abundance of small seedlings in the matrix portion in our study may also be due to the proximity to greater number of seed trees. However, there was no definite spatial pattern found between the seedling density in the gaps and the distance from the gap border. These initial observations suggest a harvest treatment that combines gap openings from a group selection treatment with a 'cut 2 leave 3' row thin treatment in the matrix of the stand may be among the best treatments to obtain natural regeneration across the entire treatment area.

Though not quantified, an ocular observation of regeneration response across all the treatments suggested that the regeneration hot spots had three common attributes. These spots represented areas or patches in the stands which were (1) free of shrubs and excessive hydric conditions, (2) had sufficient illumination, and (3) did not have more than 15cm thick litter layer.

Groundcover Response

After a period of about one growing season following harvesting, all of the treatment plots had decreased amounts of total percent groundcover as compared to the uncut control (fig. 2). This initial effect was mainly due to the fact that the harvesting operations had decimated a considerable amount of shrubs that existed prior to harvesting. Shrub cover in the uncut control was 44 percent compared to 18 to 26 percent in the harvested plots. Shrubs also represented 98 percent of the relative cover of all plant functional groups in the uncut controls, but only 66 to 88 percent of relative cover in the harvested treatment plots. The shelterwood treatment, which involved the most intensive harvesting, had the least amount of total groundcover (approximately 17 percent) primarily due to the reduction in the shrub layer.

The shelterwood treatment despite resulting in the least groundcover had the highest absolute as well as relative proportion of graminoid cover (approximately 25 percent of total groundcover) of all harvesting treatments (fig. 2). These graminoids mostly consisted of sedges and rushes along with a few species of grasses typical of hydric sites. The high cover of graminoids in shelterwood treatment plots as compared to the other harvest treatments was possibly due to the lower residual basal area (which allowed high level of light at forest floor) in combination with the high moisture conditions (numerous small water pools for most of the year) created by harvest operations across the stand. Similar kinds of groundcover responses were observed in the clearcut gap portions of the group selection harvest plots where a relatively higher proportion of graminoids was also observed.

Total species richness values also differed between the uncut controls and harvested areas (fig. 3). Richness was lowest in the uncut control (only 21 genera) and as high as 40 genera in the third row thin treatment. Shrubs and graminoids typically represented the majority of all species. Group selection treatments had 39 genera, and other treatments had between 25 to 30 genera. The most dominant shrub species across all plots were giant gallberry (*Ilex coriacea)*, muscadine (*Vitis rotundifolia)*, fetterbush (*Lyonia lucida)*, titi (*Cyrilla racemiflora)*, and sweet pepperbush (*Clethra alnifolia*). The most common graminoids detected were *Andropogon glomeratus*,

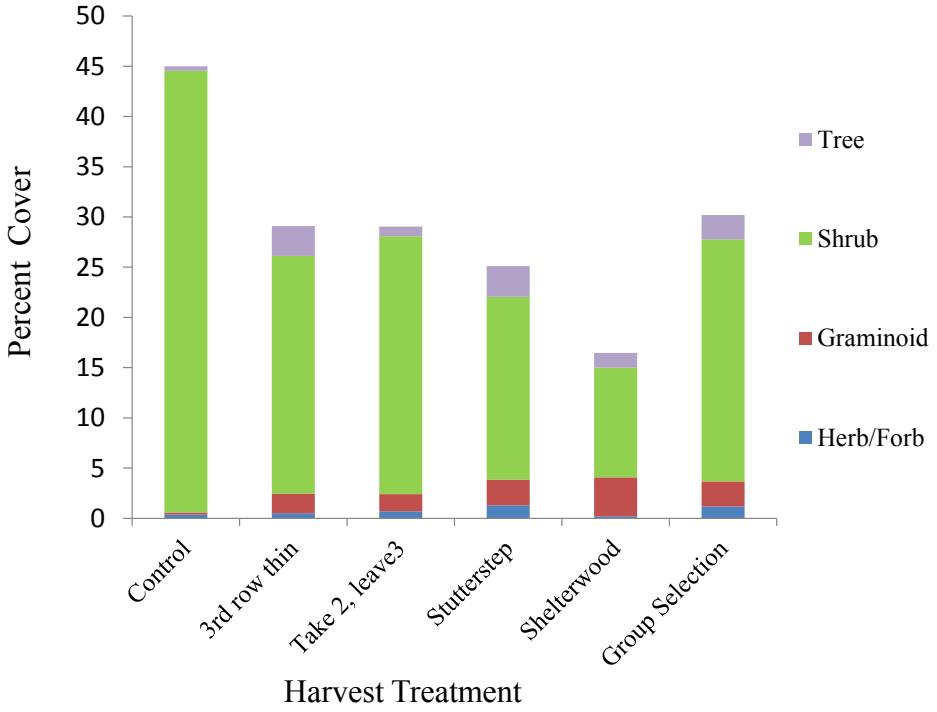


Figure 2—Percent cover and relative proportion of plant functional groups (forbs/herbs, graminoids, shrubs, and trees) observed in the groundcover one growing season following harvesting in slash pine plantations.

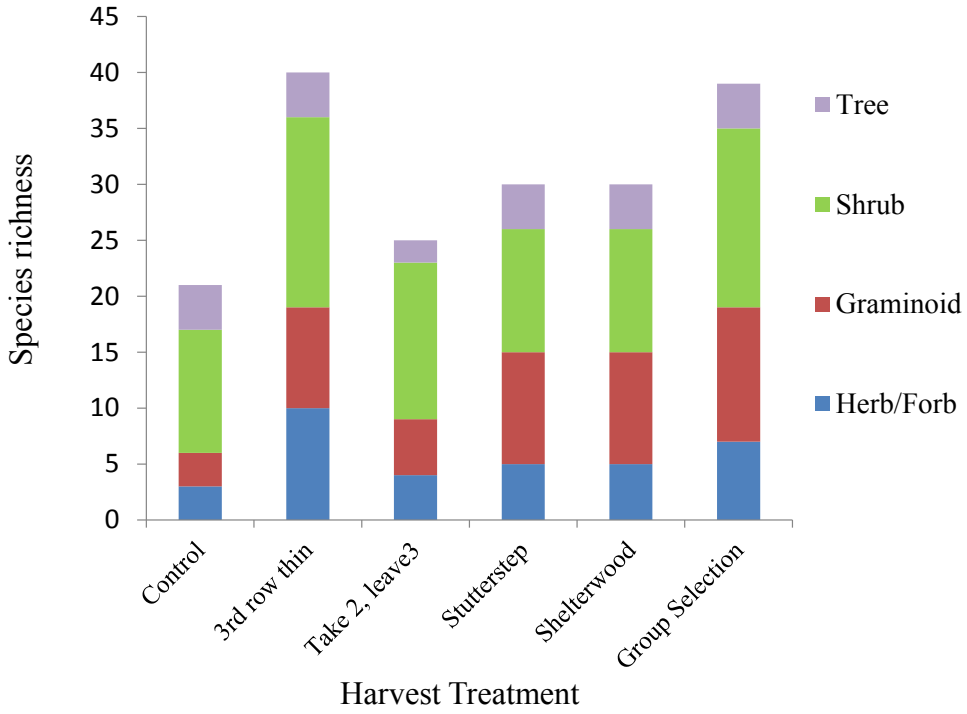


Figure 3—Total species richness across treatment plots by functional groups (forbs/herbs, graminoids, shrubs, and trees) observed one growing season following harvesting in slash pine plantations.

Juncus dichotomus, and *Rhynchospora* spp, and common forbs included *Drosera* spp., *Hypericum* spp., *Pteridium aquilinum*, and *Xyris* spp. Notably, yellow topped pitcher plants (*Sarracenia flava*) were also observed within the thinned rows of several treatment plots. Multi stemmed sprouts of sweetbay (*Magnolia virginiana*), blackgum (*Nyssa sylvatica*) and cypress (*Taxodium* spp) were the dominant tree regeneration aside from slash pine.

CONCLUSIONS

The early response of slash pine regeneration and groundcover to the harvest treatments suggests that these harvest methods have potential to initiate reestablishment of slash pine regeneration and groundcover. However, these results are only early responses and mostly represent the effects of open canopy conditions and harvest related physical changes caused to the soil and vegetation cover, in particular decreases in the shrub component. The regeneration and groundcover dynamics have the potential for considerable changes as a burn regime is introduced, which might lead to high regeneration mortality, other successional changes in groundcover, and increased decomposition of the logging residues. Given the status of groundcover, logging residues, and the seedling size and distribution, we expect considerable mortality to the existing smaller regeneration due to the introduction of burn regime. However, heterogeneity in the microtopography across treatment plots may result in small pockets of protected regeneration following fire, particularly among larger sized seedlings that could withstand a rapid and cool surface fire. Additionally, prescribed burns may create post burn conditions conducive for seed germination and seedling growth by creating receptive mineral soil floor and reducing competition (Jose and others 2006), and groundcover responses may be desirable as burning has been observed to enhance herbaceous and graminoid cover (Jose and others 2006, Kush and others 2000, Lewis and Harshbarger 1976). The treatment plots in this study were just recently burned in November 2014 and a follow up assessment of regeneration and groundcover will determine effects of re introduced fire regimes in addition to harvesting.

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LITERATURE CITED

Barnett, J.P.; Sheffield, R.M. 2005. Slash pine: characteristics, history, status, and trends. In: Dickens, E.D., Barnett, J.P., Hubbard, W.G., Jokela, E.L., eds. Slash pine: still growing and growing! Proceedings of the slash pine symposium. Gen. Tech. Rep. SRS-76. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 1-6.

Dickens, E.D.; Barnett, J.P.; Hubbard, W.G.; Jokela, E.J. 2004. Slash pine: still growing and growing! Proceedings of the slash pine symposium. Gen. Tech. Rep. SRS-76. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 145.

Doren. R.F.; Platt, W.J.; Whiteaker, L.D. 1993. Density and size structure of slash pine stands in the Everglades region of south Florida. Forest Ecology and Management. 59: 295-311.

Gagnon, J.L., Jokela, E.J., Moser, W.K., Huber, D. A. 2004. Characteristics of gaps and natural regeneration in mature longleaf pine flatwoods ecosystems. Forest Ecology and Management. 187: 373-380.

Jose, S, Jokela, E.J., Miller, D.L., eds. 2006. Longleaf pine ecosystems: Ecology, management and restoration. New York: Springer. 438 p.

Kush, J.S., Meldahl, R.S., Boyer, W.D. 2000. Understory plant community response to season of burn in natural longleaf pine forests. In: Moser, W.K., Moser, C.E., eds. Fire and forest ecology: Innovative silviculture and vegetation management. Tall Timbers fire ecology conference proceedings, No. 21. Tallahassee, FL: Tall Timbers Research Station: 32 39.

Langdon, O.G.; Bennett, F. 1976. Management of natural stands of slash pine. Res. Pap. SE-147. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 12 p.

Lewis, C.E.; Harshbarger, T.J. 1976. Shrub and herbaceous vegetation after 20 years of prescribed burning in the South Carolina coastal plain. Journal of Range Management. 29: 13-18.

McGuire, J.P.; Mitchell, R.J.; Moser, E.B. [and others]. 2001. Gaps in a gappy forest: plant resources, longleaf regeneration, and understory response to overstory removal. Canadian Journal of Forest Research. 31: 765-785.

McMinn, J.W. 1981. Site preparation for natural regeneration of slash pine. Southern Journal of Applied Forestry. 5: 10-12.

Palik, B.J.; Mitchell, R.J.; Houseal, G.; Pederson, N. 1997. Effects of canopy structure on resource availability and seedling responses in a longleaf pine ecosystem. Canadian Journal of Forest Research. 27: 1458-1464.

Pederson, N.; Varner, J.M; Palik, B.J. 2008. Canopy disturbance and tree recruitment over two centuries in a managed longleaf pine landscape. Forest Ecology and Management. 254: 85-95.

Platt, W.J.; Evans, G.W.; Rathbun, S.L. 1993. The population dynamics of a long lived conifer (Pinus palustris). American Naturalist. 131: 491-525.

Smith, W.; Miles, P.; Perry, C.; Pugh, S. 2009. Forest resources of the United States, 2007: A technical document supporting the Forest Service 2010 RPA assessment. Gen. Tech. Rep. WO-78. Washington, DC: U.S. Department of Agriculture. 336 p.

SILVICULTURAL TREATMENTS TO IMPROVE PONDBERRY STEM LENGTH GROWTH

Brian Roy Lockhart[1]

Abstract—Pondberry (*Lindera melissifolia* (Walter) Blume) is a deciduous woody shrub in the Lauraceae that is endemic to low-lying forests in seven southeastern states. In the Mississippi Alluvial Valley, pondberry occurs in the understory of bottomland hardwood forests. This rare shrub was listed as an endangered species in 1986. The U.S. Fish and Wildlife Service published a pondberry recovery plan in 1993, establishing objectives for recovery and delisting the species. One action step to delisting pondberry is the development of management practices to recover and conserve the species. The Center for Bottomland Hardwoods Research, Forest Service, U.S. Department of Agriculture (Southern Research Station) initiated a comprehensive research program to study pondberry in 2002. Findings from much of this research, particularly research on pondberry ecophysiology, provide implications for active management of this species. This manuscript draws upon research findings to propose silvicultural treatments for improving stem length growth of existing pondberry plants. I also describe field experiments that are needed to confirm that proposed silvicultural treatments will sustain existing pondberry colonies.

INTRODUCTION

Pondberry (*Lindera melissifolia* (Walter) Blume) is a woody shrub in the Lauraceae. It is extant in seven states across the southeastern United States including Alabama, Arkansas, Georgia, Mississippi, Missouri, North Carolina and South Carolina, and presumed extirpated from Louisiana and Florida. The species was listed in 1986 as endangered under the federal Endangered Species Act of 1973 (Currie 1986). A subsequent recovery plan was completed in 1993 with the objective to delist the species (DeLay and others 1993). Based on this recovery plan, pondberry would be down-listed from endangered to threatened when there are 15 protected, self-sustaining populations distributed throughout the species' historic range, and would be considered for delisting when there are 25 such populations (DeLay and others 1993). A recent 5-year review by the U.S. Fish and Wildlife Service indicated no change in the endangered status of pondberry as populations were considered stable or declining depending on location (Wiggers 2014).

Little knowledge of pondberry biology was available when the initial recovery plan was written (Klomps 1980, Priest and Wright 1991, Richardson and others 1990, Tucker 1974, Wright 1989, 1990a, 1990b, 1994). In the past 20 years, information on pondberry biology has greatly increased (Aleric and Kirkman 2005a, 2005b, Beckley and Gramling 2013, Connor and others 2007, 2012, Devall and others 2001, Echt and others 2006, 2011, Fraedrich and others 2011, Godt and Hamrick 1996, Gustafson and others 2013, Hawkins and others 2007, 2009a, 2009b, 2010, 2011, Lockhart and others 2012, 2013, Smith and others 2004, Taylor 2008, Unks 2011, Unks and others 2014). A formal program of pondberry research in the Mississippi Alluvial Valley (MAV) was initiated in 2002 by the Forest Service, U.S. Department of Agriculture, in cooperation with the U.S. Army Corp of Engineers and the U.S. Fish and Wildlife Service. This research was prompted by the need for more biological information on pondberry in the MAV that would better inform the question as to whether construction of the Yazoo Backwater Pump Project proposed by the U.S. Army Corp of Engineers would affect pondberry populations in the Delta National Forest, Sharkey County, MS. After 12 years of research on the ecology and ecophysiology of pondberry in the MAV, information has been developed to consider silvicultural treatments designed to increase pondberry stem length growth. My objectives in this paper are to 1) review the life history characteristics of pondberry, including recent research findings, 2) use this information from pondberry life-history characteristics to propose silvicultural treatments to increase pondberry stem length growth, and 3) discuss potential silviculture research that could increase pondberry stem length growth in the MAV. The focus of this paper is pondberry found in the MAV unless otherwise explicitly noted.

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LIFE-HISTORY CHARACTERISTICS

Pondberry is a deciduous, aromatic shrub that grows 30 cm to 2 m in height. Disjunct populations grow in seasonally flooded bottomland hardwood forests in the MAV (Hawkins and others 2009b) and along the edges of isolated Carolina bays, limestone sink ponds, sand ponds and lowland sand prairie depressions in the southern Atlantic Coastal Plain and eastern Gulf Coastal Plain (Wiggers 2014). Hydroperiod at these latter sites is typically maintained by precipitation, although some sites may be supplemented with groundwater. In contrast, hydroperiod in bottomland hardwood forests is thought to be maintained by overbank flooding, local precipitation or storage in depressions (Wiggers 2014). Habitat destruction, fragmentation, altered hydroperiods, and competing vegetation are potential threats to pondberry populations. Further, pondberry is susceptible to the lethal laurel wilt, a vascular disease caused by the fungus *Raffaelea lauricola*, which is transmitted by the invasive redbay ambrosia beetle (*Xyleborus glabratus)* (Fraedrich and others 2011). Therefore, pondberry populations are particularly vulnerable to local extirpation (Beckley 2012).

Pondberry is considered flood tolerant. However, recent research indicates that soil flooding is not required for the development of vigorous pondberry plants. For example, Lockhart and others (2013) showed 2-year-old pondberry plants were minimally affected by 45 or 90 consecutive days of soil flooding over two growing seasons in an outdoor flooding impoundment facility. Survival, stem length and stem diameter were similar between non-flooded and flooded plants. However, Hawkins and others (2009a) reported less biomass for juvenile pondberry receiving 30 days or 60 days of soil flooding than for plants receiving no soil flooding. Hawkins and others (2009a) indicated that actively-growing first-year plants are not adapted to tolerate extended flooding conditions.

Wright (1990b) hypothesized that soil flooding serves to minimize interspecific competition in pondberry populations. Wright (1989) found competing species had twice the stem length as pondberry when grown at the higher end of a pond bank, suggesting that soil flooding at the lower end of the bank was helpful in reducing interspecific competition. Pondberry populations are found in areas where they are most competitive but not necessarily in areas where they grow optimally in the absence of competition. Possible competing species, such as American buckwheat vine (*Brunnichia ovata* (Walters) Shinners) (Wright 1990b), *Smilax* spp. and *Vitis* spp. (Hawkins and others 2010), are less flood tolerant than pondberry and therefore may be at a competitive disadvantage when found in areas with longer hydroperiods.

Pondberry in the MAV is observed in the understory of mature bottomland hardwood forests (Klomps 1980, Priest and Wright 1991, Smith 2003, Wright 1989). Therefore, it has been classified as shade tolerant (Devall and others 2001). Wright (1990a) showed that pondberry could generate positive net photosynthesis at light levels as low as 5 percent of full sunlight, and increased photosynthetic rate as light increased to about 50 percent of full sunlight. Aleric and Kirkman (2005a) showed that best Atlantic Coastal Plain (ACP) pondberry growth occurred at light levels near 40 percent of full sunlight. Lockhart and others (2013) showed that pondberry plants raised under 37 percent light had greater stem length growth than plants raised beneath 70 percent or 5 percent light. Unks and others (2014), using ACP pondberry plants, confirmed the findings of Lockhart and others (2013) that low light levels (< 5 percent light) resulted in decreased growth. These studies show that pondberry can acclimate to a wide range of light availability, but best stem growth occurred at around 40 percent of full sunlight. This conclusion is further substantiated by Lockhart and others (2015). Pondberry raised for four growing seasons in 5 percent light were released to either 70 percent or 37 percent light. After three growing seasons with increased light availability, plants raised in 37 percent light had 23 percent greater stem length than plants raised in either 70 percent or 5 percent light.

Observations of extant pondberry populations found in the shaded understory of mature bottomland hardwood forests do not take into account past disturbance. Current pondberry populations may be legacies from past forest disturbance that provided conditions for establishment and development of vigorous pondberry populations. When the forest canopy closed during normal stand development processes, the amount of available light decreased, leading to decreased pondberry growth. Beckley (2012) indicated that many pondberry populations in the ACP have been exposed to anthropogenic or natural disturbances through time. Therefore, Lockhart and others (2012) also consider pondberry in the MAV a disturbance-dependent species. Field observations and results from research cited above indicate that silvicultural treatments designed to (1) increase light levels reaching the forest understory, and (2) control vegetation competing with pondberry, could be used to increase pondberry stem length growth.

SILVICULTURE TREATMENTS

Mature bottomland hardwood forests in the MAV typically have a closed or partially closed overstory canopy due to minor natural disturbances or past harvesting practices. Sunlight that filters through these canopies has led to the development of a dense midstory canopy composed of shade-tolerant

species. Available light in the understory of these forests is oftentimes less than 5 percent of full sunlight (Cunningham and others 2011, Jenkins and Chambers 1989, Lockhart and others 2000, Lhotka and Loewenstein 2006). Therefore, silvicultural treatments are needed to increase the amount of available light reaching the understory to increase pondberry stem length growth.

Research indicates that silvicultural treatments, such as midstory canopy removal, designed to increase the vigor and growth of advance bottomland red oak (*Quercus* spp.) reproduction may also be applicable to increase the stem length growth of pondberry. For example, Lockhart and others (2000) found that removal of the midstory canopy by chainsaw felling and treatment of stumps to prevent sprouting increased understory light availability to 40 percent of full sunlight. Lhotka and Loewenstein (2006) found that chainsaw felling of the midstory canopy (defined as trees not in the dominant or codominant crown class) increased understory light availability to 21 percent of full sunlight. Guttery and others (2011) found that chemical treatment of the midstory increased light availability reaching the understory by 56 percent, from 14 percent of full sunlight in untreated plots to 32 percent of full sunlight in treated plots. The high variability among these studies of understory light levels following stand treatment is due to differences in initial stand structure, species composition and level of treatment. Further, additional midstory canopy treatments may be necessary to control future competing vegetation, especially if shade-tolerant stems that were considered too small to treat in the initial treatment respond to increased light availability (Lockhart and others 2010). However, these results indicate that treatment of the midstory canopy will significantly increase light availability to understory plants such as pondberry.

While these treatments have been successful in increasing advance oak reproduction vigor and growth, they have not been scientifically tested for pondberry. Glitzenstein (2007) reported on the creation of forest canopy gaps over pondberry colonies in the Francis Marion National Forest in South Carolina. Sub-canopy trees were also felled and pondberry shrubs were clipped to induce sprouting. Pondberry can produce numerous sprouts and ramets when the shoot is clipped (personal observation). Three years following initial treatment, pondberry stem numbers increased 9 percent and total stem length (sum of the lengths of all stems in the colony) increased 119 percent (Glitzenstein 2007). Glitzenstein (2007) indicated that vegetation competing with pondberry was reduced or eliminated during the 3-year period following gap creation and initial competition control.

Glitzenstein (2007) and Lockhart and others (2015) show that pondberry will increase stem length growth with increases in light availability. Other plant species will also increase stem length with increases in light availability, including herbaceous plants and vines than can compete vigorously with pondberry. In the MAV, flood water may reduce this competing vegetation. If flood water does not reduce competing vegetation, then mechanical or chemical treatment will be needed to reduce vegetation that threatens to overtop pondberry.

In addition to variations in hydroperiod, pondberry populations in the ACP experience periodic fire disturbance. Infrequent, low-intensity fire is hypothesized to reduce competition from other plant species (Unks and others 2014). Concurrently, pondberry is able to survive such fires by regenerating from rhizomes. However, Unks (2011) cautioned that high-intensity fires may kill pondberry. Wiggers (2014) stated that the relationships between pondberry and fire is complex and requires further study. It is unknown if prescribed fire can be a silvicultural treatment in managing competing vegetation in pondberry populations in the MAV. Fire is poorly understood in MAV natural disturbance regimes and requires further study.

FUTURE RESEARCH

Priorities for research to determine silvicultural treatments that increase pondberry stem length growth would logically include long-term applied field studies with existing pondberry populations. Many pondberry colonies in the MAV are declining due to a lack of canopy disturbance, which results in low light availability. For example, a 30-m buffer is established around extant pondberry colonies on the Delta National Forest prior to tree harvesting operations (Banker and Goetz 1989). No silvicultural treatments are allowed near pondberry populations; therefore, low understory light levels remain despite forest canopy disturbance that increase light levels reaching the understory in the remainder of the stand

Overstory canopy and midstory canopy manipulation with mechanical and chemical treatments should be tested to determine specific protocols that increase light reaching pondberry plants. Further, individual stems can be severed at the ground line to induce sprouting. Stem clipping, in concert with increased light availability, has shown promise as a way to reinvigorate individual pondberry plants (Unks and others 2014). Additional research should include silvicultural treatments to control expected increases in competition from vines and other plants that will also respond to increased light availability. Ideally, study sites should be located along an elevational gradient. Low-elevation sites will flood more frequently than relatively higher

elevation sites; therefore, natural hydroperiod effects on pondberry and potentially competing plants can be observed. Field experiments should be located on public lands to reduce the risk of changes in land ownership or landowner management objectives during ongoing long-term field experiments.

Pondberry field research will be difficult. First, colonies are scattered and small in area. This will largely negate the use of standard experimental research designs with proper replication on individual populations, such as split-plot designs to test various midstory competition control treatments. The risk of treatment effects from one plot affecting adjacent plots will be high, as little space will be available for effective buffers between treatments. A completely randomized design or a paired-plot design may be necessary, with individual pondberry colonies within populations used as experimental units, but this will require a large number of colonies to reduce the variability within colonies. Second, individual pondberry colonies will have high variability, as individual stems may be male or female, of different ages, and of seedling or ramet origin. High individual stem variability will require a large number of stems in each experimental unit. Planted pondberry seedlings or stecklings can be used in concert with research on extant plants to reduce variability in addition to testing artificial regeneration techniques (see Devall and others (2004) and Smith (2003) for examples of planting pondberry). Third, pondberry is an endangered species. Field experiments will require special use permits from the U.S. Fish and Wildlife Service. Further, experiments on public land, such as on the Delta National Forest, may require additional documentation, including possible National Environmental Policy Act (NEPA) approval. A considerable amount of time and planning will be necessary to obtain all approvals before field research can begin.

CONCLUSIONS

Pondberry is a disturbance-dependent species that requires somewhere between 40 percent and 70 percent of full sunlight for best stem length growth. In the MAV, pondberry populations are often found in the understory of bottomland hardwood forests–presumably relic populations reflecting past disturbance. Light levels in these populations can be as low as 5 percent of full sunlight, resulting in poorly-formed plants.

Research has shown that pondberry can respond to increases in light availability through greater stem length growth. Silvicultural treatments, such as mechanical or chemical control of midstory forest canopies, may be an effective tool to increase light available to extant pondberry populations. Further treatments will likely be necessary to control vegetation that will also respond to increased light availability and compete with pondberry, especially if flood waters are ineffective at reducing this competition.

Field research is needed to confirm that application of treatments suggested in this paper can be effective silvicultural tools to increase pondberry stem length growth. Results from this research could be used to develop a pondberry population maintenance program that will require initial treatments to release extant pondberry with periodic treatments to control new competing understory vegetation. This program will require monitoring of extant pondberry populations and discovery of new pondberry populations to coordinate timely application of competition control treatments and ensure long-term population viability.

LITERATURE CITED

Aleric, K.M.; Kirkman, L.K. 2005a. Growth and photosynthetic response of the federally endangered shrub, *Lindera melissifolia* (Lauraceae), to varied light environments. American Journal of Botany. 92: 682-689.

Aleric, K.M.; Kirkman, L.K. 2005b. Seed germination observations of the federally listed *Lindera melissifolia*. Castanea. 70: 157-160.

Banker, B.; Goetz, E.J. 1989. Biological evaluation for pondberry (*Lindera melissifolia*) on the Delta National Forest. Jackson, MS: U.S. Fish and Wildlife Service. 22 p.

Beckley, A.C. 2012. A habitat characterization and suitability model for the endangered wetland plant *Lindera melissifolia* in the southeastern Coastal Plain. Charleston, SC: College of Charleston. 134 p. M.S. thesis.

Beckley, A.C.; Gramling, J.M. 2013. Description and classification of *Lindera melissifolia* habitat in the southeastern Coastal Plain. Castanea. 78: 277-289.

Connor, K.; Schaefer, G.; Donahoo, J. [and others]. 2007. Development, fatty acid composition, and storage of drupes and seeds from the endangered pondberry (*Lindera melissifolia*). Biological Conservation. 137: 489-496.

Connor, K.F.; Schaefer, G.M.; Donahoo, J.B. [and others]. 2012. *Lindera melissifolia* seed bank study in a Lower Mississippi Alluvial Valley bottomland forest. Seed Technology. 34: 163-172.

Cunningham, K.L.; Peairs, S.E.; Ezell, A.W. [and others]. 2011. Understory light conditions associated with partial overstory removal and midstory/understory control applications in a bottomland hardwood forest. Forests. 2: 984-992.

Currie, R.R. 1986. Endangered and threatened wildlife and plants; determination of endangered status for *Lindera melissifolia.* Federal Register. 51(147): 27495-27500.

DeLay, L.; O'Conner, R.; Ryan, J.; Currie, R.R. 1993. Recovery plan for pondberry (*Lindera melissifolia*). Atlanta, GA: U.S. Department of the Interior Fish and Wildlife Service. 56 p.

Devall, M.; Schiff, N.; Boyette, D. 2001. Ecology and reproductive biology of the endangered pondberry, *Lindera melissifolia* (Walt) Blume. Natural Areas Journal. 21: 250-258.

Devall, M.S.; Schiff, N.M.; Skojac, S.A. 2004. Outplanting of the endangered pondberry. In: Connor, K.F., ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Science, Southern Research Station: 574-577.

Echt, C.S.; Demeer, D.; Kubisiak, T.; Nelson, C.D. 2006. Microsatellites for *Lindera* species. Molecular Ecology Notes. 6: 1171-1173.

Echt, C.S.; Demeer, D.; Gustafson, D. 2011. Patterns of differentiation among endangered pondberry populations. Conservation Genetics. 12: 1015-1026.

Fraedrich, S.W.; Harrington, T.C.; Bates, C.A. [and others]. 2011. Susceptibility to laurel wilt and disease incidence in two rare plant species, pondberry and pondspice. Plant Disease. 95: 1056-1062.

Glitzenstein, J. 2007. Native Plant Society and Forest Service re-invigorate endangered pondberry at Honey Hill. The Journal of the South Carolina Native Plant Society. Fall 2007: 1, 4-5.

Godt, M.J.W.; Hamrick, J.L. 1996. Allozyme diversity in the endangered shrub *Lindera melissifolia* (Lauraceae) and its widespread congener *Lindera benzoin*. Canadian Journal of Forest Research. 26: 2080-2087.

Gustafson, D.J.; Giunta, A.P.; Echt, C.S. 2013. Extensive clonal growth and biased sex ratios of an endangered dioecious shrub, *Lindera melissifolia* (Walt) Blume (Lauraceae). Journal of the Torrey Botanical Society. 140: 133-144.

Guttery, M.R.; Ezell, A.W.; Hodges, J.D.; Londo, A.J. 2011. Obtaining adequate light for planted bottomland oak (*Quercus* spp.) seedlings: a case study utilizing midstory control. Journal of Sustainable Forestry. 30: 187-203.

Hawkins, T.S.; Schiff, N.M.; Gardiner, E.S.; [and others]. 2007. Micropropagation of the endangered shrub pondberry (*Lindera melissifolia* [Walt.] Blume). HortScience. 42: 407-409.

Hawkins, T.S.; Schiff, N.M.; Leininger, T.D. [and others]. 2009a. Growth and intraspecific competitive abilities of the dioecious *Lindera melissifolia* (Lauraceae) in varied flooding regimes. Journal of the Torrey Botanical Society. 136: 91-101.

Hawkins, T.S.; Skojac, D.S.; Lockhart, B.R. [and others]. 2009b. Bottomland forests in the Lower Mississippi Alluvial Valley associated with the endangered *Lindera melissifolia*. Castanea. 74: 105-113.

Hawkins. T.S.; Skojac, D.A.; Schiff, N.M.; Leininger, T.D. 2010. Floristic composition and potential competitors in *Lindera melissifolia* (Lauraceae) colonies in Mississippi with reference to hydrologic regime. Journal of the Botanical Resources Institute of Texas. 4: 381-390.

Hawkins, T.S.; Walck, J.L.; Hidayati, S.N. 2011. Seed ecology of *Lindera melissifolia* (Lauraceae) as it relates to rarity of this species. Journal of the Torrey Botanical Society. 138: 298-307.

Jenkins, M.W.; Chambers, J.L. 1989. Understory light levels in mature hardwood stands after partial overstory removal. Forest Ecology and Management. 26: 247-256.

Klomps, V.L. 1980. The status of *Lindera melissifolium* (Walt.) Blume, pondberry, in Missouri. Transactions of the Missouri Academy of Science. 14: 61-66.

Lhotka, J.M.; Loewenstein, E.F. 2006. Indirect measures for characterizing light along a gradient of mixed-hardwood riparian forest canopy structures. Forest Ecology and Management. 226: 310-318.

Lockhart, B.R.; Hodges, J.D.; Gardiner, E.S. 2000. Response of advance cherrybark oak reproduction to midstory removal and shoot clipping. Southern Journal of Applied Forestry. 24: 45-50.

Lockhart, B.R.; Wishard, R.J.; Ezell, A.W. [and others]. 2010. Oak regeneration following complete and partial harvesting in the Mississippi Bluff Hills: preliminary results. In: Stanturf, J.A., ed. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-121. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 439-445.

Lockhart, B.R.; Gardiner, E.S.; Stautz, T.; Leininger, T.D. 2012. Development and plasticity of endangered shrub *Lindera melissifolia* (Lauraceae) seedlings under contrasting light regimes. Plant Species Biology. 27: 30-45.

Lockhart, B.R.; Gardiner, E.S.; Leininger, T.D. [and others]. 2013. *Lindera melissifolia* responses to flood durations and light regimes suggest strategies for recovery and conservation. Plant Ecology. 214: 893-905.

Lockhart, B.R.; Gardiner, E.S.; Leininger, T.D. 2015. Initial response of pondberry released from heavy shade. In: Holley, A.G.; Connor, K.F.; Haywood, J.D., eds. Proceedings of the 17th biennial southern silvicultural research conference. e– Gen. Tech. Rep. SRS-203. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 218-223.

Priest, J.D.; Wright, R. 1991. Groundwater hydrology of a population of *Lindera melissifolia* in Arkansas. Proceedings of the Arkansas Academy of Science. 45: 84-87.

Richardson, D.J.; Wright, R.D.; Walker, S. 1990. Sexual dimorphism and intersexual differences in resource allocations of a dioecious shrub, *Lindera melissifolia* (Walt.) Blume. Proceedings of the Arkansas Academy of Science. 44: 100-103.

Smith, III, C.G.; Hamel, P.B.; Devall, M.S.; Schiff, N.M. 2004. Hermit thrush is the first observed dispersal agent for pondberry (*Lindera melissifolia*). Castanea. 69: 1-8.

Smith, T.E. 2003. Observations on the experimental planting of *Lindera melissifolia* (Walter) Blume in southeastern Missouri after ten years. Castanea. 68: 75-80.

Taylor, O.S. 2008. Reproductive ecology of the endangered shrub (Lauraceae) in Pondberry Bay, North Carolina. Boone, NC: Appalachian State University. 51 p. M.S. thesis.

Tucker, G.E. 1974. The vascular plant family Lauraceae in Arkansas. Proceedings of the Arkansas Academy of Science. 28: 74-75.

Unks, R.R. 2011. Environmental controls of reproduction and early growth of *Lindera melissifolia* (Lauraceae). Raleigh, NC: North Carolina State University. 40 p. M.S. thesis.

Unks, R.R.; Shear, T.H.; Krings, A.; Braham, R.R. 2014. Environmental controls of reproduction and early growth of *Lindera melissifolia* (Lauraceae). Castanea. 79: 266-277.

Wiggers, M.S. 2014. Pondberry (*Lindera melissifolia*) 5-year review: summary and evaluation. Jackson, MS: U.S. Department of the Interior Fish and Wildlife Service, Southeast Region, Mississippi Field Office. 42p.

Wright, R.D. 1989. Reproduction of *Lindera melissifolia* in Arkansas. Proceedings of the Arkansas Academy of Science. 43: 69-70.

Wright, R.D. 1990a. Photosynthetic competence of an endangered shrub, *Lindera melissifolia*. Proceedings of the Arkansas Academy of Science. 44: 118-120.

Wright, R.D. 1990b. Species biology of *Lindera melissifolia* (Walt.) Blume in northeast Arkansas. New York State Museum Bulletin. 471: 176-179.

Wright, R.D. 1994. Sex ratio and success, an assessment of *Lindera melissifolia* in Arkansas. Proceedings of the Arkansas Academy of Science. 48: 230-233.

EASTERN HEMLOCK FOUND IN MACON COUNTY, ALABAMA

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Abstract—Alabama is currently the southern limit of the range of eastern hemlock [*Tsuga canadensis* (L.) Carriere]. In 2012, several well preserved stumps were excavated from a farm located in Notasulga, Alabama (32 ° 33" 6' N; 85° 40" 22' W). Even though they were buried in a saturated soil for approximately 1,500 years, the stumps were remarkably well preserved. The low-density wood from one stump was identified as hemlock (*Tsuga* spp.) due to presence of piceoid, cross-field pitting. We assume the excavated stump was eastern hemlock. Uncorrected carbon dating of one sample was 1580 radiocarbon years (± 25 years) before 1950. This equates to ca. 418 to 541 AD. The diameter of the largest stump excavated was approximately 50 cm. The age of a 26-cm stump was approximately 74 years. The location of this site is about 1 degree further south in latitude than the southernmost stand in Jefferson County (approximately 175 km southeast). To date, the Macon County location is likely the most southern documented Coastal Plain site where a hemlock stump has been excavated.

INTRODUCTION

Climate affects the migration of glaciers and forests. When the climate cools, North American tree species tend to migrate to warmer environments in the South (note: individual trees do not migrate). In contrast, when the climate warms and glaciers retreat, some species will migrate north and in some regions, they grow into higher latitudes (Bonnicksen 2000). Fossil findings from the Late Eocene period indicate that hemlock ancestors were growing in western Alabama over 34 million years ago (Leopold and Pakiser 1964; Frederiksen 1980). About 10,000 years ago, hemlock was growing near Birmingham, Alabama (Delcourt and others 1983).

When the environment becomes unfavorable for survival of eastern hemlock [*Tsuga canadensis* (L.) Carriere], disjunct populations can form. Disjunct populations in the 21st century occur in Michigan, Ohio, Indiana, Kentucky, Tennessee, Alabama and several Atlantic coast states (Hart 2008). Currently, the most southern extant population is in Jefferson County, Alabama. A disjunct population may have existed further south in Macon County, Alabama around 500 A.D.

MATERIALS AND METHODS

A small farm pond at Sandy Creek Stables in Notasulga, Alabama was expanded using soil moving equipment. During the expansion, several buried logs and stumps were uncovered. Some of the logs were southern pines (*Pinus* spp.) while others were not readily identifiable as to genus. The size of excavated stumps ranged in diameter from 50 cm (fig. 1) to 26 cm to less than 11 cm. The soil covering the stumps appeared to originate from ancient beach sand from the Tuscaloosa Group [perhaps from the Cretaceous, Cenomanian-Turonian Stage (Cahoon 1972)].

The wood was identified microscopically as *Tsuga* due to several factors including the presence of piceoid, cross-field pitting and no distinction between sapwood and heartwood (Kukachka 1960). The low-density wood had no particular odor, there was an abrupt to semi-abrupt transition from earlywood to latewood and the ray tracheids were narrow. The wood was not identified to species, but it was assumed to be *Tsuga canadensis.* One excavated stump had a diameter of 26 cm and an age of perhaps 74 years.

A sample from the largest stump was sent to the University of Georgia (Center for Applied Isotope Studies) for radiocarbon dating (Ramsey 2008). The results (UGAMS# 12498) indicated an uncorrected ^{14}C age of 1580 years B.P. (± 25) [note: B.P. = before 1950]. The corrected result indicates a date of 480 A.D. (± 62 yr). In theory, there is a 95 percent chance that the tree was alive from 418 A.D. to 541 A.D. Additional information about the radiocarbon age vs. calendar date is provided in figure 2.

A cross-section of a 26-cm diameter stump was sent to the University of Tennessee at Knoxville. The surface

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Figure 1—Excavation of an eastern hemlock stump in Macon County, AL.

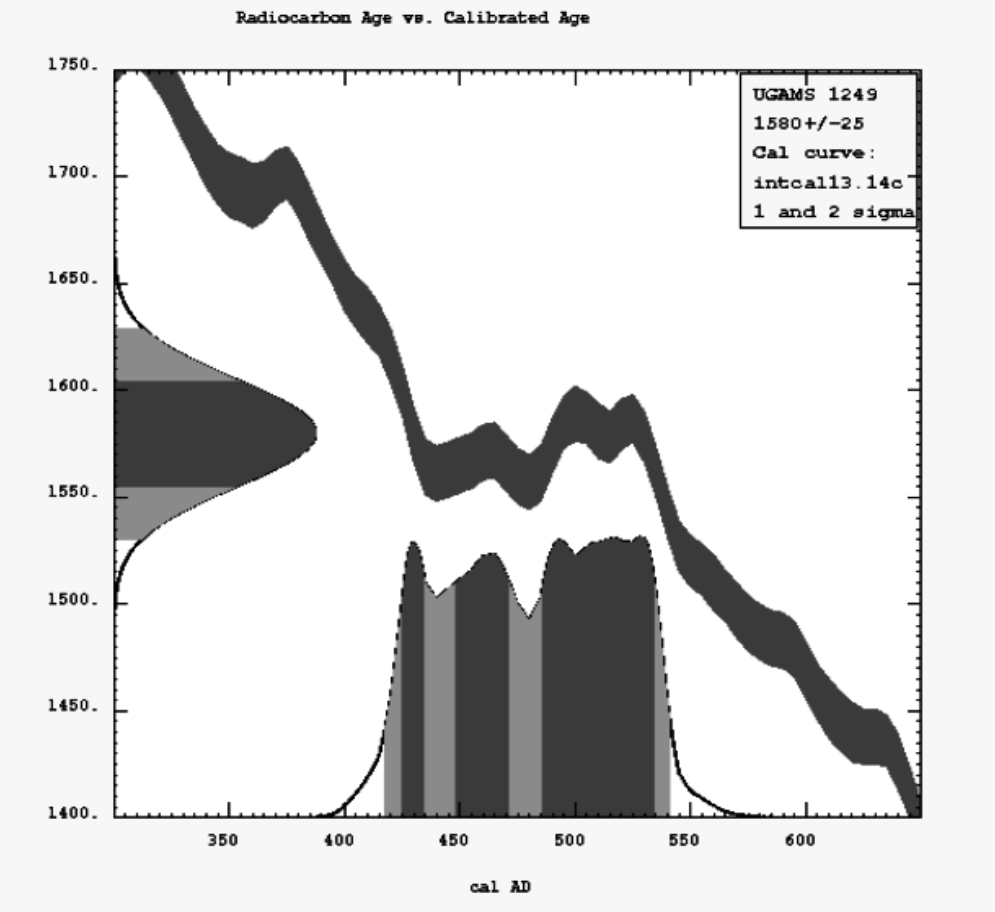


Figure 2—A calibration graph for converting radiocarbon date (Y-axis) to calendar date (X-axis).

was sanded (belt-sander) using a progression of grit sizes; beginning with ANSI 80-grit (177–210 µm) and ending with ANSI 400-grit (20.6–23.6 µm) (Orvis and Grissino-Mayer 2002). The surface was sanded until all cellular features of the rings were clearly visible under standard 7–10x magnification. Rings were annotated using the standard dot notation used in dendrochronology (Stokes and Smiley 1996). Ring widths were measured to 0.001 mm using a Velmex® moving stage micrometer interfaced with Measure J2X software (Speer 2010). These measurements were imported into ARSTAN® for Windows software to evaluate the growth trends. High resolution scans were obtained using an Epson® 10000 XL scanner.

DISCUSSION

Tree Growth

The 26-cm diameter stump contained 74 measurable tree rings (table 1, fig. 3). During the first two decades (figs. 4A, 4B), ring boundaries did not have a distinct sharp edge in the spring (between latewood of the previous ring and earlywood of the current ring). A sharp edge is normally associated with conifers that grow in temperate regions. This suggests that when young, this tree did not cease diameter growth in the winter (although indistinct ring boundaries become more distinct with increasing age). We believe this type of growth pattern (indistinct ring boundaries grading into distinct ring boundaries) is a reflection of physiological

Table 1—Seventy four tree-ring widths for one eastern hemlock sample in 0.001 mm format (e.g. "5008" = "5.008 mm"). Each row represents a decade, i.e. first decade contain rings 1-9, second decade contain rings 10-19 etc

Ring number	0	1	2	3	4	5	6	7	8	9
1-9		5008	4519	3473	2164	2000	2176	4124	3175	1993
10-19	2340	3756	5895	5131	4829	3618	3177	2994	5245	4346
20-29	3687	2033	1807	1792	3180	3905	2894	2145	1945	3516
30-39	2409	2077	1860	2105	1557	1704	1381	1506	909	1438
40-49	1286	1355	1179	883	609	508	431	443	257	97
50-59	188	162	216	478	215	359	128	95	597	790
60-69	661	541	677	816	1015	772	472	188	167	228
70-74	144	137	143	138	141					

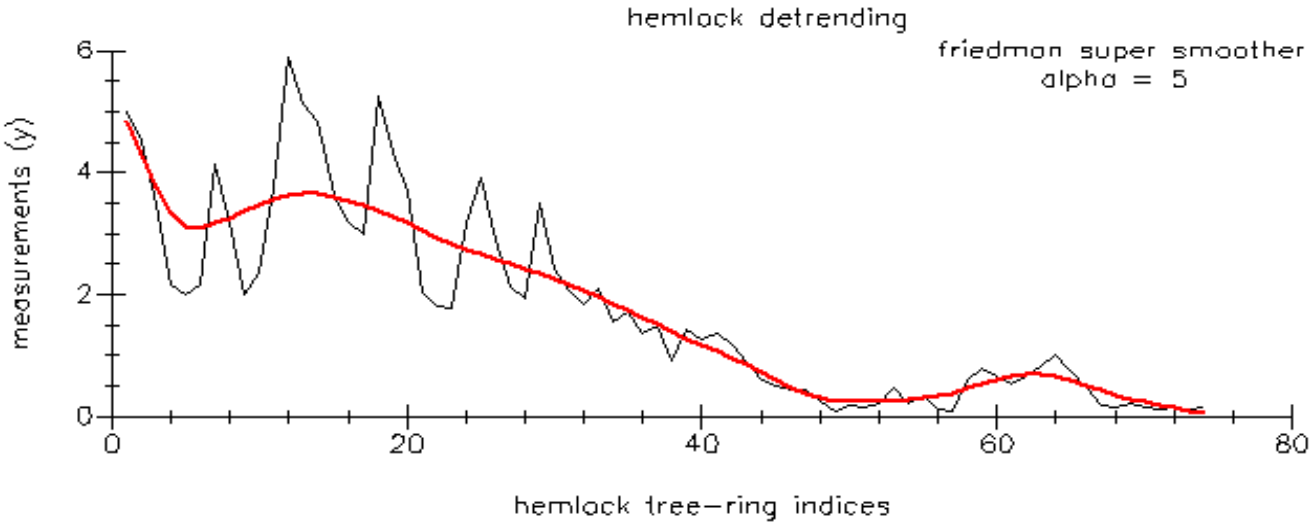


Figure 3—Measurements (in millimeters) of the 74 tree-ring widths for the eastern hemlock sample, showing the growth trend using a Friedman supersmoother variable span smoothing algorithm available in the detrending software ARSTAN.

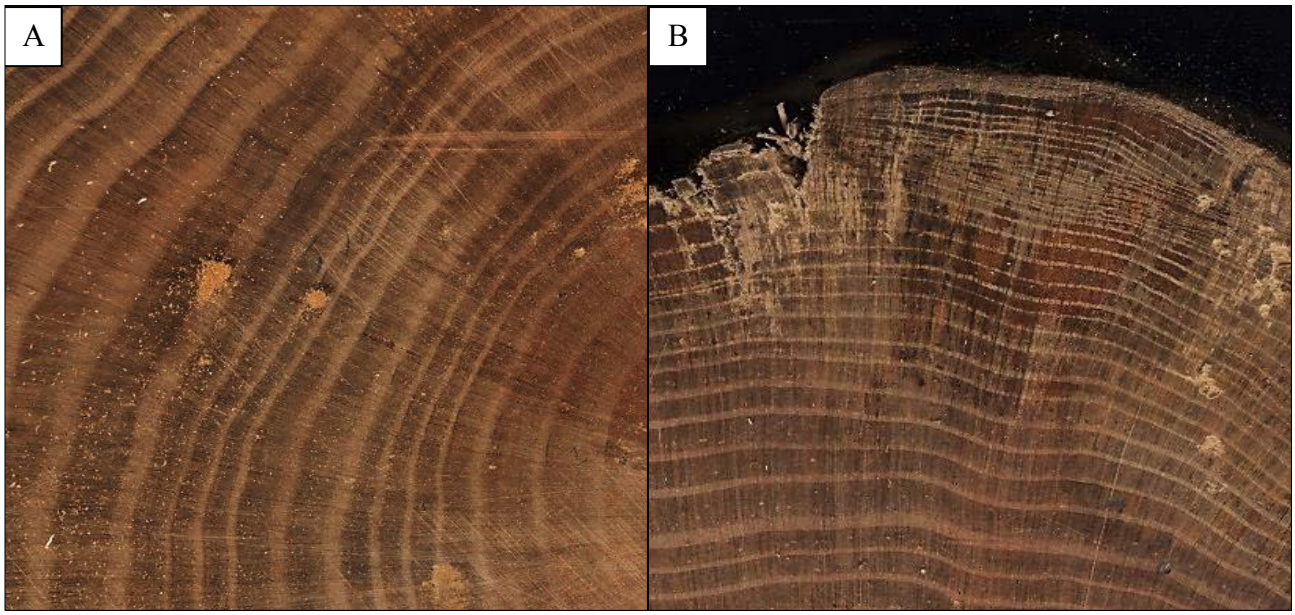


Figure 4—Rings formed in the earlier years of the eastern hemlock (A) did not have the distinct ring boundaries as rings that formed later in the tree's lifetime (B). Note also how the percentage of latewood increases from inner rings (A) to outer rings (B).

aging. When young, the tree was more vigorous and many conifers can produce photosynthates during the winter (Lundmark and others 1988; Hadley 2000) and can increase diameter. As the tree aged, the added biomass both vertically and laterally was more sensitive to seasonal changes, causing the tree to clearly shut down growth and form distinct boundaries.

Another notable feature was the large percentage of the ring dedicated to latewood. The earlywood (lighter colored) band is narrow in relation to the expansive latewood zone of the growth ring. Curiously, towards the outermost rings, the earlywood zone was demarcated by only a few cells in the radial file of cells before the thicker walled cells of the latewood formed (fig. 4A). A change in some aspect of environmental conditions is considered necessary for the initiation of latewood formation in conifers, and this can be both temperature (Begum and others 2012) or precipitation/drought related (Vaganov and others 2006). In eastern conifers, thicker-walled latewood cells can form when soil moisture that was recharged over the dormant period months becomes depleted during the early months of the growth season (Whitmore and Zahner 1966). The formation of latewood often occurs in mid-summer (i.e. July) for various eastern conifers, such as *Abies balsamea* in Canada (Deslauriers and others 2003) and *Pinus elliottii* in southern Florida (Harley and others 2012). For *Tsuga*, the percentage of latewood generally increases with increasing age, especially in the rings towards the outside of the tree bole (Edlin 1965). This pattern was clearly visible in our sample (fig. 5).

A severe suppression of growth was noted from rings 48 to ca. ring 60 (fig. 4B). This suppressed and erratic growth occurred along at least half the circumference of the sample wherever these rings were present. Growth recovered after this period of suppressed growth returning to ring widths more characteristic of growth prior to the disturbance (fig. 4B). Many environmental factors can cause a tree to undergo suppressed growth. However, it is unlikely that a decade-long drought was the cause because the suppression was not uniformly distributed for each affected ring. It is more likely that a local (or stand-wide disturbance) slowed growth rate during this decade. Such disturbances could include (1) a treefall that damaged the hemlock crown, trunk, and/or root system (Hart and Grissino-Mayer 2009); (2) a lightning strike that damaged the crown and trunk but did not reach a lethal temperature (Palik and Pederson 1996); (3) a wildfire that caused damage to the crown, trunk, and/or root system (Rogers 1978); (4) insect herbivory from a number of known insect pests on eastern hemlock [e.g. *Diaspidide* scales (McClure and Fergione 1977) and *Lambdina fiscellaria* Guenée (Bhiry and Filion 1996)]; (5) damage to the crown (e.g. lean) caused by hurricane-force winds (Peterson 2000); or (6) damage to the crown caused by biomass loss from excessive branch loading during an ice storm (Lafon and Kutac 2003). Any one of these disturbances could have caused the suppression in growth rates. As

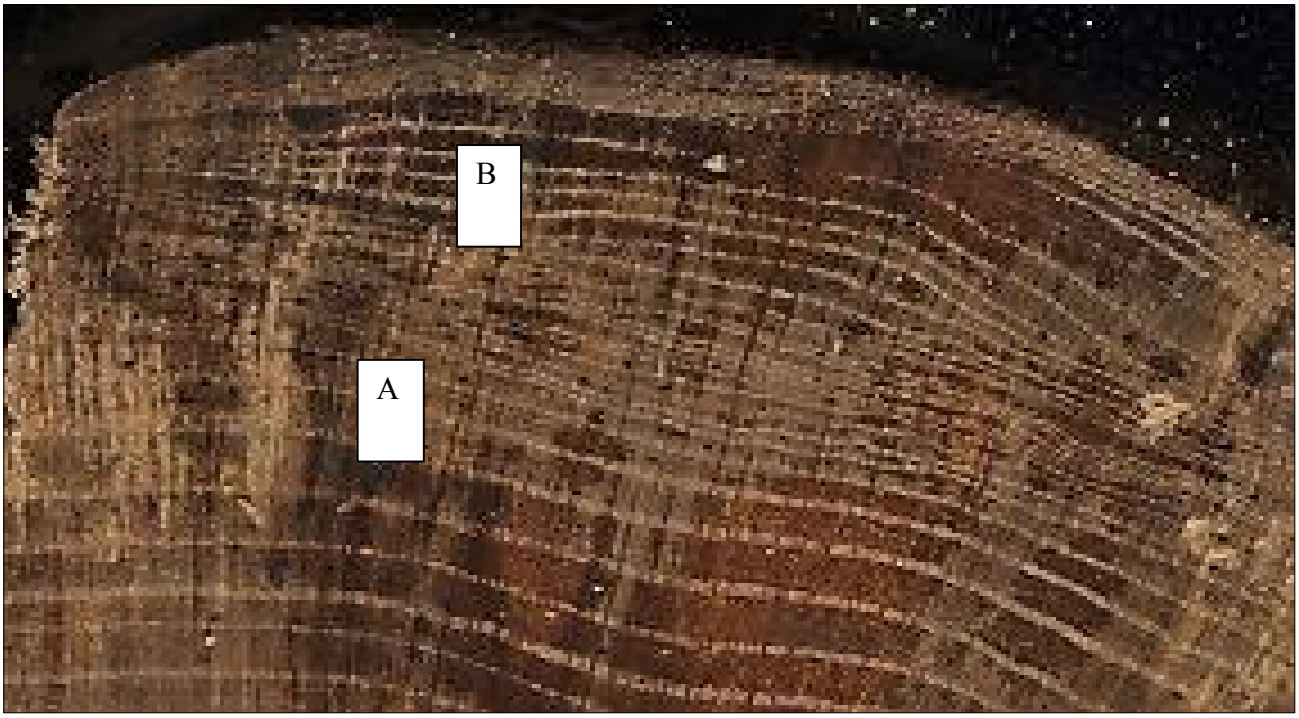


Figure 5—Close-up of the outermost rings that clearly show (A) the large percentage of latewood in each ring and (B) a change in growth rate to very narrow rings, suggesting a disturbance affected the tree's growth rate.

the tree recovered from this biomass loss or damage, diameter growth eventually recovered.

Metrics of tree growth show a mean ring width of 1.76 mm for the 74 measured rings and a mean sensitivity of 0.31 mm (mean sensitivity is a measure of ring width variability) (Speer 2010). The International Tree-Ring Data Bank houses 51 site chronologies for eastern hemlock and the mean measurement for several hundred eastern hemlock trees in these data sets is 0.75 mm with an average mean sensitivity of 0.24 mm, which indicates this sample has rings that are wider than average for eastern hemlock. Ring widths are also more variable from year to year, suggesting sensitivity to year to year weather variations. The first-order autocorrelation for these 51 data sets is 0.81 while our sample has a comparable value of 0.89. In general, this eastern hemlock was likely growing in a southern environment that was conducive to enhanced sensitivity to climate fluctuations when compared to eastern hemlock trees growing in more northern latitudes today.

Species Migration

When the climate cooled during the last ice age, trees in parts of the North were crushed due to the advancement of glaciers. Eastern hemlock south of the glaciers survived and the cooler climate allowed regeneration to occur further south. However, the extent of hemlock in the eastern United States during that time is not clear. From 19,000 to 17,000 radiocarbon years B.P., there was no hemlock pollen found at various sites in the eastern United States (Delcourt and Delcourt 1987). This does not mean that eastern hemlock did not exist, it just means that sampling intensity for these dates was low. Based on ice cores from Greenland (Dansgaard and others 1969), temperatures during glacial times were likely cool enough to allow hemlock to survive and grow. Pollen samples suggest that *Fagus* spp. were dominant in Alabama about 12,000 years ago (Williams and others 2004, Delcourt and Delcourt 1987) and hemlock was growing near Birmingham perhaps 10,000 years ago (Delcourt and others 1983). Pollen recovered from one location in Pike County, Alabama (i.e. south of Macon County) suggests hemlock did not occur that far south (Delcourt 1980).

The discovery of buried hemlock stumps in Macon County raises several questions. First, when did hemlock first arrive in Macon County, Alabama? We do not know. Fossils from the Late Eocene period indicate that hemlock ancestors were growing in western Alabama over 34 million years ago (Frederiksen 1980). About 36,000 years B.P., fossil hemlock spores were produced near the Atlantic Ocean along the border between Florida and Georgia (Pirkle and others 2013). Soil cores near Birmingham, Alabama indicate hemlock pollen was produced near Birmingham, Alabama about 11,000 radiocarbon years B.P. (Delcourt and others 1983). One might assume that hemlock arrived in Macon County after that time. However, surveys

in Macon County found no hemlock pollen in Macon County (from either 250 or 5,300 radiocarbon years B.P.) (Markewich and Christopher 1982). This does not mean that hemlock did not exist in Macon County, it just means that no hemlock pollen was recovered from the sampled strata.

Why did hemlock become extinct in Macon County? We do not know. If a disjunct population of hemlock was alive in Macon County in 500 A.D., then something increased the mortality rate. One possibility is the agent responsible for the hemlock decline about 5,000 years ago (Delcourt and Delcourt 1987; Filion and Quinty 1993) finally reached this region. Another possibility is that natives increased the use of fire. If a wildfire (or human set fire) occurred during a drought, this might have killed young, fire-sensitive seedlings. It also may be that hemlock succumbed to higher summer temperatures during the Medieveal Warm Period (fig. 6). Alternatively, an extended drought may have contributed to the decline (Haas and McAndrews 2000). The climate of Macon County during the 20th Century was almost as moist as that in northern Alabama, but the average temperatures were higher (table 2). It is believed that climate does affect the growth of hemlock in Alabama (Hart and others 2010).

Low-severity fires can kill seedlings and sapling hemlocks. In some cases, 60 percent of mature trees died or were severely injured by a fire (Swan 1970). Although fires near streams will occasionally produce fire scars on hemlock (Lafon and others 2010), they are not typically found on hemlock. Since the roots were underground, no fire scars were noted on the outside of any of the excavated stumps.

Without additional investigations, we may never know why these stumps in this sandy soil were buried. We

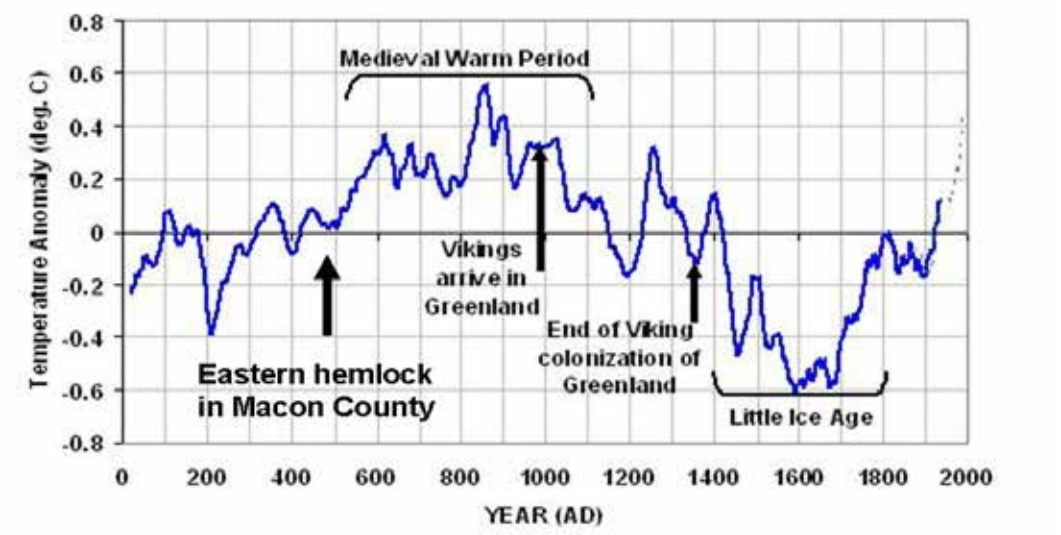


Figure 6—A global temperature reconstruction based on non-tree ring proxies (Loehle and McCulloch 2008). Data for this graph is online at www.econ.ohio-state.edu/jhm/agw/loehle/ Hemlock was growing in Macon County, Alabama prior to the Medieval Warm period.

Table 2—Average climatic conditions (1901-2000) for three NOAA climate divisions in the Southeastern United States. In the past, hemlock was present in Climate Division #9 in Georgia (Pirkle and others 2013) and in Climate Division #6 in Alabama (which includes Macon County)

State	NOAA Climate Division	Maximum July temperature (°C)	Average Maximum July temperature (°C)	Average Annual Temperature (°C)	Average Annual Precipitation (mm)	July Palmer Drought Severity Index	Hemlock present in the 21th Century
Alabama	Appalachian Mountain (#2)	36.0	32.1	15.7	1404	-0.14	Yes
Alabama	Prairie (#6)	36.5	33.1	17.8	1347	-0.22	No
Georgia	Southeast (#9)	35.9	33.2	19.1	1249	-0.04	No

know that horizontal logs in creeks can be buried due to sedimentation (Cahoon 1972). There have been documented cases where standing trees have been buried when earthquakes cause "sand blows." These sand "volcanoes" are formed when liquefaction of water-saturated sands results in rapid sand accumulation at the surface (Tuttle 2010; 2011). Roots of the hemlock stumps were growing in white sandy soil (originating from an ancient beach over 90 million years ago). When water-saturated, this soil would be a prime candidate for a sand blow. Although the upper portions of the trees were removed during the excavations, the remaining stumps appeared to be vertical with no lean. Obviously, the hemlock trees were either (A) growing on top of the ancient beach sand and then sank when the sand became "liquefied" or (2) were growing on a more recent soil type and then the trees were buried in sand from a "sand blow." Further investigations at this site by geologists might solve this puzzle.

CONCLUSION

Paleo-dominance range maps for hemlock for 500 years A.D. are based on pollen samples. These range maps do not include Macon County, Alabama (Delcourt and Delcourt 1987; p. 278). For that date, it was assumed that the southern limit for Tsuga was approximately 34 ° N. However, a buried stump from Macon County suggests that hemlock was growing at 32 ° N at that time. Updated paleo-dominance maps for hemlock for 500 years A.D. should reflect this knowledge. More intensive pollen sampling would likely support this finding.

ACKNOWLEDGMENTS

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LITERATURE CITED

Begum, S.; Nakaba, S.; Yamagishi, Y. [and others]. 2012. A rapid decrease in temperature induces latewood formation in artificially reactivated cambium of conifer stems. Annals of Botany. 110 (4): 875-885.

Bhiry, N.; Filion, L. 1996. Mid-Holocene hemlock decline in eastern North America linked with phytophagous insect activity. Quaternary Research. 45(3): 312-320.

Bonnicksen, T.M. 2000. America's ancient forests: from the ice age to the age of discovery. New York: John Wiley & Sons. 594 p.

Cahoon, E.J. 1972. Paraphyllanthoxylon alabamense – a new species of fossil dicotyledonous wood. American Journal of Botany. 59(1): 5-11.

Dansgaard, W.; Johnsen, S.J.; Møller, J.; Langway, C.C. 1969. One thousand centuries of climate record from Camp Century on the Greenland ice sheet. Science. 166(3903): 377-380.

Delcourt, P.A. 1980. Goshen Springs: late-Quaternary vegetation record for southern Alabama. Ecology. 61: 371-386.

Delcourt, P.A.; Delcourt, H. R. 1987. Long-term forest dynamics of the temperate zone. New York: Springer-Verlag. 439 p.

Delcourt, H.R.; Delcourt, P.A.; Spiker, E.C. 1983. A 12,000-year record of forest history from Cahaba Pond, St. Clair County, Alabama. Ecology. 64(4): 874-887.

Deslauriers, A.; Morin, H.; Begin, Y. 2003. Cellular phenology of annual ring formation of Abies balsamea in the Quebec boreal forest (Canada). Canadian Journal of Forest Research. 33(2): 190-200.

Edlin, H.L. 1965. The summerwood: springwood ratio in conifers: how it arises, varies, and affects end use for timber, board, and paper. Forestry. 38(1): 91-112.

Filion, L.; Quinty, F. 1993. Macrofossil and tree-ring evidence for a long-term forest succession and mid-holocene hemlock decline. Quaternary Research. 40: 89-97.

Frederiksen, N.O. 1980. Sphromorphs from the Jackson Group (Upper Eocene) and adjacent strata of Mississippi and western Alabama. Geological Survey Professional Paper 1522. Washington, DC: U.S Department of the Interior Geological Survey. 118 p.

Haas, J.N.; McAndrews, J.H. 2000. The summer drought related hemlock (*Tsuga Canadensis*) decline in eastern North America 5,700 to 5,100 years ago. In: McManus, K.A.; Shields, K.S.; Souto, D.R., eds. Symposium on sustainable management of hemlock ecosystems in eastern North America. Gen. Tech. Rep. NE-267. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Southern Research Station: 81-88.

Hadley, J.L. 2000. Effect of daily minimum temperature on photosynthesis in eastern hemlock (*Tsuga canadensis* L.) in autumn and winter. Arctic, Antarctic, and Alpine Research. 32(4): 368-374.

Harley, G.L.; Grissino-Mayer, H.D.; Franklin, J.A. [and others]. 2012. Cambial activity of *Pinus elliottii* var. densa reveals influence of seasonal insolation on growth dynamics in the Florida Keys. Trees-Structure and Function. 26: 1449-1459.

Hart. J.L. 2008. Eastern hemlock decline and persistence of disjunct populations near its southern boundary. Journal of Alabama Academy of Science. 79(3-4): 174-180.

Hart, J.L.; Grissino-Mayer. H.D. 2009. Gap-scale disturbance processes in secondary hardwood stands on the Cumberland Plateau, Tennessee, USA. Plant Ecology. 201(1): 131-146.

Hart, J.L.; van de Gevel, S. L.; Sakulich, J.; Grissino-Mayer, H.D. 2010. Influence of climate and disturbance on the growth of Tsuga canadensis at its southern limit in eastern North America. Trees. 24: 521-633.

Kukachka, K. 1960. Identification of coniferous woods. Tappi. 43(11): 888-896.

Lafon, C.W.; Grissino-Mayer, H.D.; Horn, S.P.; Klein, R.N. 2010. Fire regimes of the southern Appalachian mountains: temporal and spatial variability over multiple scales and implications for ecosystem management. U.S. Joint Fire Science Program Research Project Reports. Paper 84. http://cigitalcommons.unl.edu/jfspresearch/84 [Date accessed: April 29, 2015].

Lafon, C.W.; Kutac, M.J. 2003. Effects of ice storms, southern pine beetle infestation, and fire on Table Mountain pine forests of southwestern Virginia. Physical Geography. 24(6): 502-519.

Leopold, E.B.; Pakiser, H.M. 1964. A preliminary report on the pollen and spores of the pre-Selma Upper Cretaceous strata of western Alabama. Geological Survey Bulletin 1160-E. Washington, DC: U.S. Department of the Interior Geological Survey: 71-95.

Loehle, C.; McCulloch, J.H. 2008. Correction to: a 2000 global temperature reconstruction based on non-tree ring proxies. Energy and Environment. 19(1): 93-100.

Lundmark, T.; Hedén, J.; Hällgren, J.E. 1988. Recovery from winter depression of photosynthesis in pine and spruce. Trees. 2: 110-114.

Markewich, H.W.; Christopher, R.A. 1982. Pleistocene (?) and Holocene fluvial history of Uphapee Creek, Macon County, Alabama. Geological Survey Bulletin 1522. Washington, DC: U.S Department of the Interior Geological Survey: 1-16.

McClure, M.S.; Fergione, M.B. 1977. *Fiorinia externa* and *Tsugaspidiotus tsugae* (Homoptera: Diaspididae): Distribution, abundance, and new hosts of two destructive scale insects of eastern hemlock in Connecticut Environmental Entomology. 6(6): 807-811.

Orvis, K.H.; Grissino-Mayer, H.D. 2002. Standardizing the reporting of abrasive papers used to surface tree-ring samples. Tree-Ring Research. 58(1/2): 47-50.

Palik, B.J.; Pederson. N. 1996. Overstory mortality and canopy disturbances in longleaf pine ecosystems. Canadian Journal of Forest Research. 26(11): 2035-2047.

Peterson, C.J. 2000. Damage and recovery of tree species after two different tornadoes in the same old growth forest: a comparison of infrequent wind disturbances. Forest Ecology and Management. 135(1-3): 237-252.

Pirkle, F.L.; Pirkle, W.A.; Rich, F.J. 2013. Heavy-mining in the Atlantic Coastal Plain and what deposit locations tell us about ancient shorelines. Journal of Coastal Research. 69(sp1): 154-175.

Ramsey, C.B. 2008. Radiocarbon dating: revolutions in understanding. Archaeometry. 50(2): 249-275.

Rogers, R.S. 1978. Forests dominated by hemlock (Tsuga canadensis): distribution as related to site and postsettlement history. Canadian Journal of Botany. 56(7): 843-854.

Speer, J.H. 2010. Fundamentals of Tree-Ring Research. Tucson, AZ: The University of Arizona Press. 333 p.

Stokes, M.A.; Smiley, T.L. 1996. An Introduction to Tree-Ring Dating. Tucson, AZ: University of Arizona Press. 73 p.

Swan, F.R. 1970. Post-fire response of four plant communities in south-central New York State. Ecology. 51(6): 1074-1082.

Tuttle, M.P. 2011. About the cover. The Earth Scientist. 27(1): 6.

Tuttle, M.P. 2010. Search for and Study of Sand Blows at Distant Sites Resulting from Prehistoric and Historic New Madrid earthquakes. Georgetown, ME: U.S. Department of the Interior Geological Survey. Final Technical Report; award 02HQGR0097. 48 p.

Vaganov, E.A.; Hughes, M.K.; Shashkin, A.V. 2006. Growth Dynamics of Conifer Tree Rings. Berlin, Germany: Springer-Verlag. Ecological Studies. 351 p.

Williams, J.W.; Shuman, B.N.; Webb, T. [and others]. 2004. Late-quaternary vegetation dynamics in North America: scaling from taxa to biomes. Ecological Monographs. 74(2): 309-334.

Whitmore, F.W.; Zahner, R. 1966. Development of the xylem ring in stems of young red pine trees. Forest Science. 12(2): 198-210.

SILVICULTURE TO RESTORE OAK WOODLANDS

Daniel C. Dey, Callie J. Schweitzer, John M. Kabrick[1]

Abstract—Variability in historic fire regimes in eastern North America resulted in an array of oak savannas, woodlands and forests that were dominant vegetation types throughout the region. In the past century, once abundant woodlands have become scarce due to conversion to agriculture, or development of forest structure in the absence of fire. Restoration of oak woodlands is a primary goal for land management agencies and conservation organizations. Although oak woodlands can be restored with a long-term regimen of prescribed burning, a combination of prescribed burning, timber harvesting and forest thinning produces the desired structure and composition more efficiently. Sustaining oak woodlands requires an occasional longer fire-free period to allow for replacement of the overstory by recruitment of trees from the reserve of oak sprouts that have accumulated in the understory. Prescribed fire is useful for sustaining oak woodlands, but it must be used judiciously to minimize timber damage and decreases in value. Integrating fire in a silvicultural prescription that uses the shelterwood regeneration method to promote competitive oak reproduction has been successfully applied in the eastern US to sustain oak forests. Restoration of oak ecosystems is possible but requires innovative combinations of traditional practices, including prescribed burning.

INTRODUCTION

Restoring oak woodlands is increasingly a management priority among state and federal natural resource agencies and conservation organizations, for example, about one-third of the Mark Twain National Forest (438,000 acres) has woodland restoration as a primary management goal (MTNF 2005). Definitions of a woodland vary among the authorities but common characteristics include: overstory dominated by oak species with 30 to >80 percent crown cover and 30 to 80 ft^2 ac^{-1} in basal area, negligible midstory woody canopy layer, sparse and patchy woody understory, and diverse ground flora dominated by grasses, forbs and sedges that is highly variable in composition and structure depending on site conditions and overstory canopy cover (MDC 2010, Nelson 2010). Oak woodlands are natural communities that historically were prominent across the landscape in the Central Hardwood Region; for example, Hanberry and others (2014b) estimated that 65 percent of the Missouri Ozark Highlands was oak woodlands. Historically frequent fire created and maintained oak woodlands for thousands of years (Delcourt and Delcourt 1991, Guyette and others 2012). But today, woodlands have been diminished by succession to forests following fire suppression and conversion to agriculture land uses.

IMPORTANCE OF WOODLANDS

Woodlands are important natural communities and landscape components that need restoring. Before European settlement, woodlands added to the landscape diversity of natural communities that formed a variable complex matrix including prairie, savanna, woodland and forest throughout the Central Hardwood Region (Transeau 1935, Schroeder 1982, Anderson and others 1999, Batek and others 1999, Nigh and Schroeder 2002, Nelson 2010). The current Central Hardwood Regional landscape is less diverse in structure, complexity, function, composition and natural community type (Schulte and others 2007, Shifley and Thompson 2011, Shifley and others 2012, Hanberry and others 2012, 2014 a,c,d). Increasing ground flora diversity is associated with decreasing canopy cover along the structural gradient from forest to savanna (Taft and others 1995, Bowles and McBride 1998, Leach and Givnish 1999, Peterson and Reich 2008). Species diversity in invertebrates, small mammals, birds and herptofauna is often higher with increasing plant species richness and heterogeneity in vegetation structure, which is highest in savanna communities (Huston 1994, Leach and Givnish 1999, Haddad and others 2001).

Wildlife species often prefer structural features of woodlands such as big brown bat (*Eptesicus fuscus*), eastern red bat (*Lasiurus borealis*), evening bat (*Nycticeius humeralis*), and tri-colored bat (*Perimyotis subflavus*), who preferred savanna and open-woodland habitats over closed canopy forests in the Missouri Ozarks (Starbuck and others 2014). Thompson and

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others (2012) observed that restored savannas and woodlands in the Ozark Highlands provided habitat for a diverse mix of grassland and canopy nesting bird species that are of high conservation concern. Blue-winged warbler (*Vermivora cyanoptera*), eastern towhee (*Pipilo erythrophthalmus*), eastern wood-pewee (*Contopus virens*), field sparrow (*Spizella pusilla*), prairie warbler (*Dendroica discolor*), and summer tanager (*Piranga rubra*) were more abundant in savannas and woodlands than in closed canopy forests. Reidy and others (2014) found that large scale savanna and woodland restoration in the Missouri Ozarks provided additional habitat for woodland generalists and early-successional species, some of which are of conservation concern. In the managed restorations, most of the focal bird species they studied responded positively to a history of fire over the past 20 years. In this largely forested landscape, fire increased the diversity of habitats available to songbirds, with corresponding increases in bird species richness, diversity, and density. Others have demonstrated the importance of having savannas and woodlands on the landscape for the conservation of rare and declining bird species that rely on disturbance and early successional habitats (Davis and others 2000, Brawn and others 2001, Brawn 2006, Grundel and Pavlovic 2007, Au and others 2008). Even bird species that are known to prefer mature, closed-canopied interior forests benefit from early successional habitat in the nearby landscape because juvenile birds forage for food and use the habitat as a refuge from predators (King and Schlossberg 2014).

Managing for disturbance adapted ecosystems and increasing biodiversity at all scales are considered key management strategies to address anticipated impacts due to climate changes (Janowiak and others 2011, Brandt and others 2014, Janowiak and others 2014). Restoration of savannas and woodlands would contribute to both of these mitigation strategies for the range of future climate scenarios predicted. Tree species common to Midwestern savannas and woodlands such as post oak (*Quercus stellata*), blackjack oak (*Quercus marilandica*) and bur oak (*Quercus macrocarpa*) are expected to be favored by predicted changes in temperature and precipitation, modifications in their seasonal patterns, and frequency of extreme weather events (Brandt and others 2014). Communities such as these, with high species richness, are considered more resilient to climate change, are better able to recover from disturbance such as drought, are less vulnerable to environmental stress and biotic threats, and are less susceptible to high-severity wildfires (Brandt and others 2014). Restoration of woodlands would increase biodiversity at the community and landscape scales by increasing the diversity of habitats (vertical and horizontal structure), local environmental conditions, and spatial heterogeneity in patch size and distribution (Hunter and Schmiegelow 2011).

STATE OF MODERN WOODLANDS

Soils and climate are suitable for tree growth in most places throughout the eastern United States. Therefore, in the absence of fire, woodlands become forests by increasing in tree density and developing a midstory tree canopy layer, and by transitioning of the ground flora to a less diverse community that is dominated by shade tolerant forbs and woody species (e.g., Nelson 2010). For example, tree density (trees ≥ 5 inches dbh) has more than doubled since the early 1800s (Hanberry and others 2012, 2014a,d) over two-thirds of the Missouri Ozark Highland region (ca. 15 million acres) (Hanberry and others 2014b). Available light in forests may range from as little as 5 percent of full sunlight on mesic to hydric, productive sites (Gardiner and Yeiser 2006, Parker and Dey 2008, Lhotka and Loewenstein 2009) to 20 percent on xeric sites (Sander 1979, Blizzard and others 2013), light levels too low to support populations of sun-loving grasses, sedges and forbs. Also, it is less likely that individual oak seedlings will accumulate from successive acorn crops to develop into large advance reproduction when understory light levels are below 20 percent, and the more shade tolerant oak species such as white oak (*Quercus alba*) will be most likely to persist, though its growth is inhibited by decreasing light levels. Forest development during the fire suppression era has led to several common situations challenging managers who desire to restore oak woodlands or regenerate oak: (1) oak advance reproduction is absent, or (2) it is present but small in size with low regeneration potential, and (3) ground flora characteristic of oak woodlands have been lost or greatly diminished, especially where sites have been impacted by overgrazing, soil erosion and invasive species.

The overriding goal in woodland restoration is to reduce tree density and canopy cover by removing any midstory stratum and possibly a portion of the overstory. Removal of the midstory creates “closed” woodland structure but overstory tree density must be reduced to achieve “open” woodland structure and light levels to support heliophilic ground flora species (Nelson 2010). The degree of overstory removal is often driven by desired ground flora composition and wildlife habitat considerations. Reducing tree density achieves several objectives in woodland restoration by creating desirable woody structure and by increasing available light in the understory to stimulate ground flora recovery. Prescribed burning has been the method of choice for woodland restoration because of fire’s role in the historical ecology of woodlands. Many woodland indicator species are adapted to a regime of repeated fire, even requiring fire to stimulate germination, prepare suitable seed bed conditions, remove excessive litter,

and retard woody species dominance, thus ensuring adequate light in the understory.

METHODS FOR WOODLAND RESTORATION

The Role of Fire

Initial thinking in woodland restoration was that "If we just started burning again, all would be well." However, early experiences with low intensity prescribed fires conducted in the dormant season were less than gratifying. While a single application of low intensity dormant season fire has been shown to slightly improve species richness and percent cover of woodland indicator species, significant increases in the coverage of desired woodland ground flora required reduction of overstory density by mechanical cutting in conjunction with low intensity burning (Hutchinson and others 2005, Hutchinson 2006, Waldrop and others 2008, Kinkead and others 2013). Low intensity fires are ineffective in significantly reducing overstory tree density or canopy cover. These fires are capable of reducing tree density for hardwoods less than about 4 to 6 inches dbh, depending on species (Dey and Hartman 2005, Arthur and others 2012), which may help achieve the structural targets for closed woodlands. But such fires only increase light levels from < 5 percent of full sunlight in the understory, typical of many forests, to 10 - 15 percent by removal of the midstory canopy (Lorimer and others 1994, Ostrom and Loewenstein 2006, Motsinger and others 2010). This is insufficient to promote the proliferation of many species that demand moderate to full sunlight conditions. More intense fires are needed to thin the overstory, but fire is an imprecise tool for managing the overstory, especially on small tracts.

Mechanical and Chemical Thinning

There is less control over what trees are removed when using fire than if thinning is done by mechanical or chemical methods. It is more difficult to achieve desired stand stocking targets and to control spatial arrangements of trees by prescribed burning. Increasing fire intensity sufficient to kill large trees, also increases the probability of injuring the lower boles of surviving trees, which makes them more vulnerable to fungal infection and wood decay over time (Marschall and others 2014, Dey and Schweitzer 2015). Therefore, using commercial harvesting and mechanical/chemical thinning of unmerchantable trees that are too large for low to moderate intensity fires to control is often preferred for meeting desired future stocking conditions. Income from commercial harvesting can help pay for the cost of managing other unmerchantable material, invasive species control, site preparation, or artificial regeneration of floral species. High intensity fires capable of killing overstory trees may have a role when restoring large landscapes where timber is noncommercial due to insufficient volume, quality, or accessibility. Such fires may occur on those portions of the landscape that may support that fire behavior while maintaining public safety. In many applications though, it is a combination of timber harvesting, mechanical/chemical thinning and prescribed burning that is the preferred alternative in oak woodland restoration.

Mechanical thinning or timber harvesting alone without fire have produced positive responses in desired species richness and coverage in woodland restoration, similar to what results after a single prescribed burning (Zenner and others. 2006, Hutchinson 2006, Waldrop and others. 2008, Kinkead and others. 2013). However, these gains in diversity are ephemeral as an abundance of woody sprouts grow rapidly to form canopy closure and shade out the ground flora, especially if residual overstory density is moderate to low (≤ B-level stocking). Maintenance of a closed canopy in the overstory can retard the regrowth of hardwood and shrub sprouts (Dey and Hartman 2005) but it also inhibits ground flora production. Therefore, it is the combination of mechanical thinning of the overstory with a cycle of prescribed burning that sustains ground flora recovery in woodlands. Herbicides can be used effectively to kill woody stems when applied by stem injection or basal bark application before timber harvesting or prescribed burning; or when applied as a foliar spray to woody sprouts that form after topkill resulting from mechanical cutting or prescribed burning (DiTomaso and others 2006).

The sequencing of timber harvesting or mechanical/chemical thinning and commencement of prescribed burning to reduce stand density can be managed to allow small oak advance reproduction to persist and accumulate as large reproduction overtime. It is important to maintain a pool of larger oak advance reproduction for the day that the woodland overstory needs to be replaced. In many eastern oak forests, oak advance reproduction is commonly small seedlings that are < 12 inches tall and < 0.25 inches in basal diameter. Small oak seedlings have higher probabilities of mortality from prescribed burning than larger oak seedlings and seedling sprouts (Johnson 1974, Dey and Hartman 2005, Brose and others 2013). Hence, alternatives to fire are needed to promote development of large oak advance reproduction. Timber harvesting to create a shelterwood overstory before initiating prescribed burning has been shown to benefit oak advance reproduction survival and growth (Brose and others 1999, Brose 2010).

ADAPTATION OF THE SHELTERWOOD METHOD

Developing Large Oak Advance Reproduction

Brose and others (2013) recommended using the shelterwood method to increase light for improved oak seedling survival and growth before initiating prescribed

burning because larger oak seedlings (e.g., > 0.5 inches in basal diameter) have higher sprouting probabilities (Dey and Hartman 2005) and greater root reserves (Brose 2008) to support competitive growth rates. They suggested that burning may begin several years after the first harvest of a two-stage shelterwood system, or several years after the final overstory removal that releases the regeneration. Oak seedling growth can be promoted for several years by midstory removal and reducing overstory density by mechanical/chemical methods, that is, until competing vegetation begins to diminish available light to the oaks. At that point, a second harvest to bring final overstory density to desired levels, or the initiation of prescribed burning will be needed to release oak advance reproduction. Keeping the intensity of the first fire low reduces the deleterious effects on smaller oak reproduction. Fuel loading may be high in areas where fire has been suppressed for 15 years or more (Stambaugh and others 2006a), or where there have been increases in fuels from silvicultural activity; in these situations, controlling fire behavior to keep intensity low can avoid undesirable effects on oak reproduction or overstory trees. Later, when oak reproduction is large enough (e.g., > 0.5 inches basal diameter) and oak competitors are leafing out in the late spring, moderate to higher intensity fires may give oak an added competitive advantage (Brose and others 2013). If oak advance reproduction is absent or sparse, then prescribed burning can be done immediately to reduce deep leaf litter, decrease midstory canopy, begin controlling understory woody vegetation, release nutrients, stimulate germination of seeds that have chemical or thermal dormancy, increase soil temperature, and increase light, all of which promotes ground flora development (Hutchinson 2006). But once a good acorn crop is on the ground, fire should be delayed until large oak seedlings are developed through a modified shelterwood regeneration method approach.

Managing Ground Flora

There is much flexibility in managing overstory density and prescribed fire regime to produce the desired ground flora composition and structure. Overstory density and vertical structure (number of canopy layers) largely determines the amount of light reaching the ground flora. The overstory density can be managed to suppress shade intolerant undesirable species but this must be balanced with the physiological needs for light of the desired flora. Since there may be 300 or more species in the understory, with a range of different light requirements for good growth and reproduction, a reasonable approach would be to set overstory density targets to provide light needed for the key indicator species that represent a high quality and healthy community, or needed for the predominant species or functional groups. Overstory crown cover must be less than 50 percent for many warm season (C4) grasses to be dominant in the community (Mayer and Khalyani 2011, Starver and others 2011). Heterogeneity in the spatial arrangement of trees can create variation in understory light conditions that can accommodate the needs of more species than if the overstory were uniformly dispersed.

Fire frequency and seasonality strongly influence the dominance of plant functional groups (Anderson and others 1999, Nelson 2010). For example, annual dormant season fires favor grasses, biennial and summer fires promote forbs, periodic dormant season fires favor woody species. Varying the frequency, intensity, and seasonality of fire may provide for greater plant diversity in the long term. A common concern in restoration is the presence of invasive species in or adjacent to the restoration site. Many invasive species such as smooth brome grass (*Bromus inermis*), musk thistle (*Carduus nutans*), sericia lespedeza (*Lespedeza cuneata*) and crown vetch (*Coronilla varia*) are adapted to fire and thrive in more open environments. Zouhar and others (2008) and DiTomaso and others (2006) provide good overviews of fire effects on and control of non-native invasive species.

GRAZING IN WOODLANDS

The inclusion of large ungulate grazers such as bison (*Bison bison*), elk (*Cervus canadensis*) and white-tailed deer (*Odocoileus virginianus*) can modify plant response under a given overstory density and fire regime. By their grazing, they place selective pressure on the more palatable species and reduce fine fuel loading, lessening the occurrence and intensity of the next fire. Freshly burned areas attract large ungulates because of the abundance of nutritious, highly palatable and available forage and browse. This spatially and temporally dynamic interaction between grazers/browsers and fire at a landscape scale created a shifting mosaic and increased heterogeneity of habitats in the past that supported relatively high biodiversity in flora and fauna. The fire-grazer interaction has been termed pyric herbivory (fire driven grazing) by Fuhlendorf and others. (2008). Most research in fire-grazer interactions and effects on ecosystems have been conducted in prairie ecosystems where Collins and others (1998) and Hartnett and others (1996) have observed that bison grazing on the dominant C4 grasses led to greater spatial heterogeneity in vegetation and increases in total species richness.

SETTING STRUCTURAL TARGETS IN WOODLANDS

Shelterwood overstory stocking levels that are prescribed to enhance oak regeneration in forests also fall within the range of desired woodland overstory stocking (MDC 2010, Nelson 2010) (fig. 1). For example, MTNF (2005) set desired woodland density in the

Missouri Ozarks from 30 to 80 ft^2/ac of basal area. Historically (early 1800s) in this same region, Hanberry and others (2014b) determined that average basal area and stocking in open woodlands were 61ft^2/ac and 41 percent, respectively; and averaged for closed woodlands 100 ft^2/ac and 64 percent, respectively, with much variation among ecological subsections. It is commonly recommended that overstory density be reduced to about B-level stocking (i.e., ca 60 percent) to promote oak regeneration (e.g., Brose and others 2008, Johnson and others 2009), which is within the range of historic stocking levels (30 to 75 percent according to Hanberry and others 2014b) in Missouri. Hence, shelterwood prescriptions for oak regeneration are also good starting points in developing prescriptions for woodland restoration. However, in woodland restoration, the final overstory removal harvest would not be done, and the overstory would be retained over the long-run until it needed to be replaced due to senescence, increasing mortality and loss of acorn production. Two- and three-stage shelterwood approaches are appropriate for woodland management with the modification that the final shelterwood overstory would be retained for the long-term. Desired overstory density in woodlands is determined by considering wildlife habitat and biodiversity objectives, knowledge of reference stands and historic conditions, and the resource needs of ground flora. Stocking charts and their modifications have been developed to help managers implement woodland prescriptions and monitor restoration progress (Law and others 1994, Kabrick and others 2014) using basic structural metrics such as basal area, tree density, stocking and canopy crown cover (figs. 1 and 2). Relationships between structural metrics (e.g., crown cover and stocking) and understory light levels (Blizzard and others 2013) are indispensable for establishing structural thresholds and setting targets to ensure adequate light is available for oak regeneration and ground flora development (fig. 3).

WOODLAND MAINTENANCE

Once desired woodland structure and composition have been achieved, management changes from a restoration approach to one of maintenance of the desired condition. The period of maintenance management may persist for 100 years or more depending on the longevity of the overstory trees. Oaks can be long-lived, i.e., red oaks may live to 150-200 years and white oaks from 250-400 years. Maintenance of woodland ecosystems requires frequent fire, the timing of which depends on sustaining the desired composition and structure. Fire is needed to

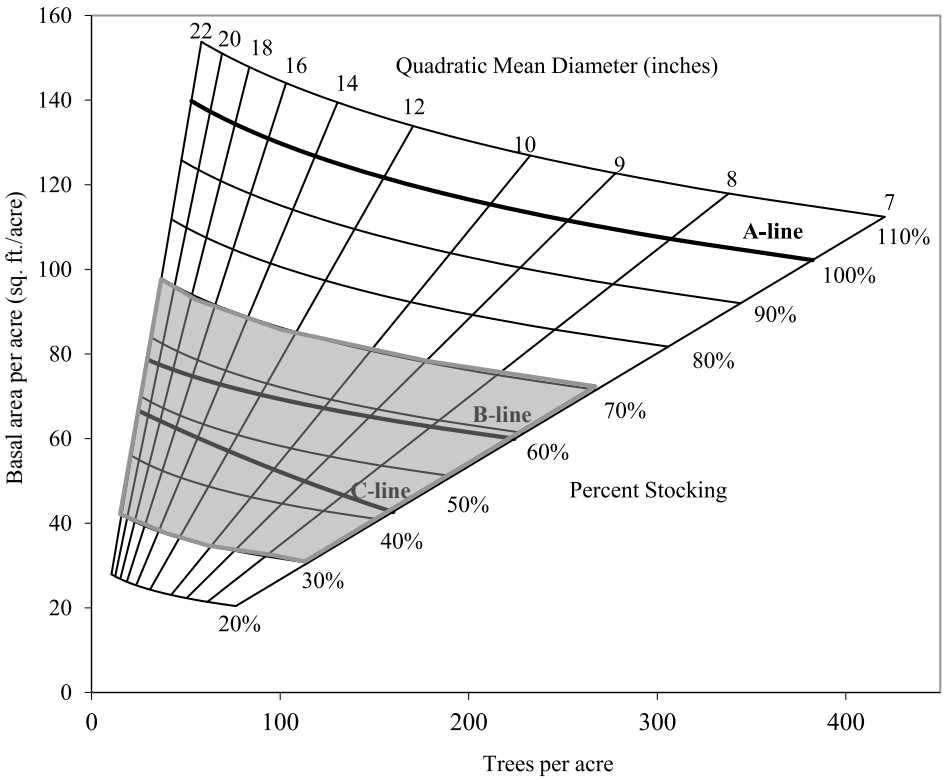


Figure 1—Stocking chart for upland *Quercus-Carya* in the Central Hardwoods Region (Gingrich 1967) modified for managing savanna and woodlands (Kabrick and others 2014). Stocking in woodlands is maintained between about 30% and 70% stocking. Closed woodlands have higher stocking (70%) compared to open woodlands (30 to 40%) and savannas (<30%). When regenerating closed woodlands, stocking is reduced to below B-level stocking.

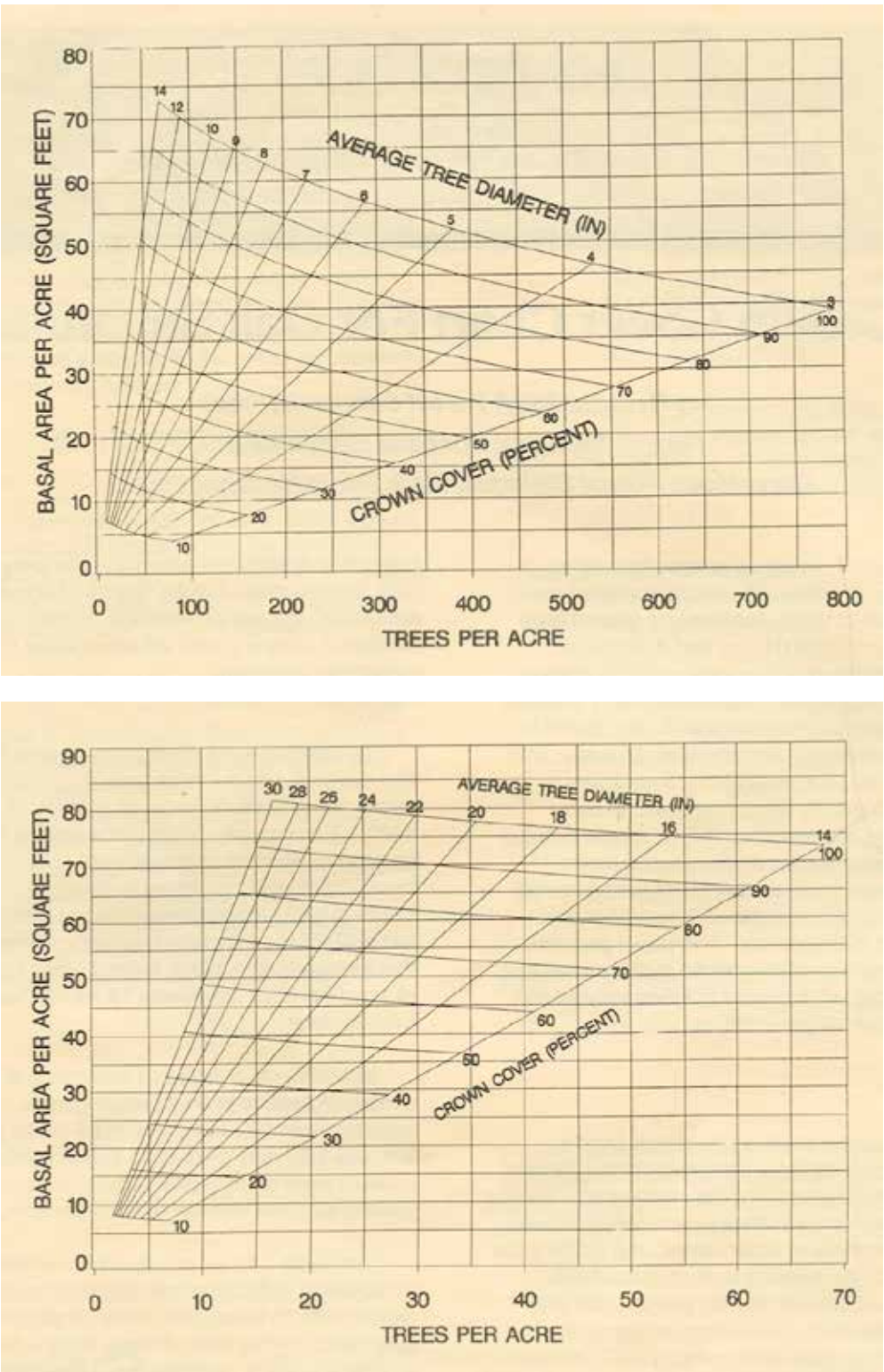


Figure 2—Estimation of crown cover percent in *Quercus-Carya* dominated savannas and open woodlands in the Central Hardwood Region based on Gingrich (1967) (from Law and others 1994, with permission).

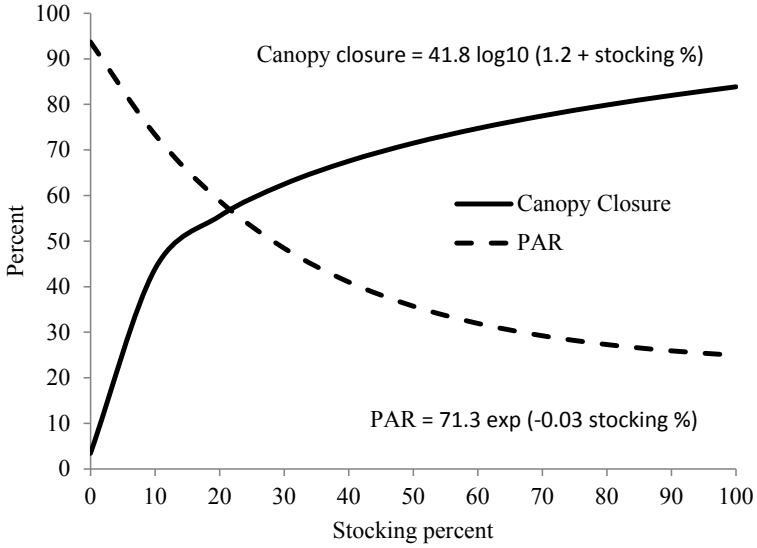


Figure 3—Relationships between stocking, canopy closure, and photosynthetically-active radiation (PAR) in *Quercus-Carya* stands that have been thinned from below in the Missouri Ozarks (adapted from Blizzard and others 2013).

both retard growth and dominance of woody species and to maintain floristic quality. Frequency of fire to control woody stems depends on the growth rates of seedlings and sprouts. Low intensity fires can readily topkill hardwoods up to about 4 inches dbh. Growth rates of hardwoods depend on reproductive origin (true seedling or sprout from well-established root system), species, site quality, and overstory density. For example, dominant white oak saplings growing in the open on sites of medium site index in Missouri (e.g., 63 feet base age 50) average 1.5 inches of diameter growth in 10 years (Shifley and Smith 1982). Hence, it would take more than 20 years for stems to grow large enough to have increased chances of surviving a low intensity fire intact. Oak stump sprouts have higher initial growth rates than other forms of oak reproduction. For example, white oak stump sprouts can grow to 2.2 inches dbh after 10 years of growing in the open on average sites in the Missouri Ozarks, whereas scarlet oak stump sprouts can average 3.1 inches dbh in the same time (Dey and others 2008). Increasing overstory density reduces growth, especially for the more shade intolerant oaks, as little as 20 ft^2/ac of basal area can significantly reduce height growth in black and scarlet oak (Green 2008), and diameters of oak stump sprouts averaged only 0.4 inches after 10 years growing under 62 ft^2/ac, 58 percent crown cover (Dey and others 2008).

If controlling hardwood regrowth were the only purpose for burning, fires would not have to be that frequent, i.e., every 10 to 20 years to retard hardwood dominance and canopy closure. However, Stambaugh and others (2006a) have shown that hardwood leaf litter reaches maximum depths in about 12 to 15 years in Ozark forests, and it takes only 4 years after a fire for leaf litter to recover to 75 percent of preburn levels. More frequent fires are needed to keep leaf litter from reducing ground flora abundance and diversity. And finally, fires may need to be even more frequent than that depending on the ecology of the desired flora, for example, annual fires may be needed to sustain grass dominance.

REPLACING THE WOODLAND OVERSTORY

Eventually, there comes a time when woodland overstories need to be replaced and this necessitates the release of oak advance reproduction and its recruitment into the overstory. Fire must be withheld long enough for oak seedling sprouts to grow large enough that they can resist being topkilled when fire is returned to the woodland system. This may take 10 to 30 years depending on the myriad of factors that affect diameter growth in hardwoods (Arthur and others 2012, Kabrick and others 2014). Fire-free periods of this length have been commonly observed in the historic period since the mid-1600s in many fire history studies in eastern hardwoods (e.g., Guyette and others 2002, Guyette and others 2003, Guyette and Spetich 2003, Stambaugh and others 2006b). Variations of either even-aged or uneven-aged methods of regeneration can be used in woodlands depending on the size of individual woodland area, length of the woodland rotation, desire to maintain continuous mature tree

cover, ability to manage intensively, and other factors. If for some reason, oak advance reproduction is not present for recruitment, then attention must be given to secure its establishment 10 to 20 years before the recruitment process begins. Artificial regeneration by planting seedlings or direct seeding of acorns may be necessary if the capacity of the overstory to produce acorns is insufficient. Supplementing natural oak advance reproduction through underplanting can shorten the time it takes to be ready for recruitment.

CONCLUSION

Open-structured oak woodlands were once prominent natural communities in eastern North America. They existed because of a long-history of frequent fire. Their distribution changed over time with changing climates, and human populations and cultures. With the advent of fire suppression, these communities succeeded to closed forests. Today, they are rare throughout the East. Restoration of oak woodlands has become a focus of land managers. Restoration and maintenance of woodlands requires active management. Reintroducing fire is fundamental to restoration, but other silvicultural practices are needed to efficiently manage vegetation composition and structure, and achieve desired future conditions. Management efforts to restore oak woodlands often precede research, provide early tests of innovative treatment combinations, and help to identify key research questions. Monitoring to inform adaptive management is an important source of knowledge and a critical part of the learning process. Restoring oak woodlands will help to expand the distribution of rare natural communities, conserve native biodiversity, create a more diverse landscape, provide habitat for wildlife species of concern, and should increase our options for responding to uncertain futures due to increasing human population, climate change and invasive species.

LITERATURE CITED

Anderson, R.C.; Fralish, J.L.; Baskin, J.M. 1999. Savannas, barrens, and rock outcrop plant communities of North America. Cambridge, UK: Cambridge University Press. 470 p.

Arthur, M.A.; Alexander, H.D.; Dey, D.C. [and others]. 2012. Refining the oak–fire hypothesis for management of oak-dominated forests of the Eastern United States. Journal of Forestry. 110: 257-266.

Au, L.; Andersen, D.E.; Davis, M. 2008. Patterns in bird community structure related to restoration of Minnesota dry oak savannas and across a prairie to oak woodland ecological gradient. Natural Areas Journal. 28: 330-341.

Batek, M.J.; Rebertus, A.J.; Schroeder, W.A. [and others]. 1999. Reconstruction of early nineteenth-century vegetation and fire regimes in the Missouri Ozarks. Journal of Biogeography. 26: 397-412.

Blizzard, E.M.; Kabrick, J.M.; Dey, D.C. [and others]. 2013. Light, canopy closure, and overstory retention in upland Ozark forests. In: Guldin, J.M., ed. Proceedings of the 15th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-GTR-175. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 73-79.

Bowles, M.L.; McBride, J.L. 1998. Vegetation composition, structure, and chronological change in a decadent Midwestern North America savanna remnant. Natural Areas Journal. 18: 14-27.

Brandt, L.; He, H.; Iverson, L. [and others]. 2014. Central Hardwoods ecosystem vulnerability assessment and synthesis: a report from the Central Hardwoods climate change response framework project. Gen. Tech. Rep. NRS-124. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 254 p.

Brawn, J.D. 2006. Effects of restoring oak savannas on bird communities and populations. Conservation Biology. 20: 460-469.

Brawn, J.D.; Robinson, S.K.; Thompson III, F.R. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics. 32: 251-276.

Brose, P.H. 2008. Root development of acorn-origin oak seedlings in shelterwood stands on the Appalachian Plateau of northern Pennsylvania: 4-year results. Forest Ecology and Management. 255: 3374-3381.

Brose, P.H. 2010. Long-term effects of single prescribed fires on hardwood regeneration in oak shelterwood stands. Forest Ecology and Management. 260: 1516-1524.

Brose, P.H.; Van Lear, D.H.; Keyser, P.D. 1999. A shelterwood-burn technique for regenerating productive upland oak sites in the Piedmont region. Southern Journal of Applied Forestry. 16(3): 158-163.

Brose, P.H.; Dey, D.C.; Phillips, R.J.; Waldrop, T.A. 2013. A meta-analysis of the fire-oak literature: does prescribed burning promote oak reproduction in eastern North America? Forest Science. 59(3): 322-334.

Brose, P.H.; Gottschalk, K.W.; Horsley, S.B. [and others]. 2008. Prescribing regeneration treatments for mixed-oak forests in the Mid-Atlantic region. Gen. Tech. Rep. NRS-33. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 100 p.

Collins, S.L.; Knapp, A.K.; Briggs, J.M. [and others]. 1998. Modulation of diversity by grazing and mowing in native tallgrass prairie. Science. 280: 745-747.

Davis, M.A.; Peterson, D.W.; Reich, P.B. [and others]. 2000. Restoring savanna using fire: impact on the breeding bird community. Restoration Ecology. 8: 30-40.

Delcourt, P.A.; Delcourt, H.R. 1991. Quaternary paleoecology of the Lower Mississippi Valley. Engineering Geology. 45: 219-242.

Dey, D.C.; Hartman, G. 2005. Returning fire to Ozark Highland forest ecosystems: effects on advance regeneration. Forest Ecology and Management. 217: 37-53.

Dey, D.C.; Schweitzer, C.J. 2015. Timing fire to minimize damage in managing oak ecosystems. In: Holley, A.G.; Connor, K.F.; Haywood, J.D., eds. 2015. Proceedings of the 17th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-203. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 143-153

Dey, D.C.; Jensen, R.G.; Wallendorf, M.J. 2008. Single-tree selection reduces survival and growth of oak stump sprouts in the Missouri Ozark Highlands. In: Jacobs, D.F.; Michler, C.H., eds. 2008. Proceedings of the 16th central hardwood forest conference. Gen. Tech. Rep. NRS-P-24. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 26-37.

DiTomaso, J.M.; Brooks, M.L.; Allen, E.B. [and others]. 2006. Control of invasive weeds with prescribed burning. Weed Technology. 20(2): 535-548.

Fuhlendorf, S.D.; Engle, D.M.; Kerby, J.; Hamilton, R. 2008. Pyric herbivory: rewilding landscapes through the recoupling of fire and grazing. Conservation Biology. 23(3): 588-598.

Gardiner, E.S.; Yeiser, J.L. 2006. Underplanting cherrybark oak (*Quercus pagoda* Raf.) seedlings on a bottomland site in the southern United States. New Forests. 32: 105-119.

Gingrich, S.F. 1967. Measuring and evaluating stocking and stand density in upland hardwood forests in the Central States. Forest Science. 13: 38-53.

Green, J. 2008. Overstory density and its effect on regeneration abundance, understory height growth, and overstory canopy expansion in the Missouri Ozark Highlands. Columbia, MO: University of Missouri. 79 p. M.S. thesis.

Grundel, R; Pavlovic, N.B. 2007. Response of bird species densities to habitat structure and fire history along a Midwestern open-forest gradient. Condor. 109: 734-749.

Guyette, R.P.; Spetich, M.A. 2003. Fire history of oak–pine forests in the Lower Boston Mountains, Arkansas, USA. Forest Ecology and Management. 180: 463-474.

Guyette, R.P.; Dey, D.C.; Stambaugh, M.C. 2003. Fire and human history of a barren-forest mosaic in southern Indiana. The American Midland Naturalist. 149(1): 21-34.

Guyette, R.P.; Muzika, R.M.; Dey, D.C. 2002. Dynamics of an anthropogenic fire regime. Ecosystems. 5(5): 472-486.

Guyette, R.P.; Stambaugh, M.C.; Dey, D.; Muzika, R.M. 2012. Predicting fire frequency with chemistry and climate. Ecosystems. 15: 322- 335.

Haddad, N.M.; Tilman, D.; Haarstad, J. [and others]. 2001. Contrasting effects of plant richness and composition on insect communities: a field experiment. The American Naturalist. 158(1): 17-35.

Hanberry, B.B.; Dey, D.C.; He, H.S. 2012. Regime shifts and weakened environmental gradients in open oak and pine ecosystems. PLoS ONE 7 (7), e41337. doi:10.1371/journal.pone.0041337.

Hanberry, B.B.; Dey, D.C.; He, H.S. 2014a. The history of widespread decrease in oak dominance exemplified in a grassland-forest landscape. Science of the Total Environment. 476-477: 591-600.

Hanberry, B.B.; Jones-Farrand, D.T.; Kabrick, J.M. 2014b. Historical open forest ecosystems in the Missouri Ozarks: reconstruction and restoration targets. Ecological Restoration. 32(4): 407-416.

Hanberry, B.B.; Kabrick, J.M.; He, H.S. 2014c. Changing tree composition by life history strategy in a grassland-forest landscape. Ecosphere. 5(3):Article 34: 1-16. www.esajournals.org [Date accessed: May 10, 2015].

Hanberry, B.B.; Kabrick, J.M.; He, H.S. 2014d. Densification and state transition across the Missouri Ozarks landcape. Ecosystems. 17: 66-81.

Hartnett, D.C.; Hickman, K.R.; Fischer Walter, L.E. 1996. Effects of bison grazing, fire, and topography on floristic diversity in tallgrass prairie. Journal of Range Management. 49: 413-420.

Hunter, Jr., M.L.; Schmiegelow, F.K.A. 2011. Wildlife, forests and forestry. 2nd ed. New York: Prentice Hall. 259 p.

Huston, M.A. 1994. Biological diversity the coexistence of species on changing landscapes. Cambridge, UK: Cambridge University Press. 681 p.

Hutchinson, T.F. 2006. Fire and the herbaceous layer in eastern oak forests. In: Dickinson, M.B., ed. Fire in eastern oak forests: delivering science to land managers, proceedings of a conference; 2005 November 15-17; Columbus, OH. Gen. Tech. Rep. NRS-P-1. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 136-149.

Hutchinson, T.F.; Boerner, R.E.J.; Sutherland, S. [and others]. 2005. Prescribed fire effects on the herbaceous layer of mixed-oak forests. Canadian Journal of Forest Research. 35: 877-890.

Janowiak, M.K.; Swanston, C.W.; Nagel, L.M. [and others]. 2011. Silvicultural decisionmaking in an uncertain climate future: a workshop-based exploration of considerations, strategies, and approaches. Gen. Tech. Rep. NRS-81. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 14 p.

Janowiak, M.K.; Iverson, L.R.; Mladenoff, D.J. [and others]. 2014. Forest ecosystem vulnerability assessment and synthesis for northern Wisconsin and western Upper Michigan: a report from the Northwoods climate change response framework project. Gen. Tech. Rep. NRS-136. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 247 p.

Johnson, P.S. 1974. Survival and growth of northern red oak seedlings following a prescribed burn. Research Note NC-177. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 3 p.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2009. The ecology and silviculture of oaks. 2nd ed. New York: CABI Publishing. 580 p.

Kabrick, J.M.; Dey, D.C.; Kinkead, C.O. [and others]. 2014. Silvicultural considerations for managing fire-dependent oak woodland ecosystems. In: Groninger, J.W.; Holzmueller, E.J.; Nielsen, C.K.; Dey, D.C., eds. Proceedings of the 19th central hardwood forest conference. Gen. Tech. Rep. NRS-P-142. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 2-15.

King, D.I.; Schlossberg, S. 2014. Synthesis of the conservation value of the early-successional stage in forests of eastern North America. Forest Ecology and Management. 324: 186-195.

Kinkead, C.O.; Kabrick, J.M.; Stambaugh, M.C.; Grabner, K.W. 2013. Changes to oak woodland stand structure and ground flora composition caused by thinning and burning. In: Miller, G.W.; Schuler, T.M.; Gottschalk, K.W. [and others], eds. Proceedings of the 18th central hardwood forest conference. Gen. Tech. Rep. NRS-P-117. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 373-383.

Law, J.R.; Johnson, P.S.; Houf, G. 1994. A crown cover chart for oak savannas. Technical Brief TP-NC-2. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 4 p.

Leach, M.K.; Givnish, T.J. 1999. Gradients in the composition, structure, and diversity of remnant oak savannas in southern Wisconsin. Ecological Monographs. 69(3): 353-374.

Lhotka, J.M.; Loewenstein, E.F. 2009. Effect of midstory removal on understory light availability and the 2-year response of underplanted cherrybark oak seedlings. Southern Journal of Applied Forestry. 33(4): 171-177.

Lorimer, C.G.; Chapman, J.W.; Lambert, W.D. 1994. Tall understorey vegetation as a factor in the poor development of oak seedlings beneath mature stands. Journal of Ecology. 82: 227-237.

Marschall, J.M.; Guyette, R.P.; Stambaugh, M.C.; Stevenson, A.P. 2014. Fire damage effects on red oak timber product value. Forest Ecology and Management. 320: 182-189.

Mayer, A.L.; Khalyani, A.H. 2011. Grass trumps trees with fire. Science. 334: 188-189.

MDC. 2010. Missouri's forest resource assessment and strategy. Jefferson City, MO: Missouri Department of Conservation. 222 p.

Motsinger, J.R.; Kabrick, J.M.; Dey, D.C. [and others]. 2010. Effects of midstory and understory removal on the establishment and development of natural and artificial pin oak advance reproduction in bottomland forests. New Forests. 39: 195-213.

MTNF. 2005. 2005 Mark Twain National Forest Land and resource management plan. Rolla, MO: U.S. Department of Agriculture Forest Service, Mark Twain National Forest. 234 p.

Nelson, P.W. 2010. The terrestrial natural communities of Missouri. Jefferson City, MO: Missouri Natural Areas Committee. 550 p.

Nigh, T.A.; Schroeder, W.A. 2002. Atlas of Missouri ecoregions. Jefferson City, MO: Missouri Department of Conservation. 212 p.

Ostrom B.J.; Loewenstein, E.F. 2006. Light transmittance following midstory removal in a riparian hardwood forest. In: Connor, K.F.; ed. 2006. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 265-268.

Parker, W.C.; Dey, D.C. 2008. Influence of overstory density on ecophysiology of red oak (*Quercus rubra*) and sugar maple (*Acer saccharum*) seedlings in central Ontario shelterwoods. Tree Physiology. 28: 797-804.

Peterson, D.W.; Reich, P.B. 2008. Fire frequency and tree canopy structure influence plant species diversity in a forest-grassland ecotone. Plant Ecology. 194: 5-16.

Reidy, J.L.; Thompson III, F.R.; Kendrick, S.W. 2014. Breeding bird response to habitat and landscape factors across a gradient of savanna, woodland, and forest in the Missouri Ozarks. Forest Ecology and Management. 313: 34-46.

Sander, I.L. 1979. Regenerating oaks with the shelterwood system. In: Proceedings of 1979 J.S. Wright forest conference. West Lafayette, IN: Purdue University: 54-60.

Schroeder, W.A. 1982. Presettlement prairie of Missouri. Natural History Series No. 2. Jefferson City, MO: Missouri Department of Conservation. 38 p.

Schulte, L.A.; Mladenoff, D.J.; Crow, T.R. [and others]. 2007. Homogenization of northern U.S. Great Lakes forests due to land use. Landscape Ecology. 22: 1089-1103.

Shifley, S.R.; Smith, W.B. 1982. Diameter growth, survival, and volume estimates for Missouri trees. Research Note NC-292. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 7 p.

Shifley, S.R.; Thompson III, F.R. 2011. Chapter 6 spatial and temporal patterns in the amount of young forests and implications for biodiversity. In Greenberg, C.H.; Collins, B.S.; Thompson III, F.R., eds. Sustaining young forests communities ecology and management of early successional habitats in the Central Hardwood Region, USA. New York: Springer: 73-95.

Shifley, S.R.; Aguilar, F.X.; Song, N.; Stewart, S.I. [and others]. 2012. Forests of the northern United States. Gen. Tech. Rep. NRS-90. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 202 p.

Stambaugh, M.C.; Guyette, R.P.; Grabner, K.W.; Kolaks, J. 2006a. Understanding Ozark forest litter variability through a synthesis of accumulation rates and fire events. In: Andrews, P.L.; Butler, B.W., comps. 2006. Fuels management-how to measure success: conference proceedings. RMRS-P-41. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 321-332

Stambaugh, M.C.; Guyette, R.P.; McMurry, E.R.; Dey, D.C. 2006b. Fire history at the eastern Great Plains margin, Missouri River loess hills. Great Plains Research. 16: 149-159.

Starbuck, C.A.; Amelon, S.K.; Thompson III, F.R. 2014. Relationships between bat occupancy and habitat and landscape structure along a savanna, woodland, forest gradient in the Missouri Ozarks. Wildlife Society Bulletin. DOI: 10.1002/wsb.512. [Date accessed: unknown].

Starver, A.C.; Archibald, S.; Levin, S.A. 2011. The global extent and determinants of savanna and forest as alternative biome states. Science. 334: 230-232.

Taft, J.B.; Schwartz, M.W.; Philippe, L.R. 1995. Vegetation ecology of flatwoods on the Illinoian till plain. Journal of Vegetation Science. 6: 647-666.

Thompson III, F.R.; Reidy, J.L.; Kendrick, S.W.; Fitzgerald, J.A. 2012. Songbirds in managed and non-managed savannas and woodlands in the central hardwoods region. In: Dey, D.C.; Stambaugh, M.C.; Clark, S.L.; Schweitzer, C.J., eds. Proceedings of the 4th fire in eastern oak forests conference. Gen. Tech. Rep. NRS-P-102. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 159-169.

Transeau, E.N. 1935. The prairie peninsula. Ecology. 16(3): 423-437.

Waldrop, T.A.; Yaussy, D.A.; Phillips, R.J.; [and others]. 2008. Fuel reduction treatments affect stand structure of hardwood forests in western North Carolina and southern Ohio, USA. Forest Ecology and Management. 255: 3117-3129.

Zenner, E.K.; Kabrick, J.M.; Jensen, R.G.; [and others]. 2006. Responses of ground flora to a gradient of harvest intensity in the Missouri Ozarks. Forest Ecology and Management. 222: 326-334.

Zouhar, K.; Smith, J.K.; Sutherland, S. 2008. Effects of fire on nonnative invasive plants and invisibility of wildland ecosystems. In: Gen. Tech. Rep. RMRS-42, Vol.6, Ch.2. Ft. Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 7-31.

THE INFLUENCE FROM COMBINATIONS OF PRESCRIBED FIRE, HERBICIDE INJECTION, AND PARTIAL OVERSTORY REMOVAL ON RESTORATION OF NATURAL OAK STANDS IN THE ARKANSAS OZARKS

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Abstract—Two studies were conducted in the Springfield Plateau province of the Arkansas Ozarks from 2009 to 2014 incorporating the applied silvicultural methods proven beneficial to promote oak natural reproduction. The first study, River Hills, was a completely randomized design with treatments including: partial overstory removal to basal area 50 (BA50), partial overstory removal to basal area 50 plus mid-story removal (BA50+MR), and a non-harvest control (NHC). An additional treatment was applied in early 2013 involving a prescribed burn of the BA50 and BA50+MR treatments. Prior to the prescribed burn application, a significant difference was detected between initial versus year 3 total (all 3 height classes) oak seedlings per acre (SPA) measurements for all treatment ($p = 0.003, 0.008, 0.007$; Student's t-test, $\alpha = 0.05$), associated with light availability. A prescribed fire conducted in year 4 had a negative impact on total oak SPA. The second study, Waugh Mountain, was a RCB, split plot design. Treatments included a partial overstory removal to basal area 50 (SW) and partial overstory removal to basal area 50 plus mid-story removal (SW+MR), with the split plot factor being either pre-harvest prescribed burn or no-burn. No significant differences were observed between initial and year 2 post-treatment total oak SPA. Significant differences were detected between initial and year 2 post-treatment oak SPA between 1 and 3 feet and greater than 3 feet in height.

INTRODUCTION

Oaks have been the predominant species in hardwood forests of the Arkansas Ozarks for thousands of years. The forests also included hickories (*Carya* spp.), ash (*Fraxinus* spp.), blackgum (*Nyssa sylvatica*), elm (*Ulnus* spp.), maple (*Acer* spp.), other hardwoods, and a component of shortleaf pine (*Pinus enchinata* L.), as stated by Foti (2004). Densification of hardwood stands within this region over the past century has resulted in reduced sunlight canopy penetration and species shifts to more shade tolerant species (Soucy and others 2004). The result from this management shift is an ongoing reduction in levels of oak regeneration in newly established stands, which could have significant impacts on the ecology and utilization of Ozark upland forests. Decades of research aimed at reversing these effects have provided methods for increasing oak reproduction abundance and size in naturally regenerated stands.

Previous studies have demonstrated the positive impacts prescribed fire, midstory removal, and shelterwood harvests can provide to establishing oak reproduction in naturally regenerated hardwoods (e.g. Loftis 1990 and 1993, Larsen and Johnson 1998, Brose and others 1998, Brose and others 2013, Cunningham 2011, and Cunningham 2015). Controlling non-oak competition and altering sunlight canopy penetration to the understory are primary factors impacted by these stand-level modifications. Brose and others 1998 and Brose and others 2013 emphasized the timing of prescribed burning and its subsequent benefit or lack thereof to oak reproduction. Cunningham 2011 and 2015 established sunlight conditions at the understory level associated with varying midstory and overstory densities. Here, we will expand on the results of Cunningham 2015, present results from an additional study and summarize our efforts to incorporate prescribed fire, midstory removal and partial overstory removal in naturally regenerated hardwood stands within the Arkansas Ozarks.

MATERIALS AND METHODS

The study sites are located in the dissected Springfield Plateau physiographic province in the Arkansas Ozarks. The predominant soils are Clarksville very cherty silt loam, 8 to 20 percent slopes and Clarksville very cherty silt loam 20 to 40 percent slopes. These soils are described as deep, somewhat excessively drained, low available water, low organic matter content, and strongly acidic (Ferguson and others 1982). The description provided is a general soil description based

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on broad ranges of slope positions. The areas selected for these studies were only on north aspects, which potentially had somewhat higher organic matter, higher moisture content, and generally considered more productive than ridge-tops and south facing slopes. Site indices for white oak, black oak, and northern red oak dominant and co-dominant trees were calculated from equations developed by Graney and Bower (1971). Site indices for oaks were 65 feet on upper slopes to 75 feet plus on lower slopes.

Study Design

The River Hills study was a completely randomized design, with three treatments and four replicates. Treatments included: 1) shelterwood harvest to BA 50 a^{-1} (BA50), 2) shelterwood harvest to BA 50 a^{-1} plus injection of non-oak stems between 1 and 5 inches DBH (BA50+MR) and 3) non-harvested control (NHC). A detailed description for the River Hills methodology was provided by Cunningham (2015). Three years of post-treatment data were collected, analyzed and presented in the original study (initial, year 1, year 2 and year 3 data). In spring 2013, a prescribed fire was conducted in the BA50 and BA50+MR treatments. The NHC treatment remained undisturbed. Post-burn oak reproduction measurements were obtained in the fall 2014, representing year 5 data presented here.

The Waugh Mountain study was a randomized complete block, split plot design. The study included two primary treatments: shelterwood harvest to residual BA 50 ft^2 a^{-1} (SW) or a residual BA 50 ft^2 a^{-1} plus midstory removal (SWI). Prescribed fire was a split plot factor providing two additional treatments: a SW treatment plus a burn (SW+B) or a SWI treatment plus a burn (SWI+B). The layout resulted in four treatments with three replicates. Mid-story removal treatments were applied in February 2012. Follow-up treatments were applied in July 2012. Mid-story removal was performed using herbicide injection. 0.03 ounces of an aqueous solution of 25 percent imazapyr and 75 percent water was injected for every three inches of diameter at breast height. Prescribed burn applications were conducted on burn plots in March 2012. A partial overstory harvest operation was applied to all treatments from September 2012 through March 2013. The target residual basal area was 50 ft^2 a^{-1}. Desirable residual tree characteristics were 1) oak species and 2) large vigorous crowns.

Each 2.5 acre treatment replicate contained nine 100^{th} acre circular regeneration sample plots spaced on a 20 meter by 20 meter grid. Stand level reproduction measurements at each plot included species and height class (<1 feet, 1 to 3 feet, and >3feet). Over-story measurements were taken from one, fifth acre circular plot per replicate. Over-story measurements included species, DBH, merchantable height, and log grade. Mid-story measurements were taken from 1, 20^{th} acre circular plot per replicate. Mid-story measurements included species and total height. Initial over-story, mid-story and understory measurements were taken in fall 2011.

Statistical Analyses

All statistical analyses were performed in SigmaPlot 11.0. Data were tested for normality and equal variances. When necessary, regeneration data were square root transformed to help meet required assumptions. For the River Hills data, regeneration responses were analyzed for treatment differences using analysis of variance (ANOVA). In the event that a square root transformation did not adequately fix assumption issues, a Kruskal-Wallace ANOVA on Ranks was performed (KW). Individual means separation was conducted using Student Newman-Kuels (SNK) tests. All tests were conducted at the alpha = 0.05 significance level. Waugh Mountain results were analyzed using a pairwise analysis of initial versus year 2 data (Student's t-test, $\alpha = 0.05$).

RESULTS

River Hills

Year 5, or year 2 post-burn, means were 854 (± 141), 1,596 (± 625), 1,534 (± 540) oak seedlings per acre (SPA) for BA50, BA50+MR, and NHC, respectively (figs. 1A and 1B). No significant differences were detected for total oak SPA ($p = 0.49$) between treatments. No significant differences were detected for oak seedlings < 1ft. or 1 – 3 ft. between treatments ($p = 0.16$ [KW] and $p = 0.19$). A significant difference was detected between treatments for oak seedlings > 3 ft. ($p = 0.007$). A SNK test determined significant differences occurred between BA50 and BA50+MR versus NHC for oak seedlings > 3ft. BA50 and BA50+MR were similar for oak seedlings > 3 ft.

Reductions in oak SPA occurred for all treatments between year 3 and year 5 post-treatment data. Year 5 post-treatment oak SPA were 50, 48, and 54 percent lower than year 3 for BA50, BA50+MR, and NHC, respectively. A one-way analysis of variance for all years detected significant differences between years for total oak SPA ($p = <0.001$). A SNK analysis determined year 3 to differ significantly from all other years and year 5 to differ from initial and year 1. Initial through year 2 were determined to be similar for total oak SPA (fig. 2).

Waugh Mountain

Year 2 post-treatment means were 581 (± 179), 1,167 (± 372.6), 1,467 (± 775.7), 1,325 (± 444.4) total oak SPA for SW, SWI, SW+B, and SWI+B. These numbers represented a 66 reduction in oak SPA in the SW treatment. However, oak SPA increased 45 percent in the SWI treatment, 11.5 percent in the SW+B treatment

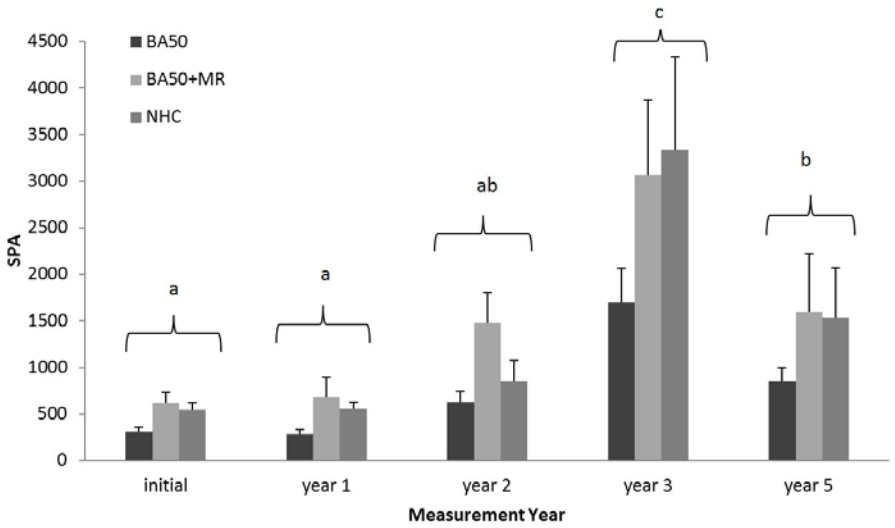


Figure 1A—Oak seedlings per acre by treatment by year (Means followed by same letter do not significantly differ, SNK, α = 0.05).

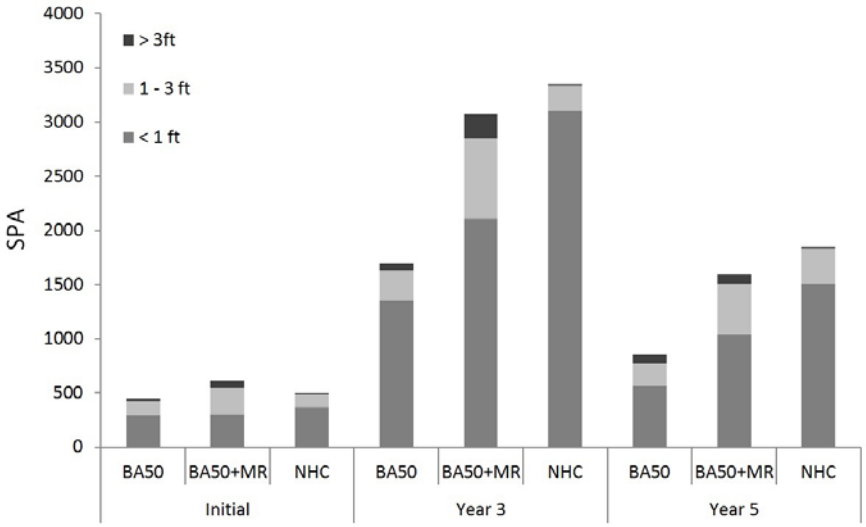


Figure 1B—Oak seedlings per acre by height class and treatment for initial, year 3 and year 5 post-treatment.

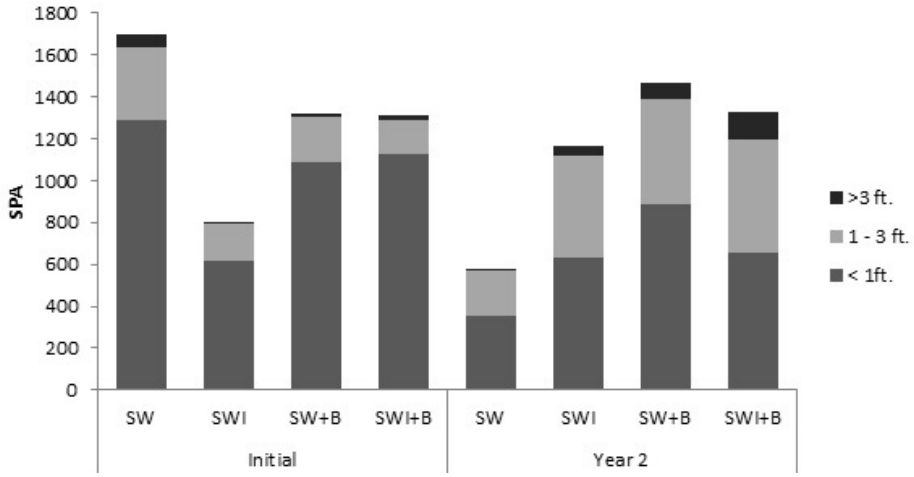


Figure 2—Oak SPA by Height Class for Waugh Mountain Initial Versus Year 2 Post-Treatment.

and 1 percent in the SWI+B treatment. No significant differences were detected between initial and year 2 values for total oak SPA by treatment (table 1).

Mean non-oak SPA < 1 foot in height decreased 72, 19, and 42 percent for SW, SW+B, and SWI+B, respectively. The SW+I experienced a 2 percent increase. Mean oak SPA 1 to 3 feet tall increased 177, 136, and 226 percent for SWI, SW+B, and SWI+B. However, oak SPA between 1 and 3 feet tall decreased 39 percent in the SW treatment. Mean oak SPA > 3 feet tall increased 1,106 (4 to 41 SPA), 425 (14 to 77) and 598 (18 to 129) percent for SWI, SW+B, and SWI+B treatments. The SW treatment experienced a 76 percent decline in oak SPA > 3 feet. No significant differences were observed between initial versus year 2 data by treatment for oak SPA < 1 ft. in height. A pairwise analysis detected significant differences between initial versus year 2 data for oak SPA 1 – 3 ft. and > 3 ft. for the SWI treatment ($p = 0.04$ and <0.001). No significant differences for oak SPA 1 – 3 ft. and > 3 ft. were observed for other treatments.

Year 2 post-treatment means were 4,793 (±710.5), 4,852 (±368.9), 4,437 (±747.6), and 5,625(±1,033.8) non-oak SPA for SW, SWI, SW+B, and SWI+B. These averages represented 36, 67, 7, and 120 percent increases for the respective treatments. There were no significant differences detected between initial and year 2 post-treatment average non-oak SPA for the SW and SW+B treatments. A pairwise analysis detected significant differences between initial and year 2 post-treatment average non-oak SPA for the SWI and SWI+B treatments (table 2).

A general increase occurred across most height classes for all treatments between initial versus year 2 non-oak SPA averages. Non-oak SPA less than 1 foot in height increased 59, 85, and 70.4 percent for SW, SWI and SWI+B treatments. However, SW+B experienced a 12 percent reduction in non-oak SPA less than 1 foot in height. Non-oak SPA from 1 to 3 feet tall increased 54, 50, and 59 percent for SW, SWI, and SW+B treatments. SWI+B experienced the highest change with a 301 percent increase for non-oak SPA between 1 and 3 feet tall. Non-oak SPA greater than 3 feet tall increased 45, 30, and 133 percent for SWI, SW+B, and SWI+B treatments. Non-oak SPA greater than 3 feet tall declined 40 percent in the SW treatment (fig. 3).

DISCUSSION

River Hills

Cunningham 2015 presented results from initial to year 3 post-treatment, demonstrating an increase in oak SPA for all treatments over the time period. Those results were significantly impacted by the resulting sunlight environments generated by the applied overstory and/or midstory treatment combination and a topographical effect. The continued results presented here described the impact of a prescribed fire conducted prior to the

Table 1—Initial Versus Year 2 Post-Treatment Oak SPA Analysis

Treatment	Initial	SE	Year 2	SE	p-value
SW	1,696	654.0	581	179	0.17
SWI	800	197.6	1,167	372.6	0.43
SW+B	1,316	481.9	1,467	775.7	0.78
SWI+B	1,311	429.9	1,325	444.4	0.98

SE = ± 1 standard error

Table 2—Mean Non-oak SPA comparisons for Initial versus Year 2 Values

Treatment	Initial	SE	Year 2	SE	p-value
SW	3,515	735.8	4,793	710.5	0.30
SWI	2,893	430.4	4,852	368.9	0.03
SW+B	4,122	205.3	4,437	747.6	0.71
SWI+B	2,559	155.2	5,625	1,033.8	0.04

SE = ± 1 standard error

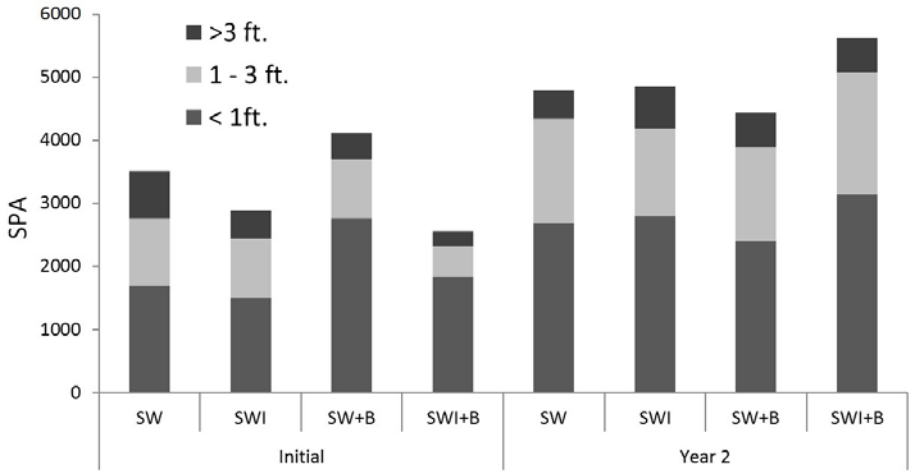


Figure 3—Initial Versus Year 2 Non-oak SPA by Height Class for the Waugh Mountain Study.

year 4 (2013) growing season. The results presented represented year 5 overall or year 2 post-burn data. Brose and others (2013) suggested that fires conducted several years after a stand disturbance often generated positive results in oak seedling abundance and were more effective than pre-harvest preparatory burns. The results from this study did not coincide with these findings. Oak SPA declined for all treatments between year 3 and year 5.

The authors believe there are several causative factors taking place. First, the NHC treatment was not burned and therefore fire had no effect on the reduction in oak SPA. The reduction in oak SPA in the NHC treatment was most likely a response to the natural flux of oak reproduction established from the bumper acorn crop of 2011 and the impacts of inadequate sunlight environments for those new germinates to perpetuate through time. However, low sunlight could not be a primary causative factor for BA50 or BA50+MR. In these treatments, where sunlight environments were at a minimum adequate for oak seedling survival and development (except for lower slope positions in BA50), the prescribed fire was likely a causative factor in the decline of oak SPA between year 3 and year 5. An analysis of the data suggests that the influx of new seedlings in year 3, followed by fire in the following dormant season resulted in seedling mortality of those new seedlings, returning oak SPA to near year 3 values. However, it should be further noted that year 3 was a year of excessive drought and the flux of new seedlings were stressed and not high-vigor in many cases. Thus additional factors may have also impacted the year 5 results.

Year 5 data did demonstrate that the sunlight environments established in the initial treatments were still having impact, with the mid-story removal treatments having the most oak reproduction in the taller height classes. Furthermore, visual observations of oak reproduction in BA50 and BA50+MR in year 5 suggest that these seedlings are physiologically active, have good leaf area indices, and are vigorous. The authors believe year 6 data will show a strong rebound in oak reproduction in BA50 and BA50+MR and predict a continued decline in oak SPA in the NHC.

Waugh Mountain

Statistically there were no differences between initial and year 2 values for any treatment in total oak SPA. The shelterwood only treatment (SW) served as the control treatment and did demonstrate expected and notable results. The SW treatment experienced a significant decline in oak SPA, 66 percent, which was the largest of any treatment applied. These reductions were likely attributable to inadequate sunlight environments from continued shading by residual midstory competition in the shelterwood only treatment.

When midstory removal was added to the shelterwood treatment (SWI) a different oak seedling response was observed. Oak SPA increased by 45 percent in the SWI treatment from 800 SPA to 1,167 SPA. This result suggests that the optimal sunlight conditions created by reducing midstory level competition positively impacted oak seedling abundance.

When prescribed fire alone was added to the shelterwood harvest (SW+B), oak seedling abundance increased slightly by 11 percent from 1,316 SPA to 1,467 SPA. A reduction occurred in seedlings < 1 foot in height likely due to seedlings moving into the taller height classes and possibly some mortality from the disturbance from harvesting and burning. While variation between replicates prevented significant differences in data analysis, notable increases in

seedling height occurred. The increases in height were likely a function of seedling sprouts from the burn treatment.

When midstory removal and prescribed fire were coupled with a shelterwood harvest (SWI+B), total oak SPA increased slightly. However, an appreciable increase in oak SPA in heights between 1 and 3 feet and greater than 3 feet occurred. This treatment had the greatest impact on the combination of survival and growth. The associated sunlight environment was viewed as the primary factor in this treatment. The apparent increase in seedling growth was likely a response from seedling sprouting in post-burn conditions, as in the SW+B treatment.

CONCLUSIONS

These two studies demonstrated the importance of creating optimal sunlight conditions to promote oak reproduction survival and development. The River Hills study demonstrated the impact of inadequate sunlight conditions in undisturbed stand conditions. Even with a significant flux of new oak reproduction appearing in year 3, evidence of that effect being short lived was observed in year 5. At both study locations, a shelterwood only harvest, with no additional treatments had limited impacts on oak reproduction abundance and growth. The areas in the shelterwood harvest where conditions were favorable did promote oak reproduction. The areas where the midstory remained intact remained inadequate sunlight environments for oak reproduction survival and development. Furthermore, midstory removal combined with a shelterwood harvest created the most uniform and favorable sunlight conditions for oak reproduction survival and development.

Prescribed fire appeared beneficial in the Waugh Mountain study, with appreciable increases in oak seedling survival and development. However, fire appeared to have negative effects on oak seedling survival at the river hills study. As stated, this negative effect could have been caused by the presence of new seedlings established prior to the burn application. In these two studies, a preparatory burn prior to midstory and overstory manipulation appeared to be more beneficial than a post-treatment burn. These results help supplement our current understanding of prescribed fire in oak regeneration. In application, forest managers should gain an understanding for positive and negative factors associated with prescribed fire in regard to their potential impact on oak regeneration success. Timing of prescribed fire applications to match existing stand and reproduction conditions is important to prescribed fire having beneficial effects on oak regeneration efforts.

LITERATURE CITED

Brose, P.H.; Van Lear, D.; Cooper, R. 1998. Using shelterwood harvests and prescribed fire to regenerate oak stands on productive upland sites. Forest Ecology and Management. 113: 125-141.

Brose, P.H.; Dey, D.C.; Phillips, R.J.; Waldrop, T.A. 2013. A meta-analysis of the fire-oak hypothesis: does prescribed burning promote oak reproduction in Eastern North America? Forest Science. 59(3): 332-334.

Cunningham, K.K. 2015. Third year response of oak natural regeneration to a shelterwood harvest and midstory control in the Arkansas Ozarks. In: Holley, A.G.; Connor, K.F.; Haywood, J.D., eds. Proceedings of the 17th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-203. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 22-28.

Cunningham, K.K.; Peairs, S.E.; Ezell, A.W [and others]. 2011. Understory light conditions associated with partial overstory removal and midstory/understory control: applications in a bottomland hardwood forest. Forests. 2(4): 984-992.

Ferguson, D.V.; Lowrance, J.S.; McFadden, C.E. 1982. Soil survey of Independence County, Arkansas. Fayetteville, AR: U.S. Department of Agriculture Soil Conservation Service, in cooperation with: Arkansas Agriculture Experiment Station. 148 p.

Foti, L.T. 2004. Upland hardwood forests and the related communities of the Arkansas Ozarks in: The early 19th century. In: Spetich, M.A., ed. Upland oak ecosystem symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 21-29.

Graney, D.L.; Bower, D.R. 1971. Site index curves for red and white oaks in the Boston Mountains of Arkansas. Res. Note SO-121. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 4 p.

Larsen, D.R.; Johnson, P.S. 1998. Linking the ecology of natural regeneration to silviculture. Forest Ecology and Management. 106(1): 1-7.

Loftis, D.L. 1990. A shelterwood method for regenerating red oak in the southern Appalachians. Forest Science. 36(4): 917-929.

Loftis, D.L. 1993. Regenerating northern red oak on high quality sites in the southern Appalachians. In: Loftis, D.L.; McGee, C.E., eds. Oak regeneration: serious problems practical recommendations (symposium proceedings). Gen.Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 202-210.

Soucy, R.; Heitzman, E.; Spetich, M. 2004. Age distribution of oak forests in North-central Arkansas. In: Spetich, M.A., ed. Upland oak ecosystem symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: US Department of Agriculture Forest Service, Southern Research Station: 53-61.

OAK WOODLAND RESTORATION IN THE MISSOURI OZARKS: TWO CASE STUDIES EXAMINING RESPONSES OF GROUND FLORA VEGETATION TO PRESCRIBED FIRE

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Absract—Prescribed fire and thinning are two primary tools for restoring overgrown oak and oak-pine woodlands in Missouri. We wanted to examine woodland restoration efforts and determine if we were meeting our goals of promoting herbaceous ground flora cover and richness. We examined herbaceous responses to fire at two restoration sites in the Missouri Ozarks. At the first research site (Fourche Creek), we compared ground flora responses following fall and spring burning. At the second site (Rocky Creek), we examined ground flora responses following harvest treatments and three prescribed fires. At Fourche Creek, herbaceous richness and cover was not different for all treatments prior to burning. In 2011 after two prescribed burns, both fall and spring burn treatments had higher herbaceous richness and cover when compared to the unburned control. Both burn treatments also had lower sapling density in 2011 when compared to the control. Despite distinct structural differences in the overstory at Rocky Creek, there were no differences in cover and richness of herbaceous ground flora pre- and post-burning. High variability within and among treatments for the herbaceous community at Rocky Creek was likely due to environmental factors not accounted for within this study. Overall, these two case studies present evidence that fire is sufficient to increase cover and richness of native ground flora species, and that herbaceous communities can vary greatly across large restoration units with varied topography and soil conditions.

INTRODUCTION

Wildland fire shaped the structure and composition of eastern oak woodlands of the United States prior to Euro-American settlement (Ryan and others 2013). Following intense exploitive harvesting and fire suppression in the early 20th century, woodland composition shifted towards mesic species with a subsequent decline in fire-tolerant species such as oak (*Quercus* spp.) and pine (*Pinus* spp.) (Nowacki and Abrams 2008). Recent efforts in the last two decades have aimed at using thinning and prescribed fire to halt the progression of mesophication and restore oak and oak-pine woodlands to pre-settlement conditions (McIver and others 2013).

In the Missouri Ozarks, pre-settlement woodlands were characterized as open with low densities of understory shrubs and a diverse and abundant ground flora (Batek and others 1999, Ladd 1991, Nelson 1997, Nigh 1992). Early travelers often noted the open woodland conditions with lush herbaceous understory free from underbrush (Houck 1908, Schoolcraft 1821), which was maintained by widespread fires ignited by American Indians (Guyette and others 2002). After decades of fire suppression, contemporary Ozark woodlands have increased in composition of fire-intolerant species and experienced dramatic increases in tree densities (Hanberry and others 2012, Hanberry and others 2014).

One objective for reintroducing fire to a landscape is to benefit native species that are promoted with periodic fires, thereby improving the resiliency of woodland natural communities to change caused by disturbance and climate change (Nelson 2005). Since the mid-1980s, natural resource agencies across Missouri began implementing prescribed fire in an effort to restore overgrown woodlands to their pre-settlement character (Hartman 2005, Neuwald and Templeton and others 2013). The main objectives of restoration efforts include controlling the woody understory and increasing the vigor and abundance of native ground flora (Hartman 2005, Nelson 2005). Projects on state owned land have used silvicultural treatments along with landscape-scale prescribed fire on three to five year intervals in an effort to restore open woodland conditions and stimulate growth of herbaceous cover (Hartman and Heumann 2003, Kinkead and others 2013, McMurray and others 2007). This study examines woodland restoration efforts at Fourche Creek and Rocky Creek Conservation Areas in the Missouri Ozarks.

OBJECTIVES

Our project focused on restoration efforts in an oak-dominated woodland at Fourche Creek Conservation Area (CA) and a pine-oak woodland at Rocky Creek CA. For the Fourche Creek project, we wanted answers for two questions: 1) Are there differences in

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herbaceous responses between early dormant season (fall) and late dormant season (spring) burning following two prescribed fires? and, 2) Do the fall and spring prescribed fires help control the woody understory? For the Rocky Creek project, we wanted to know if, 1) Following different harvest treatments, is there a difference in immediate ground flora response? and, 2) Will there be differences in ground flora responses following burning at sites with different structural characteristics?

STUDY AREA

The two study sites are located within the Ozarks Highland Ecoregion in Missouri (Nigh and Schroeder 2002). The Fourche Creek Project (FCHE) is located in Ripley County within the Oak Woodland Dissected Plain. This area is characterized by flat to gently rolling topography dominated by oaks (mainly *Quercus stellata* and *Q. velutina*) and hickories (*Carya* spp.). The Rocky Creek Project (RCKY) was established in Shannon County within the Current River Oak-Pine Woodland/ Forest Hills. This area features gently rolling dissected plains to rugged topography near major streams. Pine-oak woodlands generally dominate the landscape, especially on ridgetops and upper slopes. Dominant canopy species in this region include hickories (*Carya glabra, C. texana, C. tomentosa*), oaks (*Quercus alba*, *Q. coccinea*, *Q. stellata*, and *Q. velutina*), and shortleaf pine (*Pinus echinata*).

METHODS

We evaluated three prescribed fire treatments at FCHE: early dormant season burning (fall), late dormant season burning (spring), and no burn (control). We implemented the fall prescribed fires on 11/10/05 and 11/8/09. The spring burns occurred on 3/28/06 and 4/9/2010. The three treatment units ranged from 3.3 to 5.5 ha.

Managers initiated harvest treatments at the RCKY project in 2003 before researchers had a chance to collect baseline data, but we still wanted to determine if pre-burn overstory structure would affect herbaceous species response to fire. We evaluated four harvest treatments that occurred in 2003 and 2004: intermediate thinning with timber stand improvement (ITSI), shelterwood harvest (SW), uneven-aged management (UAM), and a non-commercial woodland thinning (NCW). We added a fifth treatment, pine-ITSI (PITSI), within the ITSI treatment because the overstory composition within this particular study area was predominantly shortleaf pine, whereas the other sites had higher proportions of oaks. See table 1 for more detailed descriptions of harvesting treatments. All harvest sites at RCKY are within the same burn unit (267 ha), and were burned in March 2006 (date unspecified), January 2009 (date unspecified), and 4/8/2011.

We established three circular permanent plots for each treatment. Within each 0.1 ha plot, we sampled all woody vegetation ≥ 10 cm DBH (overstory). Within a 0.01 ha subplot located at plot center, we measured all woody vegetation <10 cm DBH and ≥1.4 m tall (sapling). We sampled vegetation cover for herbaceous and woody species (<1.4 meters in height) in three 1-m^2 quadrats distributed at 5.64 meter intervals along the four cardinal directions (n=12).

For both studies we wanted to investigate differences in ground flora among all treatments before (2005) and after the application of fire (2011). Specifically, we wanted to determine if the treatments affected the richness (plot-level) and cover (quadrat-level) of herbaceous species. At FCHE, we also evaluated the density of saplings (<2.74 m in height) in 2005 and 2011. To examine overstory structure at RCKY, we evaluated basal area (m^2 /ha) and quadratic mean diameter (cm) in 2005 and 2011. All hypotheses were tested using one-way ANOVA, and means were compared using Fisher's least significant difference test. Because our sample size was small, we set $\alpha = 0.10$ to test for significant effects.

Table 1—Description of treatments at the Rocky Creek project (RCKY). ITSI = intermediate thinning with timber stand improvement. NCW = non-commercial woodland thinning. PITSI = intermediate thinning with timber stand improvement with a relatively high component of overstory shortleaf pine (*Pinus echinata*). SW = shelterwood. UAM = uneven-aged management

Treatment	Description
ITSI	Thinned to 13.8 m^2/ha
NCW	Cut all woody vegetation <15.2 cm DBH
PITSI	Thinned to 13.8 m^2/ha
SW	Retain 4.6 - 9.2 m^2/ha residual basal area
UAM	Group openings and thinning using Law and Lorimer (1989) method

RESULTS

Fourche Creek Project (FCHE)

Herbaceous richness—In 2005, prior to spring and fall burning, herbaceous richness ranged from 12 to 16 species per plot (fig. 1), although no significant difference was found among treatments (ANOVA; F-value = 0.75; p = 0.51). In 2011, herbaceous richness in burned treatments was significantly higher than in the control (ANOVA; F-value = 6.67; p = 0.03). For spring and fall burning treatments, there were 35 and 39 species per plot, respectively. For the control treatment, there were 24 species per plot.

Herbaceous cover—Prior to burning in 2005, there were no significant differences in herbaceous cover among treatments (fig. 2). Herbaceous cover during this sampling period was 5 percent for all treatments (ANOVA; F-value = 0.01; p = 0.99). In 2011, herbaceous cover in the burn treatments was significantly higher than in the control (ANOVA; F-value = 22.74; p < 0.001). Herbaceous cover was 13 percent for fall burning and 14 percent for spring burning, while cover in the control was 3 percent.

Sapling density—In 2005, the sapling density in spring burn sites was higher than in both control and fall burn sites (fig. 3). There were 15 stems per plot for the spring treatment, whereas the control and fall treatments each averaged 7 stems per plot (ANOVA; F-value = 12.80; p < 0.01). In 2011 the control had significantly higher sapling density than spring and fall treatments (ANOVA; F-value 14.18; p < 0.01). The control treatment had 28 stems per plot, in contrast to much lower stem density in spring (2) and fall (4) treatments.

Rocky Creek Project (RCKY)

Herbaceous richness—For herbaceous richness, there were no significant differences among harvest treatments for both 2005 (ANOVA; F-value = 0.67, p = 0.63) and 2011 (ANOVA; F-value = 0.64 p = 0.65), likely due to high amount of variability within treatments (fig. 4). In 2005, herbaceous species richness ranged from 5 species per plot for NCW to 14 species per plot for ITSI. In 2011, the NCW treatment again had the lowest species richness among all treatments (13), while the SW treatment had the highest species richness (22).

Herbaceous cover—For both sampling years, there were no significant differences among treatments for herbaceous cover (2005-ANOVA; F-value = 0.41; p = 0.80; 2011-ANOVA; F-value = 1.34; p-value = 0.33) (fig. 5). In 2005, herbaceous cover ranged from 1.7 percent to 4.3 percent at all sites. In 2011, herbaceous cover ranged from 2.3 percent to 8.9 percent.

Basal area—In 2005, basal area by treatment can be separated into two groups (fig. 6). The "low" basal area group contains the NCW (10.5 m^2/ha) and SW (4.8 m^2/ha) treatments. The basal areas for these treatments were significantly lower than the other three treatments (ANOVA; F-value 7.32; p < 0.01). The "high" basal group contains the other three treatments. Basal areas in this group were not significantly different from one another at 17.6 (PITSI), 17.9 (UAM), and 20.4 (ITSI) m^2/ha.

From 2005 to 2011 there were small changes in basal area (fig. 6). Basal area for SW (4.5 m^2/ha) was significantly lower than all other treatments (ANOVA; F-value =8.58; p < 0.01). Basal area (m^2/ha) for ITSI (20.4) was the highest, but it was not significantly different from PITSI (17.6) and UAM (14.6). The basal area for NCW increased to 12.7 m^2/ha in 2011, but it was not statistically different from PITSI or UAM.

Quadratic mean diameter (QMD)--Much like basal area in 2005, quadratic mean diameter (QMD; cm) can be separated into two groups (fig. 7). The "small" QMD group contains ITSI, PITSI, and UAM. QMDs in this group range from 21.1 cm to 22.5 cm. QMDs in this group are not significantly different from each other, but they are significantly smaller than the "large" group (ANOVA; F-value = 12.36; p < 0.001). The "large" QMD group contains NCW (29.6 cm) and SW (28.2 cm). The QMD for these two treatments were not significantly different from each other.

The QMD in 2011 differed among treatments (ANOVA; F-value = 6.67; p < 0.01). The SW and NCW QMDs were still the highest (31.7 cm and 30.1 cm, respectively), but unlike in 2005, NCW was not different from UAM (25.2 cm). As in 2005, there was no difference in QMD for ITSI (22.7 cm), PITSI (22.9 cm), and UAM (25.2 cm).

DISCUSSION

Fourche Creek (FCHE)

Many studies have shown that fire is effective at increasing herbaceous richness in oak and oak-pine woodlands of the eastern United States (Arthur and others 1998, Elliott and others 1999, Kinkead and others 2012, Masters 1991, McGee and others 1995, Nuzzo and others 1996, Phillips and others 2007, Sparks and others 1998, Taft 2003, Wilhelm and Masters 1994). Increasing herbaceous richness in the understory is a cornerstone objective in the goal of restoring oak and pine-oak woodlands in the Missouri (Hartman 2005, Nelson 2005). Our results confirm that fall and spring burning at Fourche Creek is effective at increasing herbaceous richness during oak woodland restoration.

It was no surprise that herbaceous cover was higher after fall and spring burning at FCHE when compared

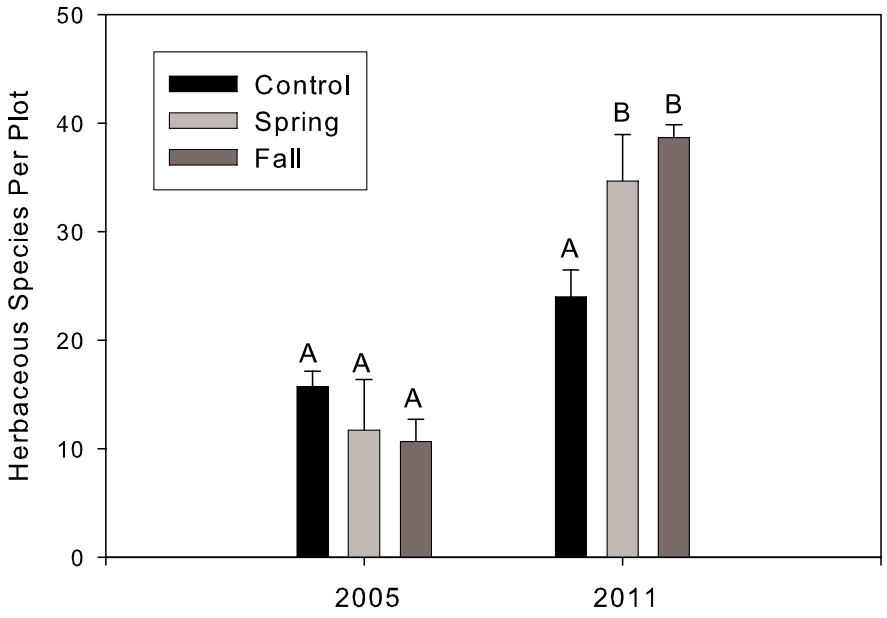


Figure 1—Herbaceous richness (mean + SE) at the Fourche Creek project (FCHE) in 2005 (pre-treatment) and 2011. The fall prescribed fires were implemented on 11/10/05 and 11/8/09. The spring burns occurred on 3/28/06 and 4/9/2010. Significant differences of means within years are denoted by different letters (α = 0.10).

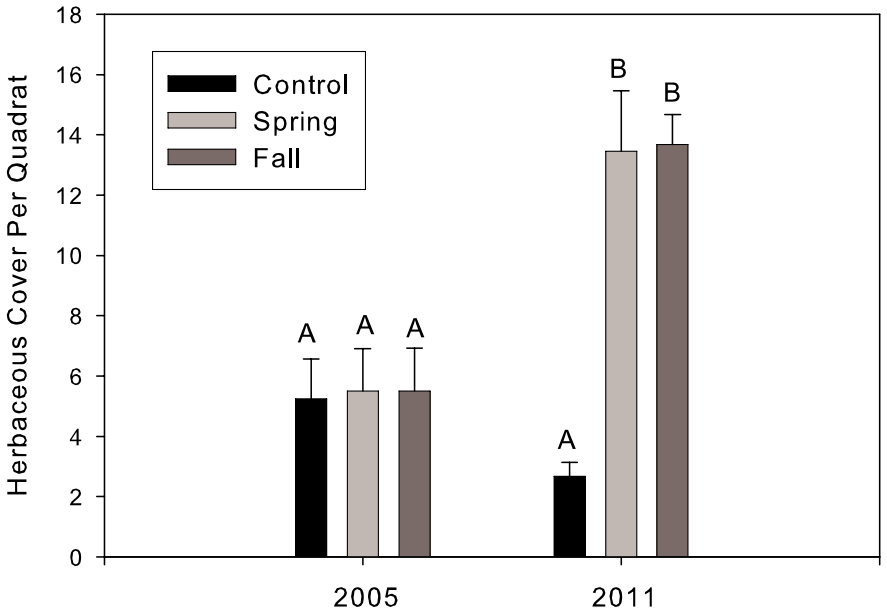


Figure 2—Herbaceous cover (mean + SE) at the Fourche Creek project (FCHE) in 2005 (pre-treatment) and 2011. The fall prescribed fires were implemented on 11/10/05 and 11/8/09. The spring burns occurred on 3/28/06 and 4/9/2010. Significant differences of means within years are denoted by different letters (α = 0.10).

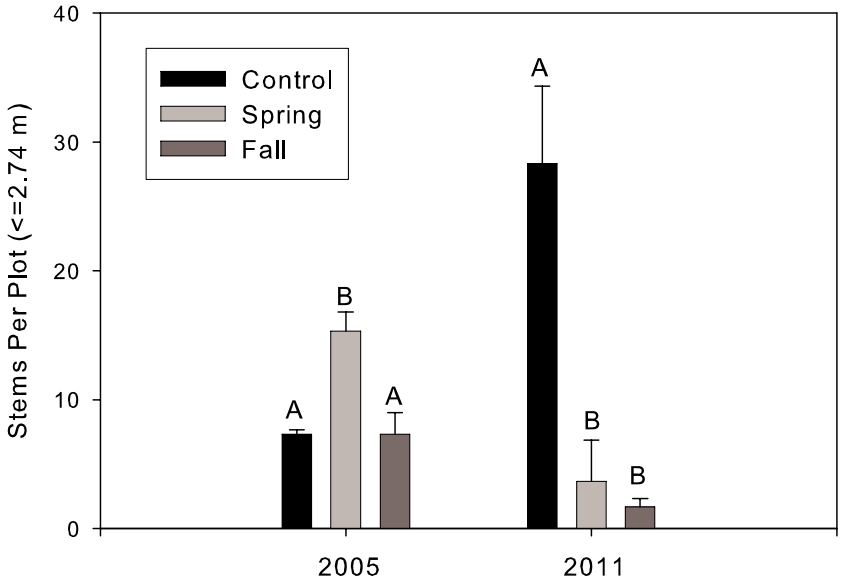


Figure 3—Sapling stem density (mean + SE) at the Fourche Creek project (FCHE) in 2005 (pre-treatment) and 2011. The fall prescribed fires were implemented on 11/10/05 and 11/8/09. The spring burns occurred on 3/28/06 and 4/9/2010. Significant differences of means within years are denoted by different letters (α = 0.10).

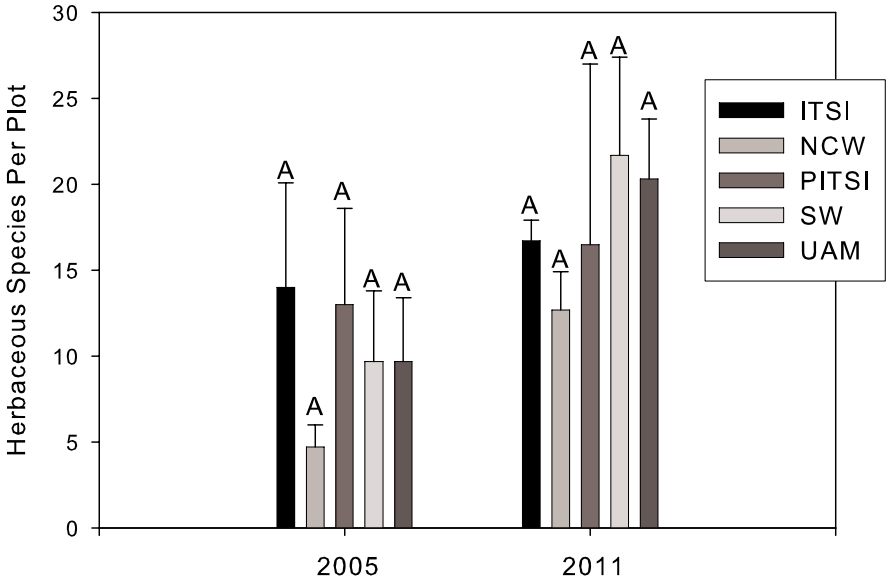


Figure 4—Herbaceous richness (mean + SE) at the Rocky Creek project (RCKY) in 2005 (post-harvest, pre-fire) and 2011. Harvesting occurred in 2003 and 2004. All treatment sites at RCKY are within the same burn unit, and were prescribed fire in March 2006 (date unspecified), January 2009 (date unspecified), and 8 April 2011. Significant differences of means within years are denoted by different letters (α = 0.10). ITSI = intermediate thinning with timber stand improvement. NCW = non-commercial woodland thinning. PITSI = intermediate thinning with timber stand improvement with a relatively high component of overstory shortleaf pine (*Pinus echinata*). SW = shelterwood. UAM = uneven-aged management.

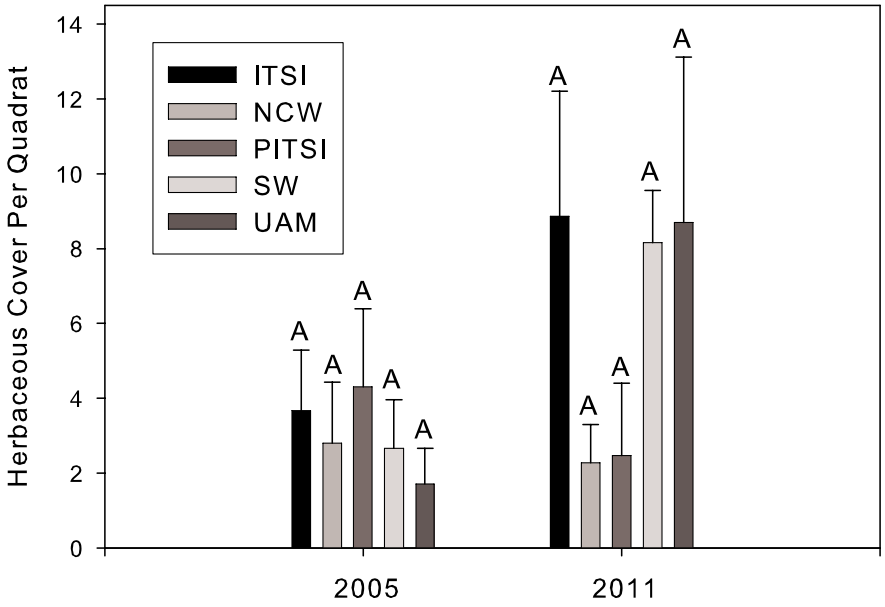


Figure 5—Herbaceous cover (mean + SE) at the Rocky Creek project (RCKY) in 2005 (post-harvest, pre-fire) and 2011. Harvesting occurred in 2003 and 2004. All treatment sites at RCKY are within the same burn unit, and were prescribed fire in March 2006 (date unspecified), January 2009 (date unspecified), and 8 April 2011. Significant differences of means within years are denoted by different letters (α = 0.10). ITSI = intermediate thinning with timber stand improvement. NCW = non-commercial woodland thinning. PITSI = intermediate thinning with timber stand improvement with a relatively high component of overstory shortleaf pine (*Pinus echinata*). SW = shelterwood. UAM = uneven-aged management.

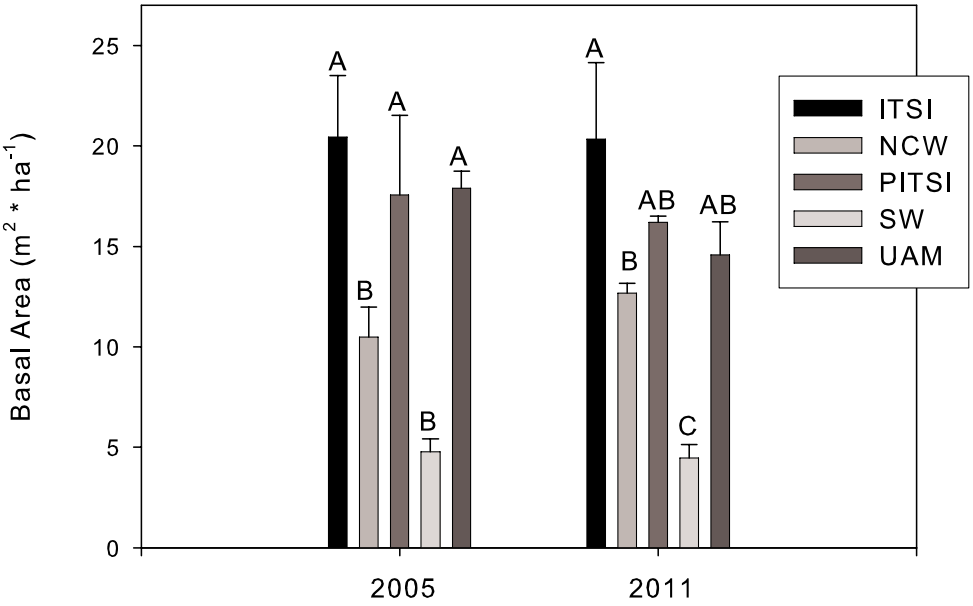


Figure 6—Basal area (mean + SE) at the Rocky Creek project (RCKY) in 2005 (post-harvest, pre-fire) and 2011. Harvesting occurred in 2003 and 2004. All treatment sites at RCKY are within the same burn unit, and were prescribed fire in March 2006 (date unspecified), January 2009 (date unspecified), and 8 April 2011. Significant differences of means within years are denoted by different letters (α = 0.10). ITSI = intermediate thinning with timber stand improvement. NCW = non-commercial woodland thinning. PITSI = intermediate thinning with timber stand improvement with a relatively high component of overstory shortleaf pine (*Pinus echinata*). SW = shelterwood. UAM = uneven-aged management.

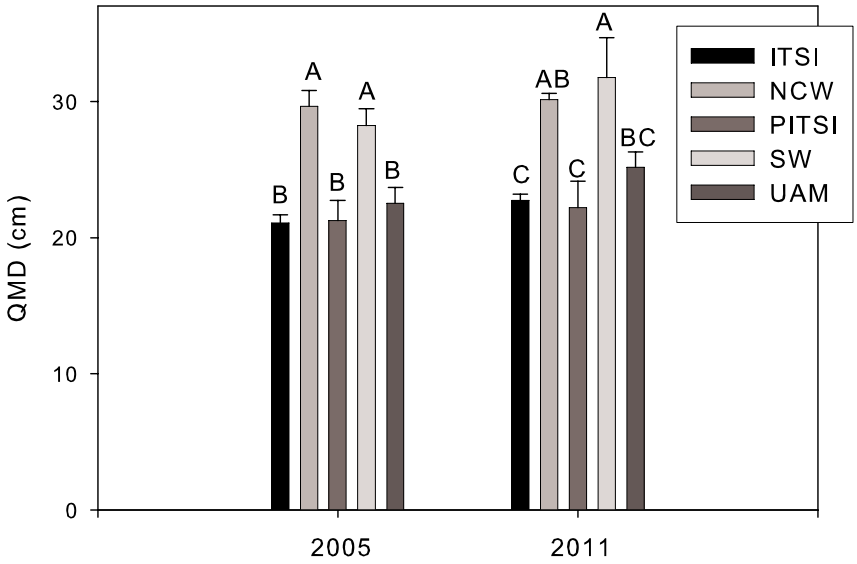


Figure 7—Quadratic mean diameter (mean + SE) at the Rocky Creek project (RCKY) in 2005 (post-harvest, pre-fire) and 2011. Harvesting occurred in 2003 and 2004. All treatment sites at RCKY are within the same burn unit, and were prescribed fire in March 2006 (date unspecified), January 2009 (date unspecified), and 8 April 2011. Significant differences of means within years are denoted by different letters (α = 0.10). ITSI = intermediate thinning with timber stand improvement. NCW = non-commercial woodland thinning. PITSI = intermediate thinning with timber stand improvement with a relatively high component of overstory shortleaf pine (*Pinus echinata*). SW = shelterwood. UAM = uneven-aged management.

to unburned sites. Numerous studies in oak dominated woodlands have shown that use of prescribed fire leads to increases in herbaceous ground flora cover (DeSelm and Clebsch 1991, Elliott and others 1999, Kuddes-Fischer and Arthur 2002, Masters 1991, Nuzzo 1996, Phillips and others 2007, Taft 2003). Results from other restoration sites in the Missouri Ozarks have also reported increased herbaceous cover following prescribed fire (Hartman and Heumann 2003, Kinkead and others 2013, McMurray and others 2007, Sasseen and Muzika 2004).

Top-killing of saplings causes an immediate decrease in woody cover following prescribed fires (Bacone and Post 1986, Glitzenstein and others 2003), but re-sprouting from species such as red maple (*Acer rubrum*) and sassafras (*Sassafras albidum*) often leads to increased density following prescribed fire (Ducey and others 1996, Phillips and others 2007). Studies in the Missouri Ozarks have also shown that fire is effective at top-killing saplings, and periodic burning leads to lower densities of saplings in burned oak and oak-pine woodlands (Blake and Schuette 2000, Hartman and Heumann 2003). Our results from FCHE confirm that multiple fires in spring or fall lead to lower density of saplings when compared to similar unburned sites. The relatively low density of saplings in the burned areas, along with the prescribed fires, likely influenced the herbaceous response (Hartman and Heumann 2003).

Rocky Creek (RCKY)

The findings from the RCKY project are difficult to interpret because of several problems with the study design. First, baseline data was not collected prior to harvest treatments. Second, control plots were not established outside of the burning area. And lastly, sampling plots were established across three different Ecological Landtypes (Nigh and Schroeder 2002) which include ultic chert uplands, exposed sandstone backslopes, and upland loess fragipans. It is reasonable to expect variability in herbaceous cover and richness ground flora within these different Ecological Landtypes due to variability in landforms, soil types, and geologic parent material.

One would posit that different structural differences in the overstory would lead to variability in herbaceous responses among treatments. Instead we found wide variability of herbaceous responses within and among treatments. As stated earlier, this could be attributed to location of plots within different Ecological Landtypes both within and among treatments.

For some harvest treatments, there did appear to be an increase in herbaceous richness and cover following

burning, although this was not statistically tested. As we said earlier, many studies have shown that prescribed fire is effective at stimulating ground flora, including our FCHE study. Although we did not see differences in cover and richness among harvest treatments, one might suggest that regardless of harvest treatment, prescribed fire may be effective at stimulating herbaceous ground flora.

CONCLUSION

Prescribed fire appears to be an effective tool for meeting common objectives of oak woodland restoration. Following multiple prescribed fires, we saw an increase in herbaceous richness and cover at two oak restoration sites in the Missouri Ozarks. Prescribed fire may also be effective at top-killing saplings, which likely enhances sunlight reaching the ground flora and further stimulates the growth of herbaceous species. Following harvest treatments and prescribed fire, variability within sites could not be accounted for with overstory metrics, while environmental factors may be driving variability in herbaceous responses within and among treatments. Future monitoring efforts will continue to track changes in vegetation responses following multiple fires.

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LITERATURE CITED

Arthur, M.A.; Paratley R.D.; Blankenship, B.A. 1998. Single and repeated fires affect survival and herbaceous species in an oak-pine forest. Journal of the Torrey Botanical Society. 125(3): 225-236.

Bacone, J.A.; Post, T.W. 1986. Effect of prescribed burning on woody and herbaceous vegetation in black oak sand savannas at Hoosier Prairie Nature Preserve, Lake County, Indiana. In: Koonce, A.L., ed. Proceedings, prescribed burning in the Midwest: state-of-the-art. Stevens Point, WI: University of Wisconsin, College of Natural Resources, Fire Science Center: 86-90.

Batek, M.J.; Rebertus, A.J.; Schroeder, W.A. [and others]. 1999. Reconstruction of early nineteenth-century vegetation and fire regimes in the Missouri Ozarks. Journal of Biogeography. 26(2): 397-412.

Blake, J.G.; Schuette, B. 2000. Restoration of an oak forest in east-central Missouri: early effects of prescribed burning on woody vegetation. Forest Ecology and Management. 139: 109-126.

DeSelm, H.R.; Clebsch, E.E.C. 1991. Response types to prescribed fire in oak forest understory. In: Nodvin, S.C.; Waldrop, T.A.; eds. Fire and the environment: ecological and cultural perspectives: proceedings of an international symposium. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 22-33.

Ducey, M.J.; Moser, W.K.; Ashton, P.M.S. 1996. Effect of fire intensity on understory composition and diversity in a *Kalmia*-dominated oak forest, New England, USA. Vegetatio. 123: 81-90.

Elliott, K.J.; Hendrick, R.L.; Major, A.E. [and others]. 1999. Vegetation dynamics after a prescribed fire in the southern Appalachians. Forest Ecology and Management. 114: 199-213.

Glitzenstein, J.S.; Streng, D.R.; Wade, D.D. 2003. Fire frequency effects on longleaf pine (*Pinus palustris* P. Miller) vegetation in South Carolina and northeast Florida, USA. Natural Areas Journal. 23(1): 22-37.

Guyette, R.P.; Muzika, R.M.; Dey, D.C. 2002. Dynamics of an anthropogenic fire regime. Ecosystems. 5: 472-486.

Hanberry B.B.; Dey, D.C.; He, H.S. 2012. Regime shifts and weakened environmental gradients in open oak and pine ecosystems. PLoS ONE. 7(7): e41337.

Hanberry, B.B.; Kabrick, J.M.; He, H.S. 2014. Densification and state transition across the Missouri Ozarks landscape. Ecosystems. 17(1): 66-81.

Hartman, G.W.; Heumann, B. 2003. Prescribed fire effects in the Ozarks of Missouri: the Chilton Creek Project 1996-2001. In: Proceedings of the second international wildland fire ecology and fire management congress. Boston, MA: American Meteorological Society: 12 p.

Hartman, G.W. 2005. Forest land management guide: use of prescribed fire. Jefferson City, MO: Missouri Department of Conservation: 9 p.

Houck, L. 1908. A history of Missouri, from the earliest explorations and settlements until the admission of the state into the union. 3 volumes. Chicago, IL: R.R. Donnely & Sons Co. 1255 p.

Kinkead, C.O.; Kabrick, J.M.; Stambaugh, M.C.; Grabner, K.W. 2013. Changes to oak woodland stand structure and ground flora composition caused by thinning and burning. In: Miller, G.W., Schuler, T.M.; Gottschalk, K.W. [and others] eds. Proceedings of the 18th central hardwood forest conference. Gen. Tech. Rep. NRS-P-117. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 373-383.

Kuddes-Fischer, L.M.; Arthur, M.A. 2002. Response of understory vegetation and tree regeneration to a single prescribed fire in oak-pine forests. Natural Areas Journal. 22(1): 43-52.

Ladd, D. 1991. Reexamination of the role of fire in Missouri oak woodlands. In: Burger, G.V.; Ebinger, J.E.;Wilhelm, G.S., eds. Proceedings of the oak woods management workshop. Charelston, IL: Eastern Illinois University: 67-80.

Law, J.R.; Lorimer, C.G. 1989. Managing uneven-aged stands. In: Clark, F.B.; Hutchinson, J.G., eds. Central Hardwood Notes. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station: 6 p.

Masters, R.E. 1991. Effects of fire and timber harvest on vegetation and cervid use on oak-pine sites in Oklahoma Ouachita Mountains. In: Nodvin, S.C.; Waldrop, T.A., eds. Fire and the environment: ecological and cultural perspectives: proceedings of an international symposium. Gen. Tech. Rep. SE-69. Asheville, NC. U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 168-176.

McGee, G.G.; Leopold, D.J.; Nyland, R.D. 1995. Understory response to springtime prescribed fire in two New York transition oak forests. Forest Ecology and Management. 76: 149-168.

McIver, J.D.; Stephens, S.L.; Agee, J.K. [and others]. 2013. Ecological effects of alternative fuel-reduction treatments: highlights of the National Fire and Fire Surrogate study (FFS). International Journal of Wildland Fire. 22: 63-82.

McMurry, E.R.; Muzika, R.M.; Lowenstein, E.F. [and others]. 2007. Initial effects of prescribed burning and thinning on plant communities in the southeast Missouri Ozarks. In: Buckley, D.S.; Clatterbuck, W.L., eds. Proceedings of the 15th central hardwood forest conference. Gen. Tech. Rep. SRS-101. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 241-249.

Nelson, J.C. 1997. Presettlement vegetation patterns along the 5th principal meridian, Missouri Territory, 1815. American Midland Naturalist. 137(1): 79-94.

Nelson, P.W. 2005. The Terrestrial Natural Communities of Missouri. Jefferson City, MO: Missouri Natural Areas Committee. 550 p.

Neuwald, J.L.; Templeton, A.R. 2013. Genetic restoration in the eastern collared lizard under prescribed woodland burning. Molecular Ecology. 22(14): 3666-3649.

Nigh, T. 1992. The forests prior to European settlement. In: Journet, A.R.P.; Spratt, H.G., eds. Proceedings: towards a vision for Missouri's public forest. Cape Girardeau, MO: Southeast Missouri State University: 6-13.

Nigh, T.A.; Schroeder, W.A. 2002. Atlas of Missouri Ecoregions. Jefferson City, MO: Missouri Department of Conservation. 212 p.

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the eastern United States. BioScience. 58: 123-138.

Nuzzo, V.A.; McClain, W.; Strole, T. 1996. Fire impact on groundlayer flora in a sand forest 1990-1994. American Midland Naturalist. 136(2): 207-221.

Phillips, R.; Hutchinson, T.; Brudnak, L.; Waldrop, T. 2007. Fire and fire surrogate treatments in mixed-oak forests: effects on herbaceous layer vegetation. In: Butler, B.W.; Cook, W., eds. 2007. Proceedings: the fire environment—innovations, management, and policy. RMRS-P-46CD. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 475-485.

Ryan, K.C.; Knapp, E.E.; Varner, J.M. 2013. Prescribed fire in North American forests and woodlands: history, current practice, and challenges. Frontiers in Ecology and the Environment. 11: e15-e24.

Sasseen, A.N.; Muzika, R.M. 2004. Timber harvesting, prescribed fire and vegetation dynamics in the Missouri Ozarks. In: Yaussy, D.A., Hix, D.M.; Long, R.P.; Goebel, P.C., eds. Proceedings of the 14th central hardwood forest conference. Gen. Tech. Rep. NE-316. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station: 179-192.

Schoolcraft, H.R. 1821. Journal of a tour into the interior of Missouri and Arkansaw performed in the years 1818 and 1819. London, England: Phillips and Co. 91 p.

Sparks, J.C.; Masters, R.E.; Engle, D.M. [and others]. 1998. Effects of late growing-season and late dormant-season prescribed fire on herbaceous vegetation in restored pine-grassland communities. Journal of Vegetation Science. 9: 133-142.

Taft, J.B. 2003. Fire effects on community structure, composition, and diversity in a dry sandstone barrens. Journal of the Torrey Botanical Society. 130(3): 170-192.

Wilhelm, G.; Masters, L. 1994. Floristic changes after five growing seasons in burned and unburned woodland. Erigenia. 13: 141-150.

Nutrition

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SOIL FERTILITY ASSESSMENT IN THE 3-PG MODEL USING SITE INDEX IN THE SOUTHEASTERN UNITED STATES

Santosh Subedi and Thomas R. Fox[1]

Soil fertility is one of the most important, yet least understood aspects of forest ecosystems. Study of soil fertility in forest ecosystems is complicated by the complex relationship between soil properties and stand productivity and immense variability in properties and characteristics of soils within relatively small geographic areas. Furthermore, the deep rooting systems of trees, which are capable of exploring soil throughout the soil profile, complicates the relationship between soil properties and stand productivity (Landsberg and others 2003, Dye and others 2004). Due to these complexities, soil fertility in forest ecosystems is measured by indirect approaches namely phytocentric and geocentric. The phytocentric approach uses tree based metrics to estimate soil fertility, while the geocentric approach uses the physical site characteristics as a relative indicator of soil fertility.

Site quality determines the potential productivity of forest ecosystem and site quality to a large extent is a function of the physical, chemical, and biological properties of soil as modified by anthropogenic factors (Augusto and others 2002). Many traditional growth and yield models and process-based models require some quantitative information about site quality / soil fertility to predict growth, yield, and physiological outputs of crop species. Site index is a realized measure of site quality and is widely used in many traditional growth and yield models as a driver of productivity (Burkhart and Tomé 2012). Similarly, several geocentric approaches using soil physical properties, topographic features, and climatic characteristics have attempted to quantify site quality/soil fertility (Carmean 1975, Fontes and others 2003, Hagglund 1981, Sampson and others 2008).

Quantification of soil fertility in forested ecosystems has important implications for modeling forest productivity. Knowledge of soil fertility is equally important for making fertilizer decisions and estimating fertilizer response. A key question arising from previous research is how can site index, a widely used indicator of site quality in forested ecosystem, be used as a potential tool for soil fertility estimation. Specifically, site index is a realized measure of site quality that is affected by not just soil fertility but a variety of edaphic and climatic factors including soil moisture and temperature. However, previous studies on loblolly pine have suggested that soil moisture is not a strong determinant of productivity in the southeastern US as this region receives fairly well distributed rainfall through out the year and annual precipitation exceeds potential evapotranspiration.

The 3-PG model (Physiological Principles Predicting Growth) has been parameterized and tested on many commercially important species throughout the globe (Almeida and others 2004, Stape and others 2004, Landsberg and others 2001, Bryars and others 2013, Rodŕiguez and others 2002, Coops and others 2005, Landsberg and Sands 2011). Past results with the 3-PG model have shown that 3-PG can be used to predict productivity with a limited number of input variables and unique parameter sets for each species.

3-PG is a monthly time-step process-based model. It requires 40 species-specific parameters. Additionally, 3-PG requires three sets of input variables: climatic data, stand initialization data, and site-specific data. Climatic data includes monthly averages of maximum temperature, minimum temperature, vapor pressure deficit, and radiation. Stand initialization data consists of initial stocking and initial biomass of seedlings. Site-specific data includes available soil water, soil textural class, and fertility rating. The 3-PG model has four biological sub-models to model C assimilation, C allocation, stand dynamics, and evapotranspiration. Biomass pools and stand density are the primary outputs from 3-PG. Fertility rating is a site-specific variable in 3-PG that describes soil nutrient status on a scale of 0 to 1. FR is an important variable in 3-PG that affects leaf area index and hence absorbed photosynthetically active radiation (APAR), canopy light use efficiency, and canopy conductance.

We developed a method to predict the soil fertility rating in the model 3-PG for loblolly pine plantations based on the relationship between stand productivity and site index. Then FR was used in 3-PG to predict loblolly pine yield and mortality on 21 sites across the southeastern

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United States. When observed yield and stem number were compared against the simulated values, 89 percent of the variation in yield and 89 percent of the variation in stand density were explained by simulated values. The USDA NRCS SSURGO dataset contains site index values for loblolly pine for the major soil series in most of the counties in the southeastern US. We observed good correlation between site index reported in SSURGO database and site index values observed in field inventory across major soil series in the southeastern US. When site index values from SSURGO dataset were used to derive FR values to predict loblolly pine productivity at a regional scale, the 3-PG model provided realistic outputs of loblolly pine productivity.

LITERATURE CITED

Almeida, A.C.; Landsberg, J. J.; Sands, P. J. 2004. Parameterization of 3-PG model for fast-growing Eucalyptus grandis plantations. Forest Ecology and Management. 193: 179-195.

Augusto, L.; Ranger, J.; Binkley, D.; Rothe, A. 2002. Impact of several common tree species of European temperate forests on soil fertility. Annuals of Forest Science. 59: 233-253.

Bryars, C.; Maier, C.; Zhao, D. [and others]. 2013. Fixed physiological parameters in the 3-PG model produced accurate estimates of loblolly pine growth on sites in different geographic regions. Forest Ecology and Management. 289: 501-514.

Burkhart, H.E.; Tomé, M. 2012. Modeling Forest Trees and Stands. The Netherlands: Springer. 457 p.

Carmean, W.H. 1975. Forest site quality evaluation in the United States. Advances in Agronomy. 27: 209-269.

Coops, N.C.; Waring, R.H.; Law, B.E. 2005. Assessing the past and future distribution and productivity of ponderosa pine in the Pacific Northwest using a process model, 3-PG. Ecological Modelling. 183: 107-124.

Dye, P.J.; Jacobs, S.; Drew, D. 2004. Verification of 3-PG growth and water-use predictions in twelve Eucalyptus plantation stands in Zululand, South Africa. Forest Ecology and Management. 193: 197-218.

Fontes, L.; Tome, M.; Thompson, F. [and others]. 2003. Modeling the Douglas-fir (*Psuedotsuga menziesii* (Mirb.) Franco) site index from site factors in Portugal. Forestry. 76: 491-507.

Hagglund, B. 1981. Evaluation of forest site productivity. Forestry Abstracts. 42: 515-527.

Landsberg, J.; Sands, P. 2011. Physiological ecology of forest production-principles, processes and models. Vol. 4. Amsterdam: The Netherlands: Elsevier. 331 p.

Landsberg, J.; Waring, R.; Coops, N. 2003. Performance of the forest productivity model 3-PG applied to a wide range of forest types. Forest Ecology and Management. 172: 199-214.

Landsberg, J.J.; Johnson, K.H.; Albaugh, T.J. [and others]. 2001. Applying 3-PG, a simple process-based model designed to produce practical results, to data from loblolly pine experiments. Forest Science. 47: 43-51.

Rodŕiguez, R.; Espinosa, M.; Real, P.; Inzunza, J. 2002. Analysis of productivity of radiata pine plantations under different silvicultural regimes using the 3-PG process-based model. Australian Forestry Journal. 65: 165-172.

Sampson, D.A.; Wynne, R.H.; Seiler, J.R. 2008. Edaphic and climatic effects on forest stand development, net primary production, and net ecosystem productivity simulated for Coastal Plain loblolly pine in Virginia. Journal of Geophysical Research. 113: 1-14.

Stape, J. L.; Ryan, M.; Binkley, D. 2004. Testing the utility of the 3-PG model for growth of *Eucalyptus grandis x urophylla* with natural and manipulated supplies of water and nutrients. Forest Ecology and Management. 193: 219-234.

Fire

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CONSUMPTION AND REACCUMULATION OF FOREST FUELS IN OAK SHELTERWOOD STANDS MANAGED WITH PRESCRIBED FIRE

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Abstract—In the shelterwood-burn technique, a moderate- to high-intensity growing-season prescribed fire is essential to achieve desired oak regeneration goals. These levels of fire intensity are dependent on the increased fuel loadings created by the preceding first removal cut. However, the loadings of forest fuels and their fluctuation during implementation of the shelterwood-burn technique have not been studied. From 1994 to 1997, three mature uncut oak stands, three oak shelterwoods, and three burned oak shelterwoods in central Virginia were examined to determine fuel loadings before, during, and after implementation of the shelterwood-burn technique. Prior to the first removal cut, total fuel loadings averaged 10.5 tons/acre and this was evenly divided between small and large fuels. The harvest increased total fuel loadings to between 20 and 25 tons/acre with nearly all this increase occurring in medium and large fuels. The prescribed burns consumed virtually all the small fuels and about half the medium/large fuels, but loadings for all size classes were 80 percent of pre-treatment levels within 3 years after the fires. It appears that forest fuel loadings in oak shelterwoods return to their initial levels within 5 years post-fire.

INTRODUCTION

Throughout the eastern United States, resource managers are increasingly using prescribed fire to regenerate, restore, or sustain upland oak (*Quercus* spp.) forests (Dickinson 2006, Hutchinson 2009, Yaussy 2000). A popular regeneration method to use when a sufficient density of oak seedlings exists is to proceed with stand renewal via the shelterwood-burn technique (Brose and others 1999a, b). This technique entails integrating a moderate to high intensity prescribed fire between the first and final removal cuts of a two-step shelterwood sequence. Because of the harvest – burn – harvest order of this method, an understanding of the forest fuels that will carry the fire and help give it the required intensity is necessary to safely and successfully implement this technique. However, information on the consumption and re-accumulation of forest fuels in upland oak stands is limited, especially those partially harvested or prescribe burned.

Much of the early oak forest fuel research was conducted in Missouri. Loomis and Crosby (1968, 1970) reported that a mixed oak – shortleaf pine (*Pinus echinata*) stand contained 5.0 to 6.5 ton/acre of fuel on its forest floor at age 21. Control of hardwoods via aerial herbicide application increased the fuel loading to approximately 10.4 tons/acre five years after the spraying. The increase was almost entirely in forest fuels that were less than 3 inches in diameter. Crosby and Loomis (1974) and Loomis (1975) followed the annual accumulation and decomposition of leaf litter in a 40-year-old oak stand. They showed that (1) litter fall added approximately 2 tons/acre/year to the fuel loading; (2) leaf litter loadings were largest in November (6.2 tons/acre); and (3) smallest in September (4.2 tons/acre). They also showed that the decay rate was uneven; the freshly fallen leaves of November did not substantially decompose until the following summer, then they decayed quite rapidly.

More recently, research into oak forest fuels has involved the eastern replicates of the National Fire and Fire Surrogate (NFFS) project. In North and South Carolina, several studies reported that forest fuel loadings in mature, undisturbed oak forests averaged approximately 6.0 tons/acre and fluctuated little over time (Phillips and Waldrop 2013, Waldrop and others 2004, 2007). Thinning decreased the leaf litter loading via mechanical disturbance and increased loading of small woody fuels, i.e., less than 1 inch diameter. Loading of large woody fuels, i.e., more than 3 inch diameter, was not affected because the treatment was a pre-commercial thinning. Prescribed fire alone and as a follow-up treatment to the thinning decreased leaf litter and small woody fuel loadings, but had no effect on large woody fuel loading. At the southern Ohio site of the NFFS study, Graham and McCarthy (2006) reported that a commercial thinning increased loading of all forest fuel categories and the subsequent burning reduced just the leaf litter and small woody fuel loadings. However, those reductions were no longer evident by 3 years post-fire. In a Missouri study

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closely related to the NFFS project, Kolaks and others (2004) found that thinning decreased leaf litter and small woody fuel loadings, increased large woody fuel loading, and resulted in an overall increase in total fuel loadings from 8.0 to 25.0 tons/acre. A subsequent prescribed fire in the thinned oak stands reduced total fuel loading to approximately 18.0 tons/acre. Nearly all the reduction occurred in the leaf litter and small woody fuel.

Also in the southern Appalachian Mountains, the Coweeta Hydrologic Laboratory in North Carolina has examined numerous understory burns for a myriad of ecological effects including reductions of forest fuels. Clinton and others (1998) reported a 50-percent reduction in leaf litter and small woody fuel loadings following such a fire in a mixed oak – pine stand. Understory burning in mixed oak – pine stands on dry and mesic upland sites resulted in reductions of leaf litter and small woody fuel loadings of 28 and 14 percent, respectively (Clinton and Vose 2007). Another understory burning study found 82 to 91 percent of the Oi layer and 26 to 46 percent of the Oe + Oa layers consumed by dormant-season prescribed fires (Knoepp and others 2009).

This paper addresses four forest fuel questions important to resource managers interested in using the shelterwood-burn technique to regenerate upland oak forests. Those questions are:

1. What are the fuel loadings in upland oak stands that are ready to begin the shelterwood – burn regeneration process?
2. How much fuel does the first removal cut create?
3. How much fuel does the prescribed fire consume?
4. How quickly do fuel loadings re-accumulate postfire?

The answers to these four questions will help forest managers make better use of the shelterwood-burn technique to regenerate upland oak forests.

METHODS

Study Sites

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

At each of three sites (Dunnavant, Lake Road, and Ward Farm), two adjacent mixed-oak stands (one undisturbed and one shelterwood) were selected in 1994 for the study. Each of the six stands was 20 to 50 acres in size and was on productive upland soils; oak site index$_{50}$ ranged from 70 to 80 feet. The undisturbed stands were mature (~100-year-old), fully stocked (basal areas ranged from 110 to 130 ft^2/acre), and dominated (67 percent of the basal area) by four upland oak species [black (*Quercus velutina*), northern red (*Q. rubra*), scarlet (*Q. coccinea*), and white (*Q. alba*)]. Other hardwoods present in these stands included American beech (*Fagus grandifolia*), blackgum (*Nyssa sylvatica*), flowering dogwood (*Cornus florida*), hickory (*Carya* spp.), red maple (*Acer rubrum*), and yellow-poplar (*Liriodendron tulipifera*). According to VDGIF records, the three uncut stands suffered some light ice storm damage in 1989; otherwise they had been undisturbed for at least 30 years. The three shelterwoods had been partly harvested between 1990 and 1992 as a result of moderate levels of ice storm damage. The harvests removed the poor-quality oaks and the low-value species resulting in average residual basal areas of 50 ft^2/acre in each shelterwood. Slash was left in place following the harvesting.

Sampling Procedures

Each of the shelterwood stands was divided into two units with one unit being burned while the other remained unburned. These two shelterwood units in conjunction with the adjacent undisturbed stand formed a chronosequence of the treatments (uncut, shelterwood, and burned shelterwood) that constitute the shelterwood-burn technique.

In each treatment, 15 to 45 fuel inventory transects were systematically installed to uniformly sample the area. Each transect was 50 feet in length and was inventoried for woody fuels using the planar-intersect method (Brown 1974). Sound woody fuels (those not in an advanced state of decay) were tallied by the time-lag size classes of Fosberg (1970). One-hour (1-hr) fuels were less than 0.25 inch diameter while ten-hour (10-hr) fuels were 0.25 to 1.0 inch diameter. These two size classes were tallied along the proximal 6-foot section of each transect. Hundred-hour (100-hr) fuels were 1.0 to 3.0 inch diameter and were tallied along the distal 12-foot section of each transect. Thousand-hour (1000-hr) fuels were 3.0 to 8.0 inch diameter and were tallied along the entire length of each transect. Rotten woody fuels in all size classes and fuels larger than 8 inch diameter were ignored because it was felt that these would not appreciably contribute to fire spread and fire intensity. For each transect, the woody fuel counts in each time-lag class were converted to tons/acre using established fuel equations (Brown 1974) and hardwood specific gravities (Anderson 1978). For each treatment, a mean fuel loading for each time lag class was calculated by summing the appropriate loading of each transect and dividing by the number of transects. Similarly, a mean total fuel loading for each treatment was calculated by summing all the time-lag fuel

loadings of each transect and dividing by the number of transects.

Leaf litter loading was measured by collecting two 1.36-ft^2 samples per transect of the Oi and Oe horizons. Samples were collected near the midpoint and distal end of each transect. The samples included the litter and the woody material smaller than 1.0 inch diameter found among the leaves. Samples were dried at 122° F for 72 hours then weighed on an electronic scale to the nearest 0.1 ounce. The masses of the two leaf litter samples from each transect were averaged and the average was converted to tons/acre.

Each transect was inventoried for leaf litter and woody fuels during winter 1994. Following that initial inventory, the portion of the each shelterwood designated for burning was burned by VDGIF personnel in accordance to department policy and state law. The prescribed fires were not all conducted at the same time because they were part of an oak regeneration study (Brose and Van Lear 1998). Rather, they were conducted in different seasons during 1995 (table 1). All prescribed fires were lit using drip torches in a strip headfire ignition pattern commencing at the downwind side of the burn block. Generally, burning conditions were good to excellent with clear skies, dry fuels, low humidity, steady winds, and typical seasonal temperatures (table 1). Fire behavior was most active in the spring burn treatment with headfire flame lengths ranging from 3 to 6 feet and rates-of-spread averaging 4 feet/minute. The summer and winter fires were quite similar; headfire flame lengths ranged from 1 to 3 feet and rates-of-spread averaged 1.5 feet/minute. Subsequent fuel inventories occurred in September 1995 and September 1997.

Statistical Analysis

The fuels data were grouped into three classes (small, medium, and large) based on their sizes and how they affect fire behavior. Small fuel included leaf litter, 1-hr, and 10-hr woody fuels as these primarily affect the rate-of-spread of a fire. Large fuel was the 1000-hr woody fuels as these principally affect fire intensity. Medium fuel was the 100-hr woody fuel as these contribute to both fire intensity and rate-of-spread. For the rest of this paper, the forest fuels will be referred to by these three classes.

The data were analyzed as a randomized complete block using repeated measures analysis of variance (ANOVA). Each site served as a block and each block contained three treatments (uncut, shelterwood, and burned shelterwood). Additionally, there were three years of data, 1994 (preburn), 1995, and 1997, for each treatment in each block. In the ANOVA model, treatment and year were the fixed effects and block was the random effect. Dependent variables were the loadings (tons/acre) of the small, medium, and large forest fuels. The Student-Newman-Kuels (SNK) means separation test was used to determine if significant differences existed among the nine treatment-year combinations for each of the dependent variables. An alpha of 0.05 was used for all tests and residuals were checked to ensure compliance with the statistical assumptions of an repeated measures analysis of variance.

RESULTS

Initial analysis of the 1994 fuel loadings found only minor differences among the three blocks for the three treatments. Furthermore, changes in fuel loadings through time for the three treatments were consistent, regardless of block. Therefore, data were pooled across blocks to ease reporting.

At the beginning of the study (1994), total fuel loading in the uncut treatment averaged 10.5 tons/acre (table 2). This loading was concentrated in the small (4.1 tons/acre) and large fuels (4.9 tons/acre) with the remaining 1.5 tons/acre being in the medium size class. In the shelterwood and burned shelterwood treatments in 1994, the average total fuel loading ranged from 22.6 to 24.4 tons/acre with no differences detected between the two treatments for any of the fuel classes. However, this total loading was more than double that of the uncut control due to a 5-fold increase in 100-hr loading and a 1.25x increase in 1000-hr loading. Small fuel loading did not differ among the three treatments; it averaged 4.5 tons/acre.

After the prescribed fires in 1995, several significant differences in fuel loadings were detected among the three treatments (table 2). Total fuel loadings were greatest in the shelterwood (24.4 tons/acre) while total fuel loadings in the uncut and the burned shelterwood were comparable (10.7 and 9.0 tons/acre, respectively). For the large fuels, the shelterwood had more loading (12.2 tons/acre) than the burned shelterwood (6.1 tons/acre) and the uncut control (4.9 tons/acre). The same pattern of distribution among treatments was found for the medium fuels; the shelterwood had the most loading (7.9 tons/acre) followed by burned shelterwood (2.1 tons/acre) and the uncut control (1.5 tons/acre). Small fuel loadings were equal between the uncut control and the shelterwood, each was 4.3 tons/acre, and this was larger than the small fuel loading of the burned shelterwood treatment (0.8 tons/acre). Relative to the previous year, the total fuel loading and its distribution among the fuel classes were unchanged in the uncut control and the shelterwood treatments. However, the fuel loading in the burned shelterwood treatment had decreased from 23.0 to 9.0 tons/acre. The decrease occurred in all fuel size classes; large fuel loading dropped 45 percent, from 11.1 to 6.1 tons/acre, medium fuel loading declined 69 percent, from 6.8 to 2.1 tons/acre, and the small fuel loading was reduced from 5.1 to 0.8 tons/acre, an 84 percent loss.

Table 1—Dates, times, and environmental conditions of the Winter, spring and summer prescribed fires in the oak shelterwood stands at the Dunnavant, Lake Road, and Ward Farm study sites

Season/Condition	Dunnavant	Lake Road	Ward Farm
Winter Prescribed Fire			
Burn Date	25Feb1995	27Feb1995	27Feb1995
Time of Burn	1300	1100	1430
Air Temperature (°F)	46	43	48
Relative Humidity (%)	26	62	54
Wind Direction tab	NW	E	E
Wind Speed (mph)	4	1	2
Cloud Cover (%)	0	100	100
1-HR Fuel Moisture (%)	6	12	12
10-HR Fuel Moisture	10	15	15
100-HR Fuel Moisture	18	20	20
Slope (%)	10	5	5
Aspect	NE	E	E
Spring Prescribed Fire			
Burn Date	26Apr1995	26Apr1995	26Apr1995
Time of Burn	2000	1630	1830
Air Temperature (°F)	68	73	70
Relative Humidity (%)	28	20	20
Wind Direction	SW	SW	SW
Wind Speed (mph)	1	5	3
Cloud Cover (%)	0	0	0
1-HR Fuel Moisture (%)	9	5	7
10-HR Fuel Moisture	10	10	10
100-HR Fuel Moisture	12	12	12
Slope (%)	7	5	10
Aspect	E	SW	E
Summer Prescribed Fire			
Burn Date	24Aug1995	24Aug1995	24Aug1995
Time of Burn	1630	1430	1230
Air Temperature (°F)	92	95	95
Relative Humidity (%)	56	44	46
Wind Direction	SW	SW	SW
Wind Speed (mph)	0	5	8
Cloud Cover (%)	0	0	0
1-HR Fuel Moisture (%)	11	9	9
10-HR Fuel Moisture	10	15	15
100-HR Fuel Moisture	14	14	14
Slope (%)	3	10	5
Aspect	NE	W	NW

Table 2—Mean loadings (tons/acre ± 1 standard error) of small, medium, large, and total fuels by treatment and year. Means followed by different uppercase letters are different within that row while means followed by different lowercase letters are different within that column. N = 45 for each cell

Fuels	Treatment	1994	1995	1997
Small[1]	Uncut	4.1 ± 0.3Ad	4.3 ± 0.3Af	4.3 ± 0.3Af
Small	Shelterwood	4.3 ± 0.2Ad	4.3 ± 0.3Af	4.3 ± 0.2Af
Small	Burned Shwd	5.1 ± 0.4Ad	0.8 ± 0.1Ci	3.2 ± 0.3Bg
Medium[2]	Uncut	1.5 ± 0.2Ae	1.5 ± 0.2Ah	1.4 ± 0.2Ah
Medium	Shelterwood	7.5 ± 0.4Ac	7.9 ± 0.3Ad	7.8 ± 0.5Ad
Medium	Burned Shwd	6.8 ± 0.4Ac	2.1 ± 0.2Cg	6.3 ± 0.4Be
Large[3]	Uncut	4.9 ± 1.5Ad	4.9 ± 1.4Af	4.9 ± 1.5Af
Large	Shelterwood	10.8 ± 1.4Ab	12.2 ± 1.4Ab	12.1 ± 1.5Ac
Large	Burned Shwd	11.1 ± 0.8Ab	6.1 ± 0.9Be	10.2 ± 0.7Ac
Total	Uncut	10.5 ± 0.5Ab	10.7 ± 0.6Abc	10.6 ± 0.6Ac
Total	Shelterwood	22.6 ± 2.1Aa	24.4 ± 2.0Aa	24.2 ± 2.3Aa
Total	Burned Shwd	23.0 ± 2.1Aa	9.0 ± 1.1Cc	19.7 ± 1.7Bb

[1]Small fuels are leaf litter and woody material less than 1.0 inch diameter.
[2]Medium fuels are woody material between 1.0 and 3.0 inch diameter.
[3]Large fuels are woody material greater than 3.0 inch diameter.

In fall 1997, 2 to 2.5 years after the fires, fewer differences in fuel loadings among the three treatments were detected (table 2). The shelterwood had a total fuel loading of 24.2 tons/acre. This amount differed from that of the burned shelterwood (19.7 tons/acre), but both were more than the total fuel loading found in the uncut control (10.6 tons/acre). The two shelterwood treatments had equivalent loadings of large fuels (12.1 and 10.2 tons/acre) and equivalent loadings of medium fuels (7.8 and 6.3 tons/acre). All of these amounts were greater than the corresponding loadings of the uncut control (4.9 tons/acre for large fuels and 1.4 tons/acre for medium fuels). However, the uncut control and the unburned shelterwood treatments had greater leaf litter loadings (4.3 tons/acre for each) than the shelterwood + fire treatment (3.2 tons/acre). Relative to 1995, fuel loadings had not changed in the uncut and the shelterwood treatments. However, in the burned shelterwood treatment, loadings for all fuel classes had substantially increased from 1995 to 1997. Small fuel loading rose 300 percent, from 0.8 to 3.2 tons/acre. Medium fuel loading climbed 200 percent, from 2.1 to 6.3 tons/acre. Large fuel loading increased 67 percent, from 6.1 to 10.2 tons/acre. These increases made the large and medium loadings in 1997 equivalent or nearly so to what they had been in 1994, but the leaf litter loading (3.2 tons/acre) was only 60 percent of what it had been prior to the prescribed fires.

DISCUSSION

In this study, the uncut control treatment represents upland oak stands at the beginning of the shelterwood-burn technique. Small fuel loading was approximately 4 tons/acre. This amount of small fuel is consistent with the findings of numerous researchers studying forest fuels in mature, fully-stocked upland oak stands in Georgia (Clinton and others 1998), Iowa (Kucera 1952), Missouri (Loomis 1975), Ohio (Graham and McCarthy 2006), South Carolina (Metz 1954), and Tennessee (Blow 1955). The combined loading of medium and large fuels, 6.4 tons/acre, is slightly less than the range of 7.5 to 9.5 tons/acre reported for upland oak stands of comparable ages and site conditions (Lang and Forman 1978, MacMillan 1981, McCarthy and Bailey 1994) indicating that medium and large fuels will likely continue accumulating in the uncut stands as time passes. Apparently, the uncut oak stands used in this study are quite similar to that of other undisturbed upland oak forests throughout the eastern United States when it comes to total fuel loadings and how those loadings are distributed among the fuel size classes.

In the shelterwood-burn technique, the purpose of the first removal cut is to promote the development of the existing oak reproduction and increase the fuel loadings so the subsequent prescribed fire has the intensity to control the competing mesophytic hardwoods (Brose

and others 1999a, b). In this study, the first removal cuts created shelterwood stands by removing intermediate trees from the midstory strata and weak co-dominant stems from the main canopy. On average, the residual basal area and canopy closure of each shelterwood stand were 50 feet^2/acre and 50 percent, respectively. Because these first removal cuts occurred 2 to 4 years before the start of the study, it is unknown what the fuel loadings were immediately after the harvesting. However, after 2 to 4 years, these oak shelterwoods had total fuel loadings that were more than double those of the undisturbed oak stands (~23 tons/acre versus ~10 tons/acre). This drastic change was concentrated in the large and medium fuels due to the logging slash being left on site. Small fuel loading was probably initially diminished by the harvesting operation (Graham and McCarthy 2006, Kolaks and others 2004), but subsequent leaf fall replaced this loss resulting in no net change in small fuel loading 2 to 4 years later. In the shelterwood-burn technique, the recommended time interval between the first harvest and the prescribed fire is 4 to 7 years (Brose and others 1999b). Forest managers wanting to implement this regeneration method can expect candidate oak shelterwood stands to have 20 to 25 tons/acre of leaf litter and woody fuels when they are 4 to 7 years old.

The desired prescribed fire in the shelterwood-burn technique is a medium- to high-intensity fire (2- to 4-foot flame lengths) in mid spring when the leaves of the mesophytic hardwood seedlings are at least 50 percent expanded (Brose and others 1999a, b). With total fuel loadings ranging from 20 to 25 tons/acre and a ubiquitous leaf layer, achieving the desired fire behavior is rather easy provided the weather conditions are conducive for prescribed burning. Based on the results of this study, the fire will incinerate between half and two-thirds of the total fuel loading. Nearly all of the small fuels, especially the leaf litter, and more than 50 percent of the medium fuels will be consumed. Decreases in large fuel loading will be centered where there are concentrations of logging slash that increase fire intensity and residence time.

After the prescribed fire, the oak and mesophytic hardwood reproduction must be evaluated to determine if the burn objective was met. Generally, this is done in the second growing season postfire (Brose and others 1999a, b). If the mesophytic hardwood reproduction has not been adequately controlled, a second burn may be necessary. The timing of this second burn is dependent on the re-accumulation of the small fuels, especially the leaf litter, so that the fire can easily move through the stand. In mature, fully-stocked oak stands, leaf litter production is about 2 tons/acre/year with a total leaf litter loading of 4 to 6 tons/acre once equilibrium between foliage production and litter decomposition is reached (Loomis 1975). Consequently, repeat understory burns in mature stands can occur on a 2- to 4-year interval to allow for adequate re-accumulation of the leaf litter to carry the fire. In oak shelterwoods, leaf litter production will be considerably less due to the removal of half or more of the basal area via the first removal cut. In this study, small fuel loading had risen from 0.8 tons/acre shortly after the fires to 3.2 tons/acre by the end of the third growing season. The 3.2 tons/acre loading was only 60 percent of the pre-burn small fuel loading (5.1 tons/acre). Obviously, more time is needed for the leaf litter to re-accumulate to pre-burn levels, probably an additional 2 to 4 years. In oak shelterwoods, a second prescribed fire should be feasible 4 to 7 years after the first one, depending on site quality.

The re-accumulation of the large fuel loadings to pre-burn levels was somewhat surprisingly, but not totally unexpected. The prescribed fires had injured or killed a few of the overstory trees, especially ones with slash piles near their boles. By the third year after the fire, some of the branches from these injured or killed trees had fallen to the ground and were inventoried as large fuels.

In conclusion, the shelterwood-burn technique is an oak regeneration method that takes 8 to 14 years. Most of that time occurs in two separate intervals of 4- to 7-years each. Both intervals are absolutely necessary and are based, to some degree, on the dynamics of the forest fuels. The first interval comes between the first removal cut and the prescribed fire. It allows the leaf litter to re-accumulate following the logging disturbance so the prescribed fire can easily burn through the stand. It also permits the woody fuels to dry so that the desired fire intensity can be readily attained. The second 4- to 7-year wait occurs after the prescribed fire. During this interval, leaf litter re-accumulates so if a second fire is needed, it can be easily conducted.

LITERATURE CITED

Anderson, H.E. 1978. Graphic aids for field calculations of dead downed forest fuels. Gen. Tech. Rep. INT-45. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 15 p.

Blow, F.E. 1955. Quantity and hydrologic characteristics of litter under upland oak forests in eastern Tennessee. Journal of Forestry. 53: 190-195.

Brose, P.H.; Van Lear, D.H. 1998. Responses of hardwood advanced regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. Canadian Journal of Forest Research. 28: 331-339.

Brose, P.H.; Van Lear, D.H.; Cooper, R. 1999a. Using shelterwood harvests and prescribed fire to regenerate oak stands on productive upland sites. Forest Ecology and Management. 113: 125-141.

Brose, P.H.; Van Lear, D.H.; Keyser, P.D. 1999b. A shelterwood-burn technique for regenerating oak stands on productive upland sites in the Piedmont region. Southern Journal of Applied Forestry. 23(3): 158-163.

Brown, J.K. 1974. Handbook for inventorying downed woody material. Gen. Tech. Rep. INT-16. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 24 p.

Clinton, B.D.; Vose, J.M. 2007. Fuels consumption and nitrogen loss following prescribed fire: a comparison of prescription types in the southern Appalachians. In: Clatterbuck, W., ed. Proceedings of the 15th central hardwood forest conference. Gen. Tech. Rep. SRS-101. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 231-240.

Clinton, B.D.; Vose, J.M.; Swank, W.T. [and others]. 1998. Fuel consumption and fire characteristics during understory burning in a mixed white pine-hardwood stand in the southern Appalachians. Res. Pap. SRS-12. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 11 p.

Crosby, J.S.; Loomis, R.M. 1974. Some forest floor fuelbed characteristics of black oak stands in southeast Missouri. Res. Note NC-162. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 4 p.

Dickinson, M.B., ed. 2006. Proceedings, Fire in eastern oak forests: delivering science to land managers. Gen. Tech. Rep. NRS-P-1. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 303 p.

Fosberg, M.A. 1970. Drying rates of wood below fiber saturation. Forest Science. 16: 57-63.

Graham, J.B.; McCarthy, B.C. 2006. Forest floor dynamics in mixed-oak forests of southeastern Ohio. International Journal of Wildland Fire. 15: 479-488.

Hutchinson, T.F. (Ed.), 2009. In: Proceedings, 3rd fire in the eastern oak forests conference. Gen. Tech. Rep. NRS-P-46. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 154 p.

Knoepp, J.D.; Elliott, K.J.; Clinton, B.D.; Vose, J.M. 2009. Effects of prescribed fire in mixed oak forests of the southern Appalachians: forest floor, soil, and soil solution nitrogen responses. Journal of the Torrey Botanical Society. 136 (3): 380-391.

Kolaks, J.J.; Cutter, B.E.; Loewenstein, E.F. [and others]. 2004. The effect of thinning and prescribed fire on fuel loading in the central hardwood region of Missouri. In: Yaussy, D.A.; Hix, D.M.; Long, R.P.; Goebel, P.C., eds. Proceedings of the 14th central hardwood forest conference. Gen. Tech. Rep. NE-316. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station: 168-178.

Kucera, C.L. 1952. An ecological study of a hardwood forest area in central Iowa. Ecological Monographs. 22: 282-299.

Lang, G.E.; Forman, R.T. 1978. Detrital dynamics in a mature oak forest: Hutcheson Memorial Forest, New Jersey. Ecology. 59: 580-595.

Loomis, R.M. 1975. Annual changes in forest floor weights under a southeast Missouri oak stand. Res. Note NC-184. St Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 3 p.

Loomis, R.M.; Crosby, J.S. 1968. Fuel evaluation for a Missouri pine plantation released by aerial spraying. Res. Note NC-184. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 4 p.

Loomis, R.M.; Crosby, J.S. 1970. Fuel hazard from breakup of dead hardwoods in Missouri. Journal of Forestry. 68(8): 490-493.

MacMillan, P.C. 1981. Log decomposition in Donaldson's Woods, Spring Mills State Park, Indiana. American Midland Naturalist .106: 335-344.

McCarthy, B.C.; Bailey, R.R. 1994. Distribution and abundance of large woody debris in a managed forest landscape of the central Appalachians. Canadian Journal of Forest Research. 24: 1317-1329.

Metz, L.J. 1954. Forest floor in the Piedmont region of South Carolina. Soil Science Society Journal. 18: 335-338.

Phillips, R.J.; Waldrop, T.A. 2013. Fuel loading following fuel-reduction treatments and impacts from natural disturbances. In: Guldin, J.M., ed. Proceedings of the 15th biennial southern silvicultural research conference. Gen. Tech. Rep. e-SRS-175. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 533-540.

Reber, E.J. 1988. Soil survey of Powhatan County, Virginia. Richmond, VA: U.S. Department of Agriculture, Soil Conservation Service. 111 p.

Waldrop, T.A.; Glass, D.W.; Rideout, S. [and others]. 2004. An evaluation of fuel-reduction treatments across a landscape gradient in Piedmont forests: preliminary results of the National Fire and Fire Surrogate Study. In: Connor, K.F., ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 54-59.

Waldrop, T.A.; Brudnak, L.; Rideout-Hanzak, S. 2007. Fuels on disturbed and undisturbed sites in the southern Appalachian Mountains, USA. Canadian Journal of Forest Research. 37: 1134-1141.

Yaussy, D. A., comp. 2000. Proceedings, Workshop on fire, people, and the central hardwoods landscape. 2000 March 12-14; Richmond, KY. Gen. Tech. Rep. NE-274. Newtown Square, PA: U.S. Department of Agriculture, Forest Service Northeastern Research Station. 129 p.

MECHANICAL MASTICATION AS A FUELS TREATMENT IN SOUTHEASTERN FORESTS

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Abstract—Mastication is an increasingly common fuels treatment that redistributes ''ladder'' fuels to the forest floor to reduce vertical fuel continuity, crown fire potential, and fireline intensity. Despite its widespread adoption, it remains unclear how mastication impacts fuels, fire behavior, or plant communities across Southeastern forest ecosystems. We evaluated these effects by reviewing studies conducted across Southeastern pine forests. Mastication is typically applied to reduce fire hazard prior to reintroducing fire to long-unburned sites and to promote desired herbaceous groundcover where woody species have become dominant. Pretreatment fuel conditions varied across the different studies, ultimately leading to variation in post-treatment fuels. Only a few studies have examined fire behavior in masticated fuels and its potential effects. Field-scale burns conducted under mild conditions have resulted in variable fuel consumption and minimal overstory tree mortality. Substantial surface fuel loads in sites with prior stand damage, however, suggests that fire hazard may not be alleviated if sites burned under wildfire conditions. Modeled fire behavior indicates the effectiveness of treatments at reducing potential fire hazard, but verifying predictions under wildfire conditions has not been done. Initial herbaceous response has been positive in some sites, but rapid recovery of woody species indicates the importance of frequent burning to sufficiently restore plant communities and vegetation structure indicative of fire dependent pine forests in the Southeastern US.

INTRODUCTION

Mechanical mastication ("mowing", "chipping", "mulching") has become a widely used fuels treatment option in forests and shrublands across the US (Kreye and others 2014a). Mastication primarily targets under- and mid-story vegetation either alone or in conjunction with other treatments (e.g., overstory thinning or prescribed fire). Treatments are aimed at reducing fire hazard by altering fuel structure through disconnecting surface and canopy fuels. Mastication is also used as a restoration tool where undesired species composition or structure has developed over long periods of fire exclusion. The widespread use of mastication as a management tool has outpaced research efforts aimed at understanding their treatment efficacy or unexpected consequences.

Much of the research evaluating the initial effects of mastication on over- and understory vegetation, post-treatment fuel characteristics, and subsequent fire behavior has been conducted in the western US (Collins and others 2007, Kane and others 2010, Kreye and others 2014a, Moghaddas and others 2008, Ostaja and others 2014, Perchlemlides and others 2008, Potts and Stephens 2009, Ross and others 2012, Young and others 2013). Studies across a variety of ecosystems have highlighted the variability of these effects.

Reductions in stand density are common in forests where under- and midstory trees were present, but as a stand-alone treatment, the impacts of mastication on basal area are less prominent, as small-diameter trees are usually targeted. Surface fuels resulting from mastication vary in loading, but are often dominated by small-diameter (<7.62 cm) fractured woody debris compacted into shallow (<10 cm) fuel depths. Surface fuel loading can be substantial, however, particularly where dense pre-treatment vegetation existed (Kreye and others 2014a). Early vegetation response varies greatly across the ecosystems in which mastication is used. Surface debris may inhibit woody and herbaceous recruitment unless bare ground is exposed from post-treatment burning.

Given the compactness of post-mastication surface fuels and their fine woody composition, early lab-scale fire behavior studies revealed their propensity for long-duration combustion and potential surface and belowground heating (Busse and others 2005, Kreye and others 2011). Field-scale studies have highlighted substantial overstory mortality following prescribed burns in masticated stands (Bradley and others 2006), even where flame lengths were minimal (Knapp and others 2011). Given the trade-off between potential crown ignition and surface fire behavior it is important

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to fully understand the fuel dynamics associated with these treatments and their links to subsequent fire effects.

While studies examining the role of mastication in western forests and shrublands have gained some momentum over the last ten years, much less research has been focused on the recent widespread mastication in the Southeast. Here we review recent research in Southeastern forest ecosystems examining the use of mastication employed as a restoration or fire hazard mitigation tool. We compile findings evaluating impacts of mastication on fuel structure and characteristics, subsequent fire behavior during lab- or field-scale burns, and vegetation response following stand-alone mastication treatments or prescribed burns conducted following these treatments. We concurrently compare and contrast treatments in the Southeast with those applied in the western US in order to put these treatments into a broader context of their impacts and effectiveness. While the breadth of mastication research in the Southeast remains modest, we prioritize research needs that will inform our understanding of these treatments in the region to better evaluate their efficacy as a land management tool.

MASTICATION TREATMENTS IN SOUTHEASTERN FORESTS

Mastication is the grinding, shredding, chipping, mulching, or mowing of understory shrubs or small trees (Kreye and others 2014a). Front- or boom-mounted rotary heads are attached to ground-based equipment, wheeled or tracked, where the operator has hydraulic control of the mastication head. Rotary mastication heads may be cylindrical, with fixed or flailing cutters (fig. 1a), or consist of rotating blades, similar to a lawnmower. In either case, masticators can target specific vegetation unlike roller-chopping or bush hogging that treat sites more bluntly.

Mastication is used in Southeastern pine forests to mitigate fire hazard in the wildland-urban-interface where burning is difficult, to restore stand structure in fire excluded sites, and as a pre-treatment to reintroducing fire into long-unburned sites. Restoration objectives often include promoting herbaceous plants where woody vegetation has become dominant with fire exclusion. Historically, frequent fire regimes maintained open canopy structure and high herbaceous diversity in the understory (e.g., longleaf pine (*Pinus palustris*) sandhills and flatwoods). Other targeted ecosystems, such as sand pine (*P. clausa*) scrub, oak-saw palmetto scrub, and coastal scrub are typified by a less frequent stand replacing fire regime; mastication is used in these ecosystems to alter vegetation structure where prescribed burning is a challenge.

Few studies have evaluated mechanical mastication treatments in Southeastern forests. Glitzenstein and others (2006), Kreye and others (2013a, 2104b, 2015), and Ottmar and Prichard (2012) characterized post-treatment fuels and assessed their burning behavior, while others have focused on treatment effects to various plant community attributes (Brockway and others 2009, Kreye 2012, Schmalzer and others 2003, Weekley and others 2008). Treatments have occurred in sites that have been unburned for as little as five years (Kreye and others 2014a) or up to several decades (Brockway and others 2009). In some sites, mastication took place following other disturbances that have resulted in significant stand damage and in high quantities of snags and large woody debris (Glitzenstein and others 2006, Stottlemyer and others 2015).

FUELS AND FIRE BEHAVIOR

Fuels have been characterized in masticated treatments in coastal plain forests in South Carolina (Glitzenstein and others 2006, Ottmar and Prichard 2012, Stottlemyer and others 2015) and northern Florida (Kreye and others 2014b) (table 1). Glitzenstein and others (2006) characterized fuels following mastication in loblolly pine (*Pinus taeda*) flatwoods on the Francis Marion National Forest in stands with dense mid- and understory regeneration and large downed woody material following hurricane damage. Total surface fuels were substantial (195 Mg ha^{-1}) in treated sites, dominated by large fuels (>7.62 cm) that were likely post-hurricane debris not fully masticated. Finer wood (<7.62 cm) comprised a smaller proportion of surface fuels (32 percent) but were much greater than fine fuel loads in untreated sites. Glitzenstein and others (2006) modeled fire behavior in these sites and compared their predictions with fire behavior during prescribed burns. Observed fire behavior was mild, with flame lengths <50 cm and rate of spread varying between 0.5 to 3.0 m min^{-1}, similar to burns conducted at the same time in untreated sites. Mastication resulted in lower scorch heights than unmasticated sites.

Stottlemyer and others (2015) also evaluated surface fuels following mastication in sites with previous stand damage. They reported substantial surface fuel loads in masticated loblolly pine plantations (18-33 years old) in the South Carolina Piedmont that had substantial overstory mortality (79 dead pines ha^{-1}) from bark beetle attack that occurred 4 to 6 years prior to fuels treatment. Total surface fuel loading increased from 27 to 192 Mg ha^{-1} following treatment, with post-treatment biomass ranging from 126 to 258 Mg ha^{-1}. Although surface fuel loads were substantial, fuel depth averaged only 15 cm across stands, ranging from 12 to 17 cm, highlighting the significant fuelbed bulk density often observed in other sites (Kreye and others 2014b). Although woody fuel biomass was not separated by size



Figure 1—A front-mounted masticator (a), a post-masticated pine flatwoods site in northern Florida (b), and a prescribed burn conducted in a pine flatwoods site in north Florida six months following treatment (c).

Table 1—Studies characterizing surface fuels and fire behavior in mastication fuels treatments in Southeastern US forests

Forest Type (State)	1h	10h	100h	1000h	Litter	Shrubs	Grass	Total	Flame Length	Rate of Spread	Fire Intensity[g]
	Mg ha^{-1}								m	m min^{-1}	
Loblolly Pine Flatwoods (SC)[a] (hurricane damage)	2.82	24.10	35.15	127.30	5.31	0.95	0.18	195.81	0.35	1.36	
Variable, Coastal Plain (SC)[b]											
Unthinned											
masticated only	4.86[‡]				9.57	0	0	14.43	1.70	0.90	396 *kW m*$^{-2}$
masticated only	4.37[‡]				7.25	0	0	11.63	1.50	0.70	362 *kW m*$^{-2}$
masticated / burned	5.15[‡]				11.31	3.24	0.02	19.72	2.10	0.90	402 *kW m*$^{-2}$
Thinned											
masticated only	6.48[‡]				7.83	0	0	14.31	0.80	1.40	384 *kW m*$^{-2}$
masticated only	4.98[‡]				6.09	0	0	11.07	0.70	1.40	375 *kW m*$^{-2}$
masticated/herbicide/burned	7.04[‡]				8.70	0	0.02	15.73	0.80	1.90	366 *kW m*$^{-2}$
Longleaf Pine Flatwoods (FL)[c]											
low load				na		na	na	10.0[‡‡]	0.49	0.75	183 *kW m*$^{-1}$
moderate load				na		na	na	20.0[‡‡]	0.91	0.98	487 *kW m*$^{-1}$
high load				na		na	na	30.0[‡‡]	1.40	1.00	773 *kW m*$^{-1}$
Longleaf Pine Flatwoods (FL)[d]											
10+ yr rough	3.1	2.1	0.4	na	12.6	0	0	18.2			
10+ yr rough	3.2	5.3	2.8	1.9	9.5	0	0	22.7			
5 yr rough	2.3	2.6	0.4	0.6	11.8	0	0	17.7			
27 yr old plantation	2.7	5.4	1.3	5.1	13.6	0	0	28.1			
Longleaf Pine Flatwoods (FL)[e]	1.1	2.1	1.1		13.4	cover 33% height 58cm			1.1	3.4	
Loblolly Pine Plantation (SC)[f] (bark beetle damage)								192.4[‡‡‡]			

[a]Glitzenstein et al. 2006
[b]Ottmar & Prichard 2012 ([‡]1,10,100 h fuels combined; fire behavior modeled in FCCS)
[c]Kreye et al. 2013 ([‡‡]lab scale burns conducted with masticated surface fuels collected immediately following treatment)
[d]Kreye et al. 2014
[e]Kreye et al. 2015
[f]Stottlemyer et al. 2015([‡‡‡]Included all surface fuels)
[g]Reported as either reaction intensity (*kW m*$^{-2}$) or fireline intensity (*kW m*$^{-1}$)

class, they indicate that all standing live and dead fuel, except for the few remaining live pines, and all surface fuels were masticated. Operators masticated surface debris so that litter, duff, and mineral soil were mixed during treatment. Glitzenstein and others (2006) and Stottlemyer and others (2015) both highlight substantial increases in surface fuels that occur when mastication is used to treat sites with significant large dead woody material that can result from other natural disturbances.

Kreye and others (2014b) quantified surface fuels following mastication in longleaf/slash pine (*Pinus elliottii*) flatwoods in northern Florida. In contrast to previous findings (Glitzenstein and others 2006, Stottlemyer and others 2015), surface fuels in masticated sites were dominated by shredded foliar litter (from saw palmetto (*Serenoa repens*) and gallberry (*Ilex glabra*) shrubs) with a smaller proportion of fine woody debris (primarily < 2.54 cm) and few large fuels (fig. 1b). Even though a dense shrub stratum was treated, post-treatment surface fuels were lighter than in many other masticated sites (Kreye and others 2014a), averaging only 17 to 23 Mg ha^{-1} across different stand types. Kreye and others (2013a) examined fire behavior in surface fuels collected from these sites during controlled lab-scale (4 m diameter) burns. They revealed the strong role of fuel load and moisture content on fire intensity and showed the potential for long-duration surface heating, but residence times were shorter than those observed in other laboratory studies that burned woody-dominated fuels (Busse and others 2005, Kreye and others 2011). Soil temperatures failed to reach lethal levels (60 °C) as shallow as 2 cm, in contrast to western studies (Busse and others 2005). Kreye and others (2014b) subsequently examined fuel response for up to two years following treatments in their sites and examined fire behavior at the stand-scale during prescribed burning. Rapid shrub recovery was evident following treatments (Kreye and others 2014b) and shrub cover, rather than surface fuel loading, was strongly correlated with observed flame lengths during dormant-season prescribed burns (fig. 1c; Kreye and others 2015). Mastication did result in lower flame lengths and subsequently lower bole char and crown scorch compared to burning of untreated controls. Overstory mortality was minimal following burns in both treated and untreated sites. Most of the masticated debris was consumed in treated sites (>80 percent), but almost no duff was consumed in any sites.

Ottmar and Prichard (2012) inventoried fuels across several stands on the Francis Marion National Forest in South Carolina, six of which had been masticated, and used their data to construct fuelbeds and predict fire behavior within the Fuel Characteristic Classification System (FCCS) (Ottmar and others 2007). Surface fuels six to eight months following mastication were 11.6 to 14.4 Mg ha^{-1} in unthinned stands with litter comprising >60 percent of mass. One year following prescribed burning in an unthinned masticated stand, surface fuels were 16.5 Mg ha^{-1} (>60 percent litter), but shrubs and grasses were also present, unlike stands that had not been burned. In thinned stands, masticated sites were comprised of 11.1 to 14.3 Mg ha^{-1} (>50 percent litter) six years following treatment. In one thinned and herbicided stand followed by mastication and subsequent burning, surface fuels were 15.7 Mg ha^{-1} (>50 percent litter) eight months after treatment. Although treatments occurred over different stand types, surface fuel characteristics were generally similar, with low fuel loading compared to other studied sites (Kreye et al. 2014a) and composed of over half to two-thirds litter by mass, similar to Kreye and others (2014b). Ottmar and Prichard (2012) subsequently used FCCS to predicted fire behavior across their fuelbeds. Their findings predict reduced fire behavior (reaction intensity, rate of spread, flame length) in treated sites, but suggest that litter and shrub accumulation following treatments may reduce treatment efficacy.

Studies characterizing surface fuels following mastication treatments in the Southeast are limited, yet fuels information is critical for evaluating the efficacy of treatments in altering fire behavior and effects. Kreye and others (2014b) and Ottmar and Prichard (2012) both indicated low to moderate surface fuel loads compared to many masticated sites studied in the west (Kreye and others 2014a) and to other Southeastern sites where substantial large woody debris resulted from previous wind (Glitzenstein and others 2006) or bark beetle (Stottlemyer and others 2015) damage. Mitigating potential wildfire hazard using mastication may be effective where masticated fuels are light or where a mulching effect retains moisture (Kreye and others 2012), thereby limiting forest floor consumption and potential deleterious effects. Moisture regimes of the Southeast may be more conducive to treatment efficacy compared to western climates where prolonged summer drying occurs (Schroeder and Buck 1970) and where fire behavior in masticated sites has resulted in overstory mortality (Bradley and others 2006, Knapp and others 2011). Using prescribed fire to consume masticated debris, however, may be challenging in the Southeast given the complexities of fuel and moisture dynamics in these treatments. Balancing surface fuel consumption with the potential for duff ignition (Kreye and others 2013b) may be critical. These few studies highlight the variability of surface fuels created from mastication treatments.

PLANT COMMUNITIES

Two studies have examined vegetation response in masticated sites in scrub ecosystems in Florida (table 2). Schmalzer and others (2003) showed that saw palmetto cover was reduced in the long-term in masticated stands in both oak (*Quercus chapmanii,*

Table 2—Vegetation response to mastication fuels treatments in Southeastern US Forests. Increase (+), decrease (-), change (Δ), no change (=), or not studied (blank). Diversity included species richness (rich), species evenness (even), or abundance of rare species (rare). Treatment differences are indicated where a treatment increased (++) or decreased (--) more than the other treatments

	Masticated			Burned			Masticated and Burned		
Forest Type (State)	Herbs	Composition	Diversity	Herbs	Composition	Diversity	Herbs	Composition	Diversity
Oak-Saw Palmetto Scrub (FL)[a]					=			Δǂ	
Coastal Scrub (FL)[a]					=			Δǂ	
Lake Apthorpe Scrub (FL)[b]		=	= rare		Δ	++ rare		Δ	+ rare
Lake Placid Scrub (FL)[b]		Δ	= rare		=	+ rare		Δ	= rare
Longleaf Pine Sandhill (GA)[c]	+	Δ	+ rich,- even					Δ	+ rich,- even
Longleaf Pine Flatwoods (FL)[d]	=	Δǂ	+ rich[†]	=	=	=	=	Δǂ	+ rich[†]

[a]Schmalzer 2003
[b]Weekley et al. 2008
[c]Brockway et al. 2009
[d]Kreye 2012
[ǂ]change reflected as a decrease in saw palmetto cover
[†]Marginal differences were reported.

Q. myrtifolia, Q. geminata)-saw palmetto scrub and coastal scrub (*Q. virginiana*) on the Merritt Island/ Cape Canaveral Barrier Island Complex in Florida. Height and cover of scrub were substantially reduced by mastication and were still less than pretreatment levels and prescribed fire only six years following treatment. Recovery of oaks, however, was rapid following either prescribed burning or mastication. Weekley and others (2008) evaluated scrub response to mastication at two sites at the Lake Wales Ridge Wildlife and Environmental Area in Florida. Woody cover was significantly reduced, a primary objective, but not as successfully as in sites that were burned, either with or without prior mastication. Woody plant heights remained low up to five years after all treatments. Mastication changed plant composition, as compared to controls, but they became more similar over time. Composition in sites that were burned, with or without mastication, remained different from that of controls and masticated sites. Abundance of rare herbaceous species, however, was only increased in sites that were burned without prior mastication.

Kreye (2012) examined the effects of mastication, burning, and their combination in mature longleaf pine/ palmetto-gallberry flatwoods on the Osceola National Forest in north Florida. Mastication reduced the dominant shrub component with little effect to trees. Similar to Kreye and others (2014b), recovery of palmetto and woody shrubs (primarily gallberry, *Vaccinium stamineum,* and *V. myrsinites*) was rapid. Saw palmetto density and cover, however, was reduced in masticated sites, including those subsequently burned, but recovered within a year in sites that were only burned. Woody shrub cover rebounded across all treatments within a year, but their heights were still reduced in all masticated sites. Similar to suggestions by Schmalzer and others (2003), the reduction in palmetto likely resulted from damage to meristems by machinery, while woody shrubs resprouted. Marginal evidence of increased species richness may have reflected herbaceous response to reductions of saw palmetto cover or increased bare ground, especially in masticated sites that were burned. Herbaceous cover and plant richness, however, were still quite low. Where treatment objectives include increasing herbaceous plants and understory plant diversity, single treatments alone are unlikely to be successful. Follow-up repeated mastication and/or fires may promote herbaceous plant cover and diversity, but these combinations have not been evaluated in any ecosystem.

Brockway and others (2009) examined vegetation response to mastication treatments in longleaf pine sandhills at Fort Benning in the western Georgia Fall-line Sandhills. They examined plant community response to mastication, but also the effects of burning in masticated sites in different seasons (winter, spring, summer). Mastication treatments targeted all hardwood trees, primarily *Liquidambar styraciflua, Quercus nigra,* and *Q. hemisphaerica*, and small pines (≤20 cm DBH), primarily loblolly pine, both of which had developed in the mid- and overstory following fire exclusion. Mastication reduced tree density by 79 percent, but basal area was only reduced by 26 percent, highlighting the effect to small-diameter trees. Tree seedling cover was also lower 13 months following treatment, but shrubs, vines, forbs, and grasses all increased. All vascular plant cover increased within 2 years post-treatment except for *Vaccinium corymbosum* and *Q. hemisphaerica*. Burning in masticated sites increased plant cover in general, but season-of-burn had differential effects on plant cover: shrubs and vines increased the most following winter and spring burning; grasses following winter burns; and forbs following summer burns. Herbaceous frequency and cover was generally low in sites prior to treatments, indicative of the dominance of woody establishment. Following all treatments, understory species richness increased significantly. Decreases in evenness occurred through time following treatments as a few species began to expand to a greater extent than others. Mechanical treatments altered stand structure by reducing ladder fuels and shifted overstory composition to more desired conditions (larger fire-tolerant pines). As understory woody species respond to treatments (e.g., via resprouting), repeated burning can be utilized to maintain or promote understory herbaceous diversity.

CONCLUSIONS

As both a fire-hazard mitigation treatment and a restoration tool, mastication is used as an initial treatment in long-unburned Southeastern pine forests where fire-sensitive species have invaded and herbaceous understories have declined. As a stand-alone treatment, mastication is an impractical long-term solution for fire hazard reduction and maintenance of understory plant diversity. Following mastication, prescribed fire is easier to implement. The divergence of fuels and potential fire behavior resulting from mastication may complicate long-term fuel and vegetation dynamics. Future research efforts should focus on mastication in other ecosystems where it is being employed, understanding the effects of seasonality of treatments or their repeated use, and evaluating fire behavior and effects under growing season and wildfire conditions.

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LITERATURE CITED

Busse, M.D.; Bussea, M.D.; Fiddler, G.O. [and others]. 2005. Lethal soil temperatures during burning of masticated forest residues. International Journal of Wildland Fire. 14: 267-276.

Bradley, T.; Gibson, J.; Bunn, W. 2006. Fire severity and intensity during spring burning in natural and masticated mixed shrub woodlands. In: Butler, B.W., Cook. W., eds. Fuels management – how to measure success: conference proceedings. RMRS–P–41. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 419-428.

Brockway, D.G.; Outcalt, K.W.; Estes, B.L.; Rummer, R.B. 2009. Vegetation response to midstorey mulching and prescribed burning for wildfire hazard reduction and longleaf pine (*Pinus palustris* Mill.) ecosystem restoration. Forestry. 82: 299-314.

Collins, B.M.; Collins, O.M.; Moghaddas, J.J.; Stephens, S.L. 2007. Initial changes in forest structure and understory plant communities following fuel reduction activities in a Sierra Nevada mixed conifer forest. Forest Ecology and Management. 239: 102-111.

Glitzenstein, J.S.; Streng, D.R.; Achtemeier, G.L. [and others]. 2006. Fuels and fire behavior in chipped and unchipped plots: implications for land management near the wildland/urban interface. Forest Ecology and Management. 236: 18-29.

Kane, J.M.; Varner, J.M.; Knapp, E.E.; Powers, R.F. 2010. Understory vegetation response to mechanical mastication and other fuels treatments in a ponderosa pine forests. Applied Vegetation Science. 13: 207-220.

Knapp, E.E.; Varner, J.M.; Busse, M.D. [and others]. 2011. Behaviour and effects of prescribed fire in masticated fuel beds. International Journal of Wildland Fire. 20: 932-945.

Kreye, J.K.; Varner, J.M.; Knapp, E.E. 2011. Effects of particle fracturing and moisture content on fire behavior in masticated fuel beds burned in a laboratory. International Journal of Wildland Fire. 20: 308-317.

Kreye, J.K. 2012. Efficacy and ecological effects of mechanical fuel treatments in pine flatwoods ecosystems of Florida. Gainesville, FL: University of Florida. 185 p. Ph.D. dissertation.

Kreye, J.K.; Varner, J.M.; Knapp, E.E. 2012. Moisture desorption in mechanically masticated fuels: effects of particle fracturing and fuel bed compaction. International Journal of Wildland Fire. 21: 894-904.

Kreye, J.K.; Kobziar, L.N.; Zipperer, W.C. 2013a. Effects of fuel load and moisture content on fire behavior and heating in masticated litter-dominated fuels. International Journal of Wildland Fire. 22: 440-445.

Kreye, J.K.; Varner, J.M.; Dugaw, C.J. [and others]. 2013b. Pine cones facilitate ignition of forest floor duff. Canadian Journal of Forest Research. 43: 512-516.

Kreye, J.K.; Brewer, N.W.; Morgan, P. [and others] 2014a. Fire behavior in masticated fuels: A review. Forest Ecology and Management. 314: 193-207.

Kreye, J.K.; Kobziar, L.N.; Camp, J.M. 2014b. Immediate and short-term response of understory fuels following mechanical mastication in a pine flatwoods site of Florida, USA. Forest Ecology and Management. 313: 340-354.

Kreye, J.K.; Kobziar, L.N. 2015. The effect of mastication on surface fire behavior, fuels consumption and tree mortality in pine flatwoods of Florida, USA. International Journal of Wildland Fire. 24: 573-579.

Moghaddas, J.J. 2008. Initial response of conifer and California black oak seedlings following fuel reduction activities in a Sierra Nevada mixed conifer forest. Forest Ecology and Management. 255: 3141-3150.

Ostoja, S.M.; Brooks, M.L.; Dudley,T.; Lee, S.R. 2014. Short-term vegetation response following mechanical control of Saltcedar (*Tamarix* spp.) on the Virgin River, Nevada, USA. Invasive Plant Science and Management. 7: 310-319.

Ottmar, R.D.; Prichard, S.J. 2012. Fuel treatment effectiveness in forests of the upper Atlantic Coastal Plain – an evaluation at two spatial scales. Forest Ecology and Management. 273: 17-28.

Ottmar, R.D.; Sandberg, D.V.; Riccard, C.L.; Prichard, S.J. 2007. An overview of the Fuel Characteristic Classification System-quantifying, classifying, and creating fuel beds for resource planning. Canadian Journal of Forest Research. 37: 2383-2393.

Perchemlides, K.; Muir, P.; Hosten, P. 2008. Responses of chaparral and oak woodland plant communities to fuel-reduction thinning in southwestern Oregon. Rangeland Ecology and Management. 61: 98-109.

Potts, J.; Stephens, S. 2009. Invasive and native plant responses to shrubland fuel reduction: comparing prescribed fire, mastication, and treatment season. Biological Conservation. 142: 1657-1664.

Ross, M.R.; Castle, S.C.; Barger, N.N. 2012. Effects of fuels reductions on plant communities and soils in a Piñon-juniper woodland. Journal of Arid Environments. 79: 84-92.

Schmalzer, P.A. 2003. Growth and recovery of oak-saw palmetto scrub through ten years after fire. Natural Areas Journal. 23: 5-13.

Schroeder, M. J.; Buck, C.C. 1970. Fire weather: a guide for application of meteorological information to forest fire control operations. Agric. Handb. 360. Washington, DC: U.S. Department of Agriculture. 229 p.

Stottlemyer, A.D.; Waldrop, T.A.; Wang, G.G. 2015. Prescribed burning and mastication effects on surface fuels in southern pine beetle-killed loblolly pine plantations. Ecological Engineering. 81: 514-524.

Weekley, C.W.; Menges, E.S.; Rickey, M.A. [and others]. 2008. Effects of mechanical treatments and fire on Florida scrub vegetation. Final Report to U.S. Fish and Wildlife Service. Vero Beach, FL: Vero Beach Office. 37 p.

Young, K. R.; Roundy, B. A.; Eggett, D. L. 2013. Plant establishment in masticated Utah juniper woodlands. Rangeland Ecology and Management. 66: 597-607.

TREE MORTALITY FOLLOWING A DROUGHT-YEAR LIGHTNING IGNITION IN THE OUACHITA MOUNTAINS, ARKANSAS: 2 YEARS POSTBURN

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Abstract—Increasingly, fire managers are using natural ignitions in conjunction with prescribed burns to restore and maintain fire-adapted ecosystems. Increased fuel loading from fire suppression and increasing drought indices associated with climate change, however, may cause natural ignitions to burn with greater intensity and severity. Managers must weigh risk factors versus benefits before allowing a lightning ignition to burn under these conditions. During the severe drought of 2011, the Ouachita National Forest had more lightning ignitions than any year in recorded history. While most were fully suppressed, one lightning ignition occurred in a particularly remote and rugged location that made suppression difficult and unsafe. Managers decided to use "less than full suppression" techniques and allowed this fire to burn within a designated containment area (~700 ha). Given the drought conditions, there was concern that significant overstory mortality would occur. We installed 32 randomly placed 10-m-radius circular plots directly after the fire in three community types: hardwood forest, pine-oak forest, and pine plantation. We identified and measured all trees ≥2.5 cm diameter at breast height (dbh) and determined scorch height and percentage, char height, and live or dead status. Plots were remeasured 1 and 2 years postburn to determine mortality. Overstory (>15.0 cm dbh) and midstory (≤15.0 cm dbh) stem densities were reduced significantly 1 year postburn by 5 and 64 percent respectively, but not significantly between 1 year postburn and 2 years postburn. The long-term outlook for the Southern Region includes hotter and drier conditions. This study provides resource managers with information on tree mortality following a lightning ignition during a drought in the High Peak area of the southeastern United States, and will allow more informed decisions to be made on the necessity of suppressing wildfires versus using them to restore and maintain forest structure and composition.

INTRODUCTION

Lightning ignitions occur at times of the year when plant and animal communities have evolved to survive fire (Frost 1998) and, in theory, these ignitions should better restore and maintain fire-adapted ecosystems than prescribed fire (Knapp and others 2009). Increasingly, fire managers are considering the use of natural ignitions, in combination with prescribed burns, to restore and maintain fire-adapted ecosystems (Cohen and others 2007). Due to the buildup of surface fuels from years of fire suppression and increasing drought indices associated with climate change, however, natural ignitions can have greater intensity and severity than occurred in the historical fire regime. As part of this buildup in fuels, ladder fuels can increase the chance of a destructive crown fire (Agee and Skinner 2005) and smoldering duff around tree trunks can cause root or basal cambial damage, especially to mature trees (Jenkins and others 2011, Varner and others 2005, Varner and others 2007, Varner and others 2009). Finally, fire-caused injury to drought-stressed trees can make them more susceptible to insects and disease outbreaks (Fettig and others 2010).

During the summer of 2011, the Ouachita National Forest (ONF) in Arkansas experienced a severe drought during which the Palmer Drought Severity Index (Palmer 1965) was -2.3 and the Keetch Byrum Drought Index (Keetch and Byram 1968) was over 700 (table 1). In conjunction with the drought, a series of thunderstorms produced many lightning ignitions on ridges in the central part of the Ouachita Mountains; in fact, more lightning ignitions were reported than any previous year in recorded history. Although most of these ignitions were managed with full suppression, one occurred in a remote and rugged area near High Peak on the Caddo-Womble Ranger District. Previous wildfires in this area had proved challenging to suppress. In the interest of

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Table 1—Fire weather indices during the High Peak Wildfire, Ouachita National Forest, Arkansas, 2011

Index	Mean	Range
Maximum temperature	40 °C (104 °F)	36–43 °C (97–110 °F)
Minimum relative humidity	27%	18–36%
10-hour fuel moisture	7	5–11
Keetch Byram Drought Index	705	670–725

firefighter safety given the rugged terrain and extreme heat—and with the opportunity to allow this natural ignition to potentially reduce stem densities and move stands towards the desired future conditions—ONF staff decided to use "less than full suppression" techniques, whereby the fire was allowed to burn naturally within a contained area. The High Peak Wildfire burned nearly 607 ha of the 729-ha designated containment area between 29 July and 11 August 2011. Heat and drought indices were extreme during this time (table 1).

Given the severe drought, some Forest Service personnel were concerned that the fire might cause excessive timber damage and mortality. Varner and others (2005) found that reintroducing fire into long-unburned longleaf pine (*Pinus palustris* Mill.) stands resulted in excessive (>50 percent) overstory pine mortality. This mortality may have resulted from smoldering duff around the bases of trees causing root and basal cambial damage, as mortality was positively correlated with decreasing duff moisture in these forests (Varner and others 2007). Overstory pine mortality in these studies, however, occurred following spring burns. As leaves begin to emerge in the spring, transpiration occurs, with water pulled through feeder roots to the new leaves. This causes duff moisture content to drop suddenly, leaving many feeder roots in the duff to be heated and killed by a smoldering fire. Knapp and others (2009) noted that in the summer, when the duff is dry, feeder roots have moved deeper in the soil; hence, duff consumption may not translate to tree mortality at this time of year. Many other factors, however, could also contribute to tree mortality, including fuel loading (Prichard and Kennedy 2012), crown damage due to scorch and/or consumption (Saveland and Neuenschwander 1990, Wyant and others 1986), and fire intensity (Prichard and Kennedy 2012, Regelbrugge and Conrad 1993).

Tree mortality has been studied in relation to wildfire (Regelbrugge and Conrad 1993), prescribed fire (Swezy and Agee 1991), and drought (Elliott and Swank 1994), but no studies have examined tree mortality in the Ozark-Ouachita Highlands in relation to lightning-ignited fire during a severe drought. The High Peak Wildfire provided a unique opportunity to document the combined effects of wildfire and drought on tree mortality. The objective of this research was to examine tree mortality following the High Peak Wildfire over a 2-year period. The Ouachita National Forest Plan indicates that a response to "unplanned, natural ignitions may include fire use, which is managing the ignition to accomplish specific resource management objectives in predefined areas" (Ouachita National Forest 2005). One of those objectives is fire fighter safety and another is returning fire to its historical role in defining forest structure and composition. Although we cannot attribute tree mortality directly to the fire because our data did not include unburned areas for comparison, we present results from a case study on tree mortality following naturally ignited fires under extreme drought conditions in the High Peak area.

METHODS

Study Site

The ONF is located in the Ouachita Mountain Ecoregion (Ouachita Ecoregional Assessment Team 2003) of western Arkansas and eastern Oklahoma. Ridges are underlain by Pennsylvania and Mississippi sandstone and shale valleys with clayey colluviums and covered with pine, pine-oak (*Quercus* L. spp.), and oak woodlands and forests (U.S. Department of Agriculture, Forest Service 1999). We conducted our study in the central part of the Ouachita Mountain Ecoregion on the Caddo-Womble Ranger District of the ONF at the High Peak Wildfire.

Data Collection

We randomly generated 50 points within the High Peak Wildfire containment area. We visited each point directly after the fire and installed rebar at the center of a 10-m-radius circular plot in burned areas (32 plots). Due to time constraints and lack of personnel, plots were not established in unburned areas at this time. Therefore, we were unable to distinguish between mortality caused by the wildfire and mortality due to other factors (e.g., drought).

The percentage of area burned per plot was assessed using ocular estimation. Ground-level burn severity was assessed using a rating system from 1 to 5, where: 5 =

unburned, 4 = scorched (litter partially blackened but wood and leaf structures unchanged), 3 = lightly burned (litter charred to partially consumed and upper duff may be slightly charred), 2 = moderately burned (litter mostly to entirely consumed and duff deeply charred), and 1 = heavily burned (litter and duff completely burned leaving fine white ash and mineral soil visibly altered).

On each plot, we identified and measured the diameter at breast height (dbh) of all trees ≥2.5 cm dbh and determined scorch height, percentage of crown volume scorched, char height, and live or dead status. Char is defined as the blackening of the boles of trees, whereas scorch is leaf mortality caused by radiant or convection heat. Since we were unable to collect preburn data, we used immediate postburn conditions to reconstruct the preburn composition of live versus dead trees. We assumed trees with scorched leaves were alive preburn. We classified trees >15 cm dbh as overstory and trees ≤15 cm as midstory. Based upon species composition, sites were classified into three forest types: hardwood, pine-oak, and pine plantation. Hardwood forests were dominated by white oak (*Quercus alba* L.), hickory (*Carya* L. spp.), blackgum (*Nyssa sylvatica* Marshall), and northern red oak (*Quercus rubra* L.); pine-oak forests were dominated by shortleaf pine (*Pinus echinata* Mill.), white oak, hickory, northern red oak, and red maple (*Acer rubrum* L.); and pine plantation by loblolly pine (*Pinus taeda* L.) All plots were remeasured 1 and 2 years postburn to determine mortality and changes in tree composition.

Eight 15.24-m planar intercepts (Brown 1974) were established randomly in the burn unit and measured preburn and immediately postburn to determine average fuel consumption on the burn. These transects were not associated with the 32 plots being used to assess tree mortality. We tallied preburn and postburn dead and down woody fuel that bisected each planar intercept. In the first 1.83 m of the planar intercept, we tallied 1- and 10-hour woody fuels (≤0.64 cm and 0.65-2.54 cm diameter, respectively). In the first 3.66 m of the planar intercept, we tallied 100-hour fuels (2.55-7.62 cm diameter). For logs (>7.62 cm diameter), we recorded diameter, decay class, and species type (hardwood or pine) along the entire 15.24-m transect. We sampled depth of litter and duff (to the nearest 0.25 cm) in an exposed profile using a trowel and ruler at 10 points along each planar intercept preburn and postburn.

Analyses

We first calculated frequency of species found in plots (percentage of plot where species occurred). We then calculated the density (live stems/ha) of trees per plot. Because plots were measured more than once and data were not normally distributed, we used generalized estimating equations (GEE) to model the effects of time since burn on live-stem density. The GEE model is an extension of the generalized linear model to longitudinal data analysis using quasi-likelihood estimation (Liang and Zeger 1986, Zeger and Liang 1986). The GEE models used a Poisson distribution and an exchangeable correlation structure (Littell and others 2002). Contrasts and Wald chi-square statistics were used to compare preburn and 1- and 2-year postburn live-stem densities.

RESULTS

White oak and shortleaf pine were the most frequently found overstory species in hardwood and pine-oak forest plots, found on 75 and 50 percent of preburn plots, respectively (table 2). Loblolly pine was the dominant species in pine plantations. Of all overstory frequencies, northern red oak frequency was the most reduced (table 2). The most frequent midstory species in preburn plots were red maple, blackgum, white oak, mockernut hickory (*Carya tomentosa* (Poir) Nutt.), and black cherry (*Prunus serotina* Ehrh.) (table 3). The frequency of all midstory species declined between preburn and 2 years postburn, but the frequency of common serviceberry [*Amelanchier arborea* (Michx. f.) Fernald], northern red oak, black cherry, flowering dogwood (*Cornus florida* L.), sweetgum (*Liquidambar styraciflua* L.), and red maple was reduced over 50 percent.

Total fuel loading was reduced from 22.0 t/ha to 13.7 t/ha. Litter was reduced from 7.2 to 2.5 t/ha (3.9 cm to 1 cm), duff was reduced from 6.9 to 5.2 t/ha (1.2 cm to 0.9 cm), and woody fuels were reduced from 7.8 to 6.1 t/ha. Note that while many logs were observed to be consumed during the High Peak Wildfire, no logs were measured in planar intercepts. Thus, this estimate of woody fuel consumption is most certainly an underestimation. Average char height was 0.20, 0.31, and 0.42 m and average scorch height was 4.51, 4.12, and 6.60 m in the hardwood forest, pine-oak forest, and pine plantation, respectively (table 4). Overstory scorch was 15, 18, and 34 percent in the hardwood forest, pine-oak forest and pine plantation, respectively. Plot area burned ranged from 76 percent in the hardwood forests to 86 percent in the pine-oak forest to 99 percent in the pine plantation. Plots in all communities burned with a moderate severity (table 4).

Across all forest types, the density of living stems was reduced 51 percent from preburn to 1 year postburn ($p < 0.05$) but only 2 percent from 1 year postburn to 2 years postburn ($p > 0.05$) (fig. 1). Midstory stem density was significantly reduced between preburn and 1 year postburn (64 percent, $p < 0.0001$) but not significantly between 1 year postburn and 2 years postburn (3 percent, $p = 0.1435$). Overstory stem density was significantly reduced from preburn to 1 year postburn (5 percent, $p = 0.0382$) but not significantly from 1 year postburn to 2 years postburn (1 percent, $p = 0.5095$) (fig. 1).

Table 2—Frequency of predominant overstory species (>8%) found in preburn and 2-year postburn plots on the High Peak Wildfire, Ouachita National Forest, Arkansas, 2011-2013

Common name	Scientific name	% Frequency	
		Preburn	Year 2
White oak	*Quercus alba* L.	75	72
Shortleaf pine	*Pinus echinata* Mill.	50	47
Northern red oak	*Quercus rubra* L.	41	25
Blackgum	*Nyssa sylvatica* Marshall	28	22
Mockernut hickory	*Carya tomentosa* (Poir) Nutt.	25	25
Black hickory	*Carya texana* Buckley	22	22
Loblolly pine	*Pinus taeda* L.	22	22
Red maple	*Acer rubrum* L.	19	13
Sweetgum	*Liquidambar styraciflua* L.	16	16
Black oak	*Quercus velutina* Lam.	13	16
Post oak	*Quercus stellata* Wangenh.	9	6
Black cherry	*Prunus serotina* Ehrh.	9	6

Table 3—Frequency of predominant midstory species (>30%) found in preburn and 2-year postburn plots on the High Peak Wildfire, Ouachita National Forest, Arkansas, 2011-2013

Common name	Scientific name	% Frequency	
		Preburn	Year 2
Red maple	*Acer rubrum* L.	78	38
Blackgum	*Nyssa sylvatica* Marshall	72	47
White oak	*Quercus alba* L.	72	44
Mockernut hickory	*Carya tomentosa* (Poir) Nutt.	69	38
Black cherry	*Prunus serotina* Ehrh.	69	25
Black hickory	*Carya texana* Buckley	56	41
Northern red oak	*Quercus rubra* L.	56	19
Winged elm	*Ulmus alata* Michx.	50	28
Flowering dogwood	*Cornus florida* L.	44	16
Shortleaf pine	*Pinus echinata* Mill.	38	28
Common serviceberry	*Amelanchier arborea* (Michx. f.) Fernald	34	9
Sweetgum	*Liquidambar styraciflua* L.	34	16
Hophornbeam	*Ostrya virginiana* (Mill.) K. Koch	34	25
Black oak	*Quercus velutina* Lam.	31	19

Table 4—Burn intensity and severity ratings for different forest types on the High Peak Wildfire, Ouachita National Forest, Arkansas, 2011

Burn variable	Hardwood (n = 9)		Pine-oak (n = 18)		Pine plantation (n = 5)		Total (n = 32)	
	-mean-	--SE--	-mean-	--SE--	-mean-	--SE--	-mean-	--SE--
Char height (m)	0.20	0.06	0.31	0.04	0.42	0.16	0.30	0.04
Scorch height (m)	4.51	0.98	4.12	0.44	6.60	1.17	4.62	0.42
Scorch (%)	46	10	54	5	58	10	52	4
Overstory scorch (%)	15	4	18	4	34	12	19	3
Plot area burned (%)	76	13	86	5	99	1	85	5
Severity rating (1-5)[a]	2.8	0.3	2.9	0.1	2.9	0.1	2.9	0.1
Total mortality (%)	42	9	47	6	49	8	46	4

[a] Burn severity: 1= high; 5 = unburned.
SE = Standard error.

Overstory density was higher in the pine plantations preburn and 2 years postburn (940 and 933 stems/ha) than both the hardwood community (307 and 267 stems/ha) and pine-oak community (379 and 355 stems/ha) (fig. 2). By forest type, overstory density was reduced 13, 6, and <1 percent 2 years postburn in the hardwood forest, pine-oak forest, and pine plantation, respectively, but none of the overstory reductions were significant ($p > 0.05$) (fig. 2). Preburn midstory density was higher in the pine plantations than hardwood and pine-oak communities (2247 versus 1459 and 1405 stems/ha, respectively), but similar to hardwood and pine-oak communities 2 years postburn (613 versus 704 and 440 stems/ha respectively) (fig. 3). By forest type, midstory density was significantly reduced from preburn to year 2 postburn by 52 ($p = 0.0245$), 69 ($p < 0.0001$), and 73 ($p = 0.0001$) percent in the hardwood forest, pine-oak forest, and pine plantation, respectively (fig. 3).

DISCUSSION

Stem density of woody plants in the Ozark-Ouachita Highlands was historically much lower than it is today (Spetich 2004). Chapman and others (2006) compared stem density of woody plants on the Sylamore Experimental Forest in northern Arkansas and found overstory stem density (≥14.1 cm dbh) increased from 124 stems/ha to 344 stems/ha between 1934 and 2002, while midstory stem density (4.1–14.0 cm) increased from 240 stems/ha to 688 stems/ha. Using the General Land Office data from the 19th century, Foti (2004) found historical stem densities of 121 stems/ha and 112 stems/ha in the Boston Mountains and Ozark Highlands, respectively. Preburn stem density of overstory and midstory canopy layers on High Peak was higher than historical conditions in all three community types (figs. 2, 3).

High stem density, often a result of fire suppression, can make stands more susceptible to insect outbreaks (Schowalter and others 1981) and high-intensity fire (Agee and Skinner 2005) or, in some cases, less susceptible to fire (Nowacki and Abrams 2008). Prescribed fire management objectives often include a reduction in midstory stems, an increase in herbaceous plant diversity, improved wildlife habitat, improved health and vigor of overstory trees, and restoration of historical forest composition and structure (Andre and others 2009, Guldin and others 2004). Attaining these objectives is possible with prescribed fire, but sometimes requires several burns or mechanical treatments (e.g., thinning) because prescribed burns are conducted under conservative burn prescriptions (Knapp and others 2009, Waldrop and others 2010). Lightning fires, on the other hand, usually occur during drought conditions when prescribed burns would typically be avoided. Thus, these fires may provide an avenue for attaining management objectives more efficiently.

Following the High Peak Wildfire, which ignited from a lightning strike during drought conditions, midstory and overstory densities were significantly reduced between preburn and 1 year postburn. Overstory mortality 1 year postburn was higher than typical background mortality (Clark and others 2008, Klos and others 2009), but mortality from 1 to 2 years postburn was not significant. Overstory density in hardwood and pine-oak forests was reduced to 267 and 356 stems/ha, respectively, moving them closer to historical stand structures. At the same time, pine plantations experienced little mortality and retained an overstory density of 933 stems/ha. The reduced competition as a result of midstory stem density reduction may benefit growth rates of

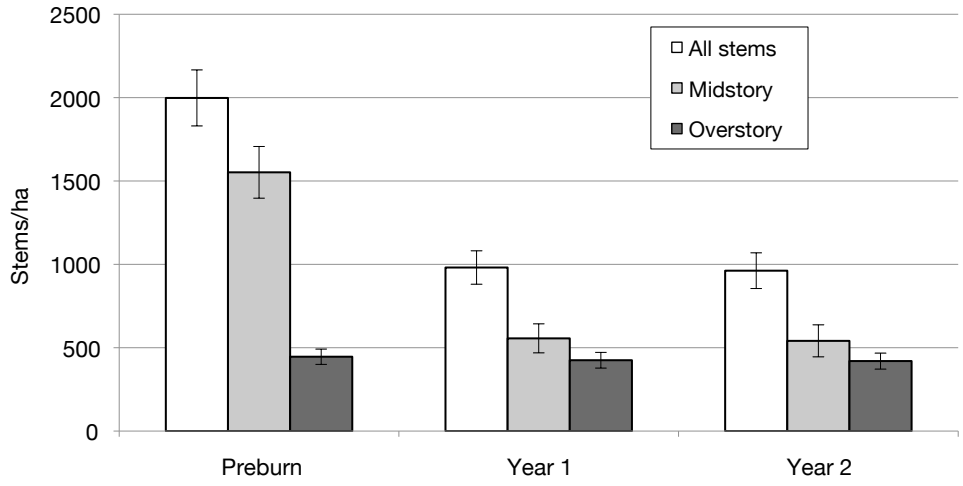


Figure 1—Density of total, midstory, and overstory stems from preburn to 2 years postburn on plots on the High Peak Wildfire in Arkansas, 2011-2013 (bars represent ±1 standard error). There was significant decline between preburn and 1 year postburn but not between 1 year and 2 years postburn in all stems, midstory, and overstory.

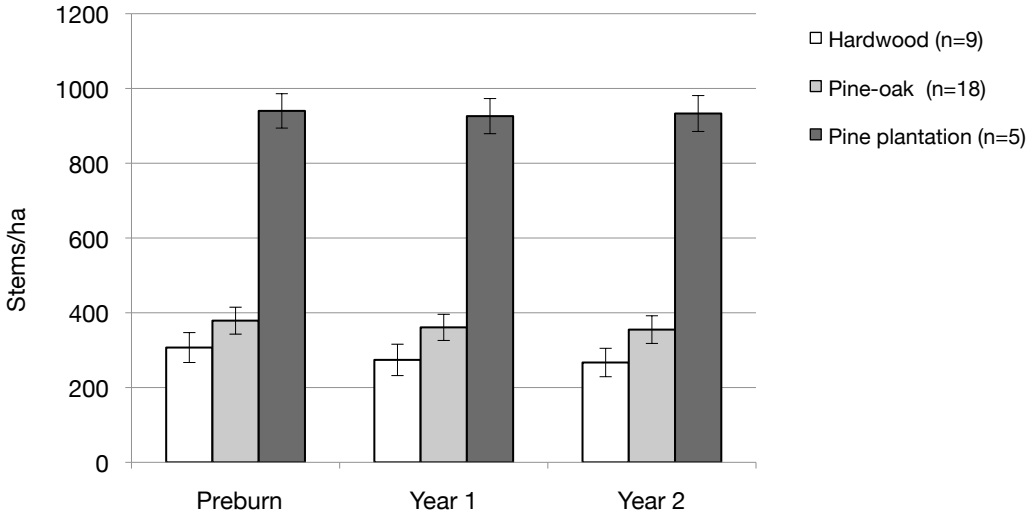


Figure 2—Overstory stem density by community type from preburn to 2 years postburn on plots on the High Peak Wildfire in Arkansas, 2011-2013 (bars represent ±1 standard error). There was no significant decline postburn in any overstory community.

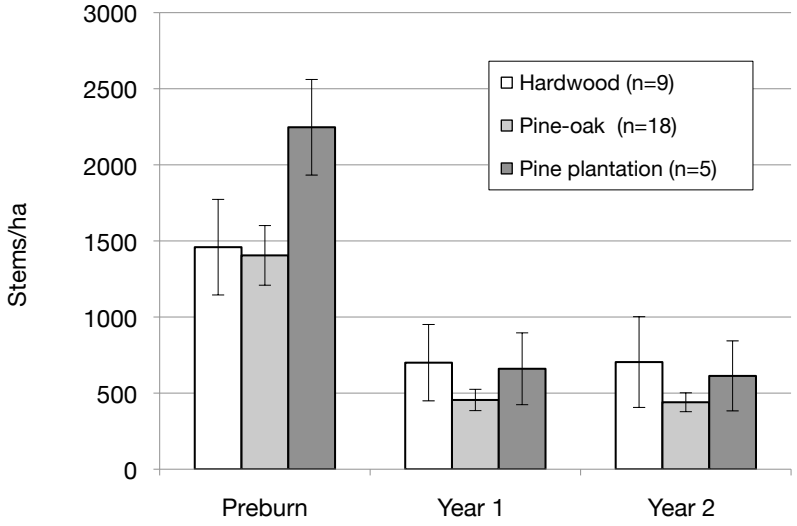


Figure 3—Midstory stem density by community type from preburn to 2 years postburn on plots on the High Peak Wildfire in Arkansas, 2011-2013 (bars represent ±1 standard error). There was significant decline between preburn and 1 year postburn but not between 1 year and 2 years postburn in hardwood, pine-oak and pine plantation communities.

overstory trees and allow for increased understory diversity (Masters and others 1998). Thinning of both the midstory and overstory could ultimately make the forest less susceptible to disease and insect outbreaks (Schowalter and others 1981). In conclusion, based upon the degree of mortality observed relative to historical stand densities, this fire assisted the ONF in reaching its resource management objectives.

The reduction in density we observed after the High Peak Wildfire has led other fire managers on the Ouachita National Forest to use "less than full suppression" tactics with other wildfires [Raspberry Mountain Wildfire, Caddo-Womble Ranger District (RD), 2012; Rough Branch Wildfire, Oklahoma RD, 2013; and Pipeline Wildfire, Jessieville-Winona-Fourche RD, 2013]. We will remeasure plots on the High Peak Wildfire 5 years postburn to document potential delayed mortality. Ultimately, we hope to expand this research to other areas and include control plots, thus enabling us to make mortality predictions in areas beyond the area of the High Peak Wildfire.

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LITERATURE CITED

Agee, J.K.; Skinner, C.N. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management. 211: 83-96.

Andre, J.; Anderson, M.; Zollner, D. [and others]. 2009. Landcape-scale fire restoration on the Big Piney Ranger District in the Ozark Highlands of Arkansas. In: Hutchinson, T.F., ed. Proceedings of the 3rd fire in eastern oak forests conference. Gen. Tech. Rep. NRS-P-46. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 126-132.

Brown, J.K. 1974. Handbook for inventorying downed woody material. Gen. Tech. Rep. INT-16. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 24 p.

Chapman, R.A.; Heitzman, E.; Shelton, M.G. 2006. Long-term changes in forest structure and species composition of an upland oak forest in Arkansas. Forest Ecology and Management. 236: 85-92.

Clark, S.; Spetich, M.; Evans, Z. 2008. Drought in the Southeast. Forest Wisdom. 12: 4-13.

Cohen, D.; Dellinger, B.; Klein, R.; Buchanan, B. 2007. Patterns in lightning-caused fires at Great Smoky Mountains National Park. Fire Ecology. 3(2): 68-82.

Elliott, K.J.; Swank, W.T. 1994. Impacts of drought on tree mortality and growth in a mixed hardwood forest. Journal of Vegetation Science. 5: 229-236.

Fettig, C.J.; McKelvey, S.R.; Cluck, D.R. [and others]. 2010. Effects of prescribed fire and season of burn on direct and indirect levels of tree mortality in ponderosa and Jeffrey pine forests in California, USA. Forest Ecology and Management. 260: 207-218.

Foti, T.L. 2004. Upland hardwood forests and related communities of the Arkansas Ozarks in the early 19th century. In: Spetich, M.A., ed. Proceedings of the upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 21-29.

Frost, C.C. 1998. Presettlement fire frequency regimes of the United States: a first approximation. In: Pruden, T.L.; Brennan, L.A., eds. Fire in ecosystem management: shifting the paradigm from suppression to prescription: proceedings, 20th Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station: 70-81.

Guldin, J.M.; Strom, J.; Montague, W.; Hedrick, L.D. 2004. Shortleaf pine-bluestem habitat restoration in the Interior Highlands: implications for stand growth and regeneration. In: Shepperd, W.D.; Eskew, L.C., comps. Silviculture in special places: proceedings of the 2003 national silviculture workshop. RMRS-P-34. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 182-190.

Jenkins, M.A.; Klein, R.N.; McDaniel, V.L. 2011. Yellow pine regeneration as a function of fire severity and post-burn stand structure in the southern Appalachian Mountains. Forest Ecology and Management. 262: 681-691.

Keetch, J.J.; Byram, G.M. 1968. A drought index for forest fire control. Res. Pap. SE-38. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 32 p.

Klos, R.J.; Wang, G.G.; Bauerle, W.L.; Rieck, J.R. 2009. Drought impact on forest growth and mortality in the southeast USA: an analysis using Forest Health and Monitoring data. Ecological Applications. 19(3): 699-708.

Knapp, E.E.; Estes, B.L.; Skinner, C.N. 2009. Ecological effects of prescribed fire season: a literature review and synthesis for managers. Gen. Tech. Rep. PSW-GTR-224. Albany, CA: U.S. Department of Agriculture Forest Service, Pacific Southwest Research Station. 80 p.

Liang, K-Y.; Zeger, S.L. 1986. Longitudinal data analysis using generalized linear models. Biometrika. 73: 13-22.

Littell, R.C.; Stroup, W.W.; Freund, R.J. 2002. SAS for linear models. 4th ed. Cary, NC: SAS Institute Inc. 466 p.

Masters, R.E.; Lochmiller, R.L.; McMurry, S.T.; Bukenhofer, G.A. 1998. Small mammal response to pine-grassland restoration for red-cockaded woodpeckers. Wildlife Society Bulletin. 26(1): 148-158.

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the eastern United States. Bioscience. 58: 123-138.

Ouachita Ecoregional Assessment Team. 2003. Ouachita Mountains ecoregional assessment. Little Rock, AR and Tulsa, OK: The Nature Conservancy. 41 p. + appendices.

Ouachita National Forest. 2005. Revised land and resource management plan: Ouachita National Forest, Arkansas and Oklahoma. Manage. Bull. R8-MB 124A. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region. 175 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsm9_039609.pdf [accessed 25 March 2012].

Palmer, W.C. 1965. Meteorological drought. Res. Pap. No. 45. Washington, DC: U.S. Department of Commerce Weather Bureau. 58 p.

Prichard, S.J.; Kennedy, M.C. 2012. Fuel treatment effects on tree mortality following wildfire in dry mixed conifer forests, Washington State, USA. International Journal of Wildland Fire. 21: 1004-1013.

Regelbrugge, J.C.; Conard, S.G. 1993. Modeling tree mortality following wildfire in *Pinus ponderosa* forests in the central Sierra Nevada of California. International Journal of Wildland Fire. 3(3): 139-148.

Saveland, J.M.; Neuenschwander, L.F. 1990. A signal detection framework to evaluate models of tree mortality following fire damage. Forest Science. 36(1): 66-76.

Schowalter, T.D.; Coulson, R.N.; Crossley, D.A., Jr. 1981. Role of southern pine beetle and fire in maintenance of structure and function of the southeastern coniferous forest. Entomological Society of America. 10(6): 821-825.

Spetich, M.A. 2004. Upland oak ecology – a synthesis. In: Spetich, M.A., ed., Proceedings of the upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 3-9.

Swezy, D.M; Agee, J.K. 1991. Prescribed-fire effects on fine-root and tree mortality in old-growth ponderosa pine. Canadian Journal of Forest Research. 21: 626-634.

U.S. Department of Agriculture, Forest Service. 1999. Ozark-Ouachita Highlands Assessment: terrestrial vegetation and wildlife. Report 5 of 5. Gen. Tech. Rep. SRS-35. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 201 p.

Varner, J.M., III; Gordon, D.R.; Putz, F.E.; Hiers, J.K. 2005. Restoring fire to long-unburned *Pinus palustris* ecosystems: novel fire effects and consequences for long-unburned ecosystems. Restoration Ecology. 13(3): 536-544.

Varner, J.M., III; Hiers, J.K.; Ottmar, R.D. [and others]. 2007. Overstory tree mortality resulting from reintroducing fire to long-unburned longleaf pine forests: the importance of duff moisture. Canadian Journal of Forest Research. 37: 1349-1358.

Varner, J.M.; Putz, F.E.; O'Brien, J.J. [and others]. 2009. Post-fire tree stress and growth following smoldering duff fires. Forest Ecology and Management. 258: 2467-2474.

Waldrop, T.; Phillips, R.A.; Simon, D.A. 2010. Fuels and predicted fire behavior in the southern Appalachian Mountains after fire and fire surrogate treatments. Forest Science. 56(1): 32-45.

Wyant, J.G.; Omi, P.N.; Laven, R.D. 1986. Fire induced tree mortality in a Colorado ponderosa pine/Douglas-fir stand. Forest Science. 32(1): 49-59.

Zeger, S.L.; Liang, K-Y. 1986. Longitudinal data analysis for discrete and continuous outcomes. Biometrics. 42: 121-130.

LONGLEAF PINE ADAPTATION TO FIRE: IS EARLY HEIGHT GROWTH PATTERN CRITICAL TO FIRE SURVIVAL?

G. Geoff Wang, Lauren S. Pile, Benjamin O. Knapp, and Huifeng Hu[1]

Abstract—Longleaf pine (*Pinus palustris* Mill.) forests are fire-dependent ecosystems because frequent surface fires prevent other species from being recruited into the canopy. The successful recruitment of longleaf pine has been attributed mainly to its unique fire adaptation – the grass stage. It is commonly believed that, while in the grass stage, longleaf pine seedlings build carbon reserves in the taproot, and this reserve is then mobilized to support fast height growth so that the apical meristem can quickly elevate above flame height. Based on this perception, we hypothesize that when longleaf pine emerges out of grass stage, (H1) height growth is a sudden process so that a critical threshold height can be reached quickly, and (H2) longleaf pine has faster height growth than its fire-susceptible co-genera, loblolly pine (*P. taeda* L.). To test H1, we examined early height growth patterns of planted longleaf pine seedlings. We found that height growth was a gradual rather than a "sudden" process, and there is no evidence for reaching a critical threshold value. To test H2, we conducted stem analysis for young longleaf and loblolly pine trees growing on the same sites. We found that longleaf pine, despite years in grass stage, did not grow any faster than loblolly pines when young. Our results suggest that the pattern and rate of height growth may not give longleaf pine any advantage for fire survival.

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.) historically dominated the southeastern United States, occurring on site types that included xeric sandhills, coastal plain flatwoods, and mountainous portions of Georgia and Alabama (Peet 2006). Its distribution range coincides with a frequent surface fire regime, with a historic return interval of 0-10 years (Brown and Smith 2000). This frequent surface fire regime maintained longleaf pine dominance throughout its range by preventing other tree species from being recruited into the canopy. For example, loblolly pine (*P. taeda* L.) was commonly restricted to wetter sites that experienced relatively infrequent fires (Schultz 1999), while upland oaks (e.g., *Quercus laevis*), despite their fire tolerance, were mostly restricted as a part of the understory vegetation in longleaf pine ecosystems (Wenk and others. 2011).

There is no doubt that the frequent surface fire regime has given longleaf pine a distinctive advantage over other tree species (Mattoon 1922, Wahlenberg 1946, Croker and Boyer 1975, Boyer 1990). As a large sapling or an adult tree, longleaf pine is very resistant to surface fire. Its apical meristems are well-protected because they are high up in the canopy and mostly out of the reach of surface fires. Additional protection to meristems is provided by insulation from needles and bud scales. Although the lower portion of the stem is exposed to fire, thick bark provides protection to the cambium. However, it is a long way to go from a tiny germinant to a fire-resistant large sapling or adult tree, and longleaf pine regeneration has to survive many repeated surface fires during this process. As a result, longleaf pine has evolved a unique adaptation to fire called the grass stage, a distinctive period of 2-20 years when seedlings remain stemless (Mattoon 1922, Wahlenberg 1946, Croker and Boyer 1975, Boyer 1990). It is believed that grass stage longleaf pine seedlings survive fire because the root collar is kept at the soil surface and the apical meristem is insulated by needles and bud scales. While in the grass stage, longleaf pine seedlings build carbon reserves in the taproot, and, once emerging out of the grass stage, this reserve is mobilized to support rapid height growth to elevate the apical meristem above flame heights (Boyer 1990).

However, our field observations suggest that rapid height growth may not be very important to the fire survival of longleaf pine regeneration. Prescribed fires often scorch the entire canopy of longleaf pine saplings, but most saplings survive nevertheless. Although some saplings may die due to fire, the mortality appears to be unrelated to height, which suggests that getting taller may not be critical to fire survival. In fact, young longleaf

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pine saplings retain dead needles on the stem (fig. 1). During burning, these dead needles likely act as a fuel ladder and spread fire to the upper part of the sapling, which could negate any advantage of growing tall. These circumstantial evidences present a compelling case to question the common wisdom, which believes that grass stage seedlings build carbon reserves in the taproot to support rapid initial height growth to quickly elevate the apical meristem above the flame of surface fires.

The objective of our study is to examine the role of early height growth patterns in fire survival. Based on the perceived importance of rapid height growth when longleaf pine emerges out of grass stage, we hypothesize that: (H1) height growth is a sudden process so that a critical threshold height can be reached quickly, and (H2) longleaf pine has faster height growth than loblolly pine, its fire-susceptible co-genera. To test H1, we used data obtained from longleaf pine seedlings planted for a longleaf pine restoration project conducted at Fort Benning, GA. To test H2, we used data obtained from naturally regenerated longleaf and loblolly pine saplings sampled at Brosnan Forest in Dorchester County SC.

METHODS

Two sets of data were used in the study. Dataset 1 was obtained from measurements of planted longleaf pine seedlings at Fort Benning GA, and dataset 2 came from conducting stem analysis of naturally regenerated longleaf and loblolly pines at Brosnan Forest, SC.

Dataset 1: This dataset was from a large project, in which longleaf pine seedlings were planted during winter 2007/2008 in clearcuts, loblolly pine stands thinned to residual basal areas ranging from 5-9 m^2/ha, and gaps (0.12-0.50 ha) created in loblolly pine stands. A detailed description of the project is given in Knapp and others (2013). We randomly selected and monitored 396 seedlings that had emerged out of the grass stage during the second (2009) or third (2010) growing seasons after planting, and we monitored those seedlings for five growing seasons. Height and diameter growth of each seedling were measured after the 2009, 2010, and 2012 growing seasons. We use 15 cm height (from the root collar to the terminal bud) as the threshold to classify a seedling as either in the grass stage or the height growth stage. Based on our 2009 measurements, seedlings were divided into those still in the grass stage and those in the height growth



Figure 1—Photos to illustrate the retention of dead needles on the stem of a young longleaf pine sapling (A and B). Photo B is a close-up photo of the lower stem portion of a longleaf pine sapling.

stage for one growing season. Similarly, we used the 2010 measurements to divide seedlings into those in the height growth stage for one growing season and those for two growing seasons. For our 2012 measurements, seedlings were divided into those in height growth for three growing seasons and those in height growth for four growing seasons.

Dataset 2: We sampled four stands with both naturally regenerated longleaf and loblolly pine saplings at Brosnan Forest. Between 5 and 10 dominant stems were destructively sampled for each species in each stand. A total of 29 stems were sampled for each species. For each destructively sampled stem, total height was measured in the field. Each stem sampled was marked at 0, 0.3, 0.6, 1.4, 2.4, and in 1 m intervals thereafter. A stem disk was taken from each marked position along the stem. Stem disks were taken to the laboratory, sanded using progressively finer sand paper, and the number of rings was counted on each disk. Based on the position of each disk and the number of rings counted, a series of height and age data were derived for each stem. We used Carmean's (1972) method of interpolation to derive the true height for the corresponding age recorded on each disk due to hidden tips between successive stem disks. For longleaf pine stems, the ranges of height, diameter at breast height, and age were 5-14 m, 4-9 cm, and 9-24 years, respectively. For loblolly pine stems, the ranges of height, diameter at breast height, and age were 5-15 m, 4-12 cm, and 5-25 years, respectively.

Data analyses: For dataset 1, we displayed height prior to the height growth stage (year 0) and height after one to four growing seasons using box plots. We also calculated frequency of seedlings in each of five height classes (< 30, 31-60, 61-90, 91-120, and >120 cm) after one to four growing seasons. For dataset 2, we plotted height over age, based on which we fitted the height-age relationship for each species using the Chapman-Richards model. The Chapman-Richards model has been commonly used to describe growth-age relationships (e.g., Pienaar and Turnbull 1973, Wang and others. 1994). The two resulting equations were then graphically compared to show differences in early height growth between longleaf and loblolly pine. Statistical analyses were conducted using SYSTAT 13 (Systat Software, Inc. 2009).

RESULTS AND DISCUSSION

Before entering the height growth stage, longleaf pine seedlings averaged 8.5 cm tall, which demonstrated a gradual, rather than sudden, increase in height when emerging from the grass stage (fig. 2). Seedlings

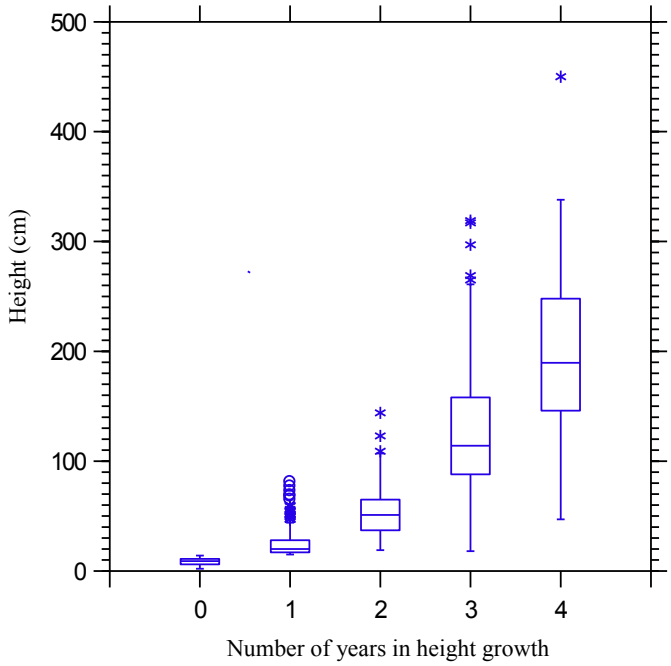


Figure 2—Boxplots showing the height of longleaf pine seedlings after different numbers of years in height growth. For each year, the height data are divided into four quartiles. The first and third quartiles from bottom to top form the box, which includes 50% of the data and divided evenly by the median in the center of box. One quartile of data each is above and below the box, illustrated by the two vertical lines (also called whiskers). The asterisks are called outside values. The dots are called far outside values.

averaged 24.2, 55.0, 124.9 and 193.2 cm tall, respectively, after their 1st, 2nd, 3rd and 4th years in height growth (fig. 2). The height growth rate accelerated over the first four years, and it averaged 16, 31, 70 and 68 cm per year during the 1st, 2nd, 3rd and 4th growing season, respectively, after emerging out of grass stage. Croker and Boyer (1975) suggested a critical height threshold of three feet (about 90 cm) for fire resistance during the height growth stage. We found that most seedlings did not reach this critical height after the first two growing seasons (table 1). After one growing season, none of the seedlings reached 90 cm tall. After two growing seasons, only 9.2 percent seedlings were 90 cm or taller. More than half of the seedlings (58.6 percent) were taller than 90 cm after three growing seasons. Most seedlings (close to 90 percent) were taller than 90 cm after four growing season (table 1).

Longleaf pine did not grow faster than loblolly pine, despite many years spent in the grass stage before initiating height growth (fig. 3). In fact, longleaf pine had slightly slower height growth than loblolly pine (fig. 3). Our results suggest that growing tall alone is not a viable strategy for fire survival. Indeed, we question the usefulness of using height to predict fire survival, but more studies are needed to fully understand the mechanisms of fire survival for juvenile longleaf pine.

Our results confirmed that longleaf pine seedlings could grow as much as 1.5 m in three years (Mattoon 1922). However, this growth rate was similar to or surpassed by a fire sensitive con-generic, loblolly pine, growing on the same sites. Other common competitors of longleaf pine, such as oaks, have the ability to sprout after fire, and it is likely their initial height growth is also comparable to longleaf pine. Given that longleaf pine, loblolly pine, and perhaps oak sprouts have comparable height growth rates, the success of longleaf pine and the failure of other species under a frequent surface fire regime must be attributed to other factors. Future studies should identify these factors and determine their role in the fire survival of small longleaf pine saplings (i.e., seedlings in early height growth stage).

CONCLUSIONS

Our results did not support our first hypothesis (H1), which height growth is a sudden process so that a critical height threshold can be reached during the first growing season. Longleaf pine seedlings did not emerge out of the grass stage suddenly. The annual height growth rate accelerated during the first four years after emerging from the grass stage, but it took 3-4 years for seedlings in the height growth stage to reach the perceived critical level (Croker and Boyer 1975).

Our results also failed to support the second hypothesis (H2), that longleaf pine has faster height growth than its fire-susceptible co-genera, loblolly pine. Despite many years in the grass stage, longleaf pine did not grow any faster in height than loblolly pine. In fact, loblolly pine grew consistently faster than longleaf pine during the first 20 years. Despite its faster height growth, however, loblolly pine regeneration fails completely in a frequent surface fire regime, with which longleaf pine copes extremely well.

Based on testing H1 and H2, we conclude that the early height growth pattern is not a critical factor responsible for fire survival of juvenile longleaf pine.

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Table 1—Percentage of longleaf pine seedlings in each height class after 1, 2, 3, and 4 years since initiating height growth

	Year 1	Year 2	Year 3	Year 4
Height < 30 cm	80.8	12.2	1	0
Height 30-60 cm	17.7	55.2	14.1	3.4
Height 60-90 cm	1.5	22.4	26.3	6.8
Height 90-120 cm	0	8.2	38.9	8.0
Height > 120 cm	0	1.0	19.7	81.8

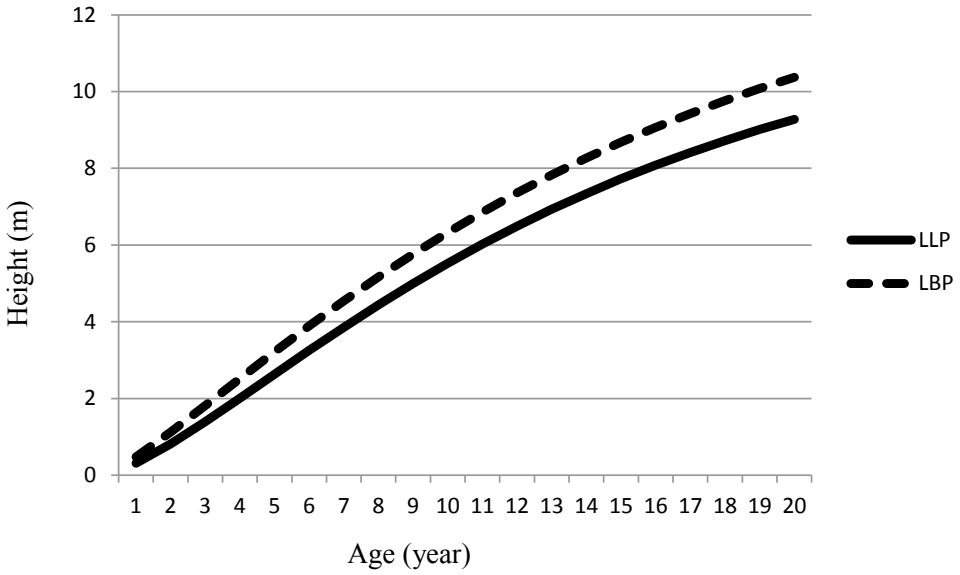


Figure 3—A comparison of height growth between longleaf (LLP) and loblolly (LBP) pine. Note that the age of longleaf pine is the number of years in height growth, while the age of loblolly is the true age since germination.

LITERATURE CITED

Boyer, W.D. 1990. *Pinus palustris* Mill. Longleaf pine. In: Burns, R.M.; Honkala, B.H., tech. cords. Silvics of North America. Vol. 1. Conifers. Agric. Handb. 65. Washington, DC: U.S. Department of Agriculture: 405-412.

Brown, J.K.; Smith J.K., eds. 2000. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-vol. 2. Ogden, UT: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 257 p.

Carmean W.H. 1972. Site index curve for upland oaks in the Central States. Forest Science. 18: 109-120.

Croker, T.C., Jr.; Boyer, W.D. 1975. Regenerating longleaf pine naturally. Res. Pap. SO-105. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 26 p.

Knapp, B.O.; Wang, G.G.; Walker, J.L. 2013. Effects of canopy structure and cultural treatments on the survival and growth of *Pinus palustris* Mill. seedlings underplanted in *Pinus taeda* L. stands. Ecological Engineering. 57: 46-56.

Mattoon W.R. 1922. Longleaf pine. Bull. No. 1061. Washington, DC: U.S. Department of Agriculture. 50p.

Peet, R.K. 2006. Ecological classification of longleaf pine woodlands. In: Jose, S.; Jokela, E.J.; Miller, D.L., eds. The longleaf pine ecosystem: ecology, silviculture, and restoration. New York: Springer: 51-94.

Pienaar, L. V.; Turnbull, K. J. 1973. The Chapman-Richards generalization of Von Bertalanffy's growth model for basal area growth and yield in evenaged stands. Forest Science. 19: 2-22.

Schultz, R.P. 1999. Loblolly – the pine for the twenty-first century. New Forests. 17: 71-88.

Systat Software, Inc. 2009. SYSTAT 13 User Manual. http://www.systat.com/Default.aspx. [Date accessed: September 15, 2015].

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth, and management. Washington, DC: Charles Lathrop Pack Forestry Foundation and U.S. Department of Agriculture Forest Service. 429 p.

Wang, G.G.; Marshall, P.L.; Klinka, K. 1994. Height growth pattern of white spruce in relation to site quality. Forest Ecology and Management. 68: 137-147.

Wenk, E.S.; Wang, G.G.; Walker, J.L. 2011. Within-stand variation in understory vegetation affects fire behavior in longleaf pine xeric sandhills. International Journal of Wildland Fire. 20: 866–875.

FIRST YEAR SPROUTING AND GROWTH DYNAMICS IN RESPONSE TO PRESCRIBED FIRE IN A MESIC MIXED-OAK FOREST

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Abstract—Prescribed fire is being used more frequently as a component of regeneration treatments in accordance with silvicultural guidelines developed to sustain and increase oak reproduction. A shelterwood-burn study was initiated in response to declining oak importance in the Allegheny Mountains of West Virginia. To remove a non-oak midstory, the study site was prescribe burned twice preceding the first removal cut of a two-cut shelterwood regeneration method that occurred in 2009/2010. A third burn occurred in April 2014 to mitigate the development of non-oak species that dominated the site four years after the seed cut. First year results indicate that northern red oak (*Quercus rubra*) survived at a higher rate than red maple (*Acer rubrum*), sweet birch (*Betula lenta*), and yellow-poplar (*Liriodendron tulipifera*), and the differences were more apparent in smaller pre-burn size classes. By contrast, differences in growth rates were significant in larger pre-burn size classes, but not in smaller size classes.

INTRODUCTION

Successful oak regeneration continues to be a challenge across the mixed oak and mixed mesophytic forests of the eastern United States. As a stand matures and overstory oaks grow in size, oak basal area per acre may increase but frequently it represents a decreasing proportion of the total basal area in the stand (Moser and others 2006). The decline in oak abundance can be attributed to either there being insufficient oak regeneration, or the regeneration present was not competitive enough to be recruited into the overstory as stands develop (Loftis 2004). Despite mitigation efforts, a decline in oak abundance is occurring in stands subjected to various management techniques as well as in unmanaged stands (Schuler and Miller 1995).

Declines are not geographically universal, but have been especially severe in the Allegheny/Appalachian plateaus in the central hardwood region (Fei and others 2011). Compositional shifts are more pronounced on mesic sites where oaks are being replaced by fire-sensitive species that are thriving in the absence of fire, including those that are shade tolerant and present in the understory at the time of disturbance, as well as those that are able to more rapidly respond to the disturbance (Smith 1993). Species on mesic sites may include shade-tolerant maples (*Acer* spp.) and American beech (*Fagus grandifolia*), and fast growing species may include yellow-poplar (*Liriodendron tulipifera*) and sweet birch (*Betula lenta*) (Schuler and Gillespie 2000, Schuler 2004, Trimble 1970).

Departure from the historic fire regime has altered canopy accession pathways and shifted understory species composition, resulting in increased forest density and understory shade (Rentch and others 2003). The oak-fire hypothesis suggests that fire defined the historical disturbance regime and can be judiciously used to maintain and restore oak forests (Arthur and others 2012, Brose and others 2014). In the last 50 years, significant advancements have been made in identifying the conditions in which prescribed fire is most likely to benefit oak communities, and silvicultural guidelines have been developed accordingly (Brose and others 2008, Brose and others 2013). Synthesis has revealed that overall, fire reduces the number of small diameter stems in the midstory, promotes establishment of new oak seedlings, preferentially selects oak reproduction over its mesophytic competitors, and reduces height differences between these species (Brose and Waldrop 2014).

There is still a need for additional information regarding the impact of fire on seedling sprouting, survival, and growth in mesic forests of the central Appalachians (Schuler and others 2013). The study discussed herein is part of a more comprehensive examination of the efficacy of using prescribed fire and shelterwood harvests (Brose and others 1999) to regenerate oak on two mesic mixed-oak sites the Fernow Experimental Forest in West Virginia. Two seedbed preparation burns were conducted in 2002/2003 and 2005 to remove interfering competition, primarily the shade-tolerant

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sapling layer, and create conditions suitable for the establishment of new oak seedlings. In 2009-2010, the seed cut of the shelterwood sequence reduced overstory basal area from 33.3 to 14.2 square meters per hectare, and from 267 to 109 stems/ha (dbh >12.7 cm) (Schuler and others 2013). The objective of this study was to determine whether the competitive position of oak regeneration could be improved by greater survival and improved height relative to its competitors post-burn. First year survival rates and post-burn sprout growth dynamics are examined herein.

METHODS

Study Area

The Fernow Experimental Forest is in the Allegheny Mountains section of the Central Appalachian Broadleaf Forest and is characterized by high, sharp ridges, low mountains, and narrow valleys (McNab and Avers 1994). Elevation of the study area is from 613 to 774 m. Annual precipitation is distributed evenly throughout the year and averages 148 cm, of which about 14 percent is snow. Mean annual temperature is approximately 9 °C. The growing season is about 149 frost-free days from May to October (Adams and others 2012).

Soils in the study area are Calvin channery silt loam, which is derived from acidic sandstone and shale, is well drained, moderately permeable, strongly acid to very strongly acid, and natural fertility is moderate to moderately low, with a minor inclusion of Gilpin channery silt loam, which is moderately permeable, very strongly acid to extremely acid, and has moderately low fertility (Schuler and others 2013).

Characteristic species on the Fernow include upland oaks, with northern red oak being most abundant, but also include chestnut (*Q. montana*), white (*Q. alba*), scarlet (*Q. coccinea*), and black oak (*Q. velutina*), yellow-poplar, black cherry (*Prunus serotina*), maples (*Acer* spp.), sweet birch, American beech (*Fagus grandifolia*), hickories (*Carya* spp.), basswood (*Tilia americana*), blackgum (*Nyssa sylvatica*), sassafras (*Sassafras albidum*), and white ash (*Fraxinus americana*). The natural conifer component consists of Eastern hemlock (*Tsuga canadensis*) and scattered red spruce (*Picea rubens*) (Madarish and others 2002). The study area could more specifically be described as mesic mixed-oak. Oak overstory species, represented in decreasing order of basal area, include northern red, chestnut, and white oak, and common associates include red maple, sugar maple (*A. saccharum*), sweet birch, and yellow-poplar (Schuler and others 2013). The study area site index is 21 m (age 50) for northern red oak and is characterized as a good site (Schnur 1937). Prescribe burn treatment areas encompass 13.8 ha between two locations 0.24 km apart.

Experimental Design and Data Collection

Seedling response plots were established in 10 overstory plots measuring 0.2 ha that were treated with prescribed fire. Five of the overstory plots were fenced and five were unfenced, facilitating the examination of deer herbivory effects. Five seedling response plots systematically distributed at the corners and middle of the overstory plots resulted in 50 total seedling response plots. Plot radius was extended outward to a distance necessary to measure sufficient sample seedlings, resulting in variable plot sizes from about 3.5 to 4.6 m. To more rigorously examine the individual response of reproduction, we tagged and measured three to four stems each of red maple, northern red oak, sweet birch, and yellow-poplar, with the intent of selecting stems so that the range of seedling diameters were as equally represented as possible.

Pre-burn seedling response plot measurements included species, stem height, ground line diameter, and presence of competition. The arboreal or non-arboreal species of the largest individual in a vertical cone with an angle extending 45 degrees above the terminal bud of the tagged seedling was determined to be the “worst aggressor” competition. Post-burn seedling data collected at the end of the 2014 growing season included stem impact class (kill, topkill and sprout, no topkill), number of sprout stems per rootstock, dominant sprout diameter, height to the highest live bud, and if browsed, the number of sprout stems browsed per rootstalk.

Unlike complete or random sampling, this sampling methodology was uniquely designed to capture an equal representation of diameters, the variable of interest, and inferences about diameter distribution or densities cannot be made from this sample on the population. Experimental design in the larger existing study captures those data.

Prescribed Fire Treatment

The dormant season prescribed fire was conducted on April 18, 2014. Except for a few larger yellow-poplar and sweet birch seedlings just starting to exhibit signs, leaf expansion had not yet begun. Ignition operations on the lower burn unit started at approximately 12:55 and total elapsed firing time was about 2 hours. Ignition operations in the upper burn unit were started at 16:00 the same day and total elapsed firing time was estimated to be 1 hour. During the burn window, temperature ranged from 19.5 to 22.0 °C, humidity ranged from 22 to 28 percent, and wind speed ranged from 0 to 13.7 km/hr. Temperature measurements during the fire were recorded by Hobo Type K data loggers connected to 25 cm long probes with thermocouples placed on a square grid at the corners

and center of the overstory plots, collocated at 48 of 50 seedling survival plots. The maximum temperature in the lower burn unit was 498.8 °C, the average maximum temperature was 171.0 °C, and the average duration >50 °C was 6.1 minutes. The maximum temperature in the upper burn unit was 498.8 °C, the average maximum temperature was 232.3 °C, and the average duration >50 °C was 6.4 minutes.

Data Analysis

Seedlings were divided by species into six pre-burn height classes for analysis. Chi-square analysis was used to test whether the number of stems surviving in each size class was evenly distributed between species. Analysis of variance (ANOVA) was used on the response variables post-burn height and diameter, with the Tukey-Kramer mean separation test to examine differences between species. All comparisons were evaluated at alpha equal to 0.05.

RESULTS

The 692 stems tagged within the 50 seedling response plots in the burn units were distributed as follows: 53 red maple, 236 red oak, 208 sweet birch, and 194 yellow-poplar. Of these seedlings, 520 had one or more aggressors. Arboreal aggressors were more common than non-arboreal aggressors. Worst aggressor was distributed by species as follows: 42 percent sweet birch, 24 percent blackberry (*Rubus* spp.), 10 percent greenbrier (*Smilax* spp.), 5 percent beech, 4 percent striped maple (*Acer pensylvanicum*), 4 percent red maple stump sprouts, and 3 percent yellow-poplar stump sprouts. Various other species represented the remaining 8 percent.

Survival

Of the 692 total stems tagged in the burn units, 691 were used in the data analysis. One seedling survived the fire without being topkilled and was excluded. Overall, red maple and yellow-poplar survival was low, with 40 and 48 percent, respectively. Sweet birch survival was moderate with 67 percent, and red oak survival was high with 87 percent. By pre-burn height class, there was a significant difference in survival between species only in the three smallest size classes. Oak survival was consistently high in all size classes (table 1).

Growth

ANOVA results indicated that there was no significant difference in post-burn average diameter between species in the smallest two and largest two pre-burn height classes. Post-burn average diameters differed between species in the middle two pre-burn height classes. Yellow-poplar diameter was significantly larger in both middle size classes (table 2). Similarly, ANOVA results indicated that there was no significant difference in post-burn average height between species in the smallest two and largest two pre-burn height classes. Post-burn mean heights differed between species in the middle two pre-burn height classes. Yellow-poplar height was significantly taller in both middle size classes (table 3).

DISCUSSION

The two previous seedbed preparation burns in earlier stages of this study were effective in reducing the maple component of the stand. Prior to the initial burning in 2002/2003 and 2005, maple had been most abundant, but after burning twice, maple, oak, and yellow-poplar were approximately equally represented; however, following the seed cut sweet birch became the most abundant (Schuler and others 2013). The reduction in red maple precluded finding an adequate sample size of red maple seedlings, particularly in the larger diameters, to tag for the study. Thus, it is hard to make any meaningful conclusions about red maple, especially at larger diameters and heights. The increased abundance of sweet birch was reflected in the "worst aggressor" assessment, representing 42 percent. Although there was not an overall shortage of sweet birch stems, very few were located that represented smaller diameters, illustrating their rapid growth.

Brose and others (2006) review of species sensitivity levels to fire was generally consistent with the results

Table 1—Average survival following the first growing season post-burn

		Pre-burn Height Class					
Species	Overall	0-60 cm	61-120 cm	121-180 cm	181-240 cm	241-300 cm	300+ cm
RM	40	24	61	100	—	—	—
RO	87	81	90	85	100	100	—
SB	67	33	24	63	73	78	83
YP	48	23	47	64	62	60	100
P-value	<0.001	<0.001	<0.001	0.032	0.077	0.754	1

Table 2—Average basal diameter in mm (±1 SE) following the first growing season post-burn.

	Pre-burn Height Class					
Species	0-60 cm	61-120 cm	121-180 cm	181-240 cm	241-300 cm	300+ cm
RM	3.7 (0.66)	4.5 (0.72)	5.3ab (1.91)	—	—	—
RO	4.4 (0.26)	5.9 (0.25)	8.0b (0.39)	9.7b (0.71)	9.1 (2.10)	—
SB	2.0 (0.84)	4.4 (1.20)	5.0a (0.55)	4.3a (0.37)	5.1 (0.50)	5.9 (0.44)
YP	4.4 (0.59)	6.4 (0.41)	9.9c (0.51)	12.9c (0.53)	5.2 (1.72)	8.6 (1.93)
P-value	0.058	0.09	<0.001	<0.001	0.202	0.193

* Means followed by different letters are significantly different within that height class (alpha=0.05)

Table 3—Average total height in cm (±1 SE) following the first growing season post-burn.

	Pre-burn Height Class					
Species	0-60 cm	61-120 cm	121-180 cm	181-240 cm	241-300 cm	300+ cm
RM	35.3 (6.9)	50.0 (8.5)	70.5b (21.1)	—	—	—
RO	29.0 (2.7)	50.3 (2.8)	75.4b (4.4)	104.9b (10.2)	81.5 (25.6)	—
SB	17.0 (8.8)	63.0 (14.2)	70.4b (5.8)	59.4c (5.4)	75.1 (6.1)	94.3 (6.1)
YP	33.8 (6.2)	60.0 (4.9)	93.0a (5.6)	136.0a (7.6)	54.0 (20.9)	87.5 (26.4)
P-value	0.362	0.321	0.033	<0.001	0.602	0.803

* Means followed by different letters are significantly different within that height class (alpha=0.05)

presented here. Northern red oak is considered resistant to fire, which was confirmed by its overall 87 percent survival rate. As a seedling, yellow-poplar is considered very sensitive to fire, which was confirmed in this study by an overall 48 percent survival rate. Sweet birch is considered fire-sensitive, which is somewhat supported by a 67 percent survival rate. Red maple is considered to be fire-sensitive to intermediate, depending on fire intensity, seasonality, and seedling size (Brose and others 2006, Ward and Brose 2004). Only 53 red maples in the desired size classes were located and most of them were less than 12.7 mm and 61 cm tall. About 40 percent of the red maple survived, illustrating their sensitivity at small sizes. Unfortunately, the small stature and size of the sample discourages making conclusions about larger red maple seedlings based on these results.

Greater differences in mortality among species are often reported in the small seedling sizes (Brose and Van Lear 1998, 2004). This is especially apparent when comparing oak to red maple, sweet birch, and yellow-poplar in the pre-burn height classes up to 120 cm. The probability of survival for oak is reported to increase with increasing diameter (Dey and Hartman 2005, Spetich 2013); however, for the range of seedling sizes examined in this study, probability of survival only slightly increased with size.

A small stem may be a seedling or a seedling sprout that has experienced one or more episodes of dieback and sprouting (Johnson and others 2009). Thus, stem size is not always indicative of age or root system size and consequently, does not always reflect the capacity to sprout, especially when growing on highly disturbed sites. Part of the reason that even small oaks were able to consistently sprout in this study was likely due to a higher root:shoot ratio that developed as seedlings survived two seedbed prep burns, as well as a harvest.

In addition to survival rates increasing with stem height for sweet birch and yellow-poplar, post-burn height growth increases with pre-burn stem height, illustrating that the capacity to control oak competitors may be reduced as the duration between the seed cut and the release burn increases. Although 67 percent of sweet birch survived, it did not recover its dominant pre-burn stature. While only 48 percent of yellow-poplar seedlings survived, survivor height growth was rapid. Reducing the height discrepancy between yellow-poplar, sweet birch, and oak is important in the short-term and, even if only temporarily, allows increased oak height growth and root development (Ward and Brose 2004). Ultimately, reducing the density of those competitors is also necessary.

These results indicate that red oak appears to exhibit a consistent ability to survive regardless of diameter, but it is questionable that the smaller diameter oaks will produce a sprout that will grow fast enough to compete successfully with other species, particularly yellow-poplar survivors, and become at least codominant in the stand. Although referring to post-harvest sprouting, not post-burn sprouting, Sander (1971) speculated that stems of oak advance reproduction should be >12.7 mm in diameter before it will produce a new sprout that will grow fast enough to compete successfully. Although oak seedlings with root collar diameters > 6.4 mm will likely sprout, it is recommended not to burn until oaks are approximately 12 mm in diameter and 30.5 cm tall (Brose and others 2014, Spetich 2013). Moreover, oaks with root collar diameters >19.1 mm, with sufficient light levels, are capable of relatively rapid, sustained height growth (Brose and others 2014). When assessing advanced regeneration in anticipation of a cut, "competitive oak" seedlings taller than 91.4 cm with root collar diameters greater than 19.1 mm are highly likely to be at least codominant at crown closure following overstory harvest (Brose and others 2008).

Following several years of monitoring, delayed mortality differences between species have been observed (Yaussy and Waldrop 2010), which suggests that the percent of surviving seedlings may decline at different rates. In addition to delayed mortality differences by species, delayed mortality can also differ on a size gradient (Sander 1972). The small oak stems < 60 cm showed a strong ability to survive the burn, but their persistence over the next several years will indicate if their survival was just a short-term phenomenon. Furthermore, height growth is frequently not linear during the first years following treatment, and even a relatively light residual overstory reduces growth of all reproduction (Sander 1972). By design, shade from the residual overstory trees will likely slow growth rates or induce mortality to a greater degree for more light-demanding species. It appears that oak responded positively to the burn, but a successful regeneration will in part hinge on how well sweet birch and yellow-poplar seedlings recover and if significant numbers of new germinants become established.

In earlier stages of this study, the influence of the seedbed preparation burns was apparent in the correlation between the post-fire conditions and enhanced red oak establishment. The timing of the 2005 burn before an abundant acorn crop was ideal, and density in 2006 was almost 4,047 oak seedling/ha of all sizes. Before the two seedbed prep burns there were about 20 larger oak seedlings/ha, but by 2009 the density of large oak seedlings was over 607 stems/ha. In contrast, there were almost no larger oak seedlings in the unburned control plots in the previous 10 years (Schuler and others 2013). Although fire helps improve the seedbed to facilitate successful oak germination and seedling establishment, it can also create favorable conditions for other species to germinate which may translate to an overall unchanged relative abundance of oak (Brose and others 2013). Reports of increased post-burn yellow-poplar seedling abundance are not uncommon, and sweet birch has also been particularly problematic (Barnes and Van Lear 1998, Shearin and others 1972, Schuler and others 2010). Following the two seedbed prep burns in 2002/2003 and 2005, sweet birch and yellow-poplar seedlings increased tenfold by 2006 (Schuler and others 2013). The seedbank was sampled prior to and after the fire in 2005 and sweet birch and yellow-poplar were 2 of 5 species comprising 76 percent of the post-burn seedbed (Schuler and others 2010). Enough seedlings survived or germinated following those burns and grew sufficiently for sweet birch to represent 42 percent of the "worst aggressor" population in 2014. Hence, fire can have unintended, although not unexpected, problematic effects. This challenge is not unique to disturbances following fire; yellow-poplar and sweet birch seedlings can have the highest importance of tallest stems and express canopy dominance in the high-light environment of young stands following clearcutting (Brashears and others 2004, Sander and Clark 1971). Stemming from prodigious seed production, seedbank accumulation, and rapid early height growth, controlling these species proves to be a significant management challenge faced on mesic sites.

First year results are a tentative assessment, but are important to incorporate in our understanding of oak ecology, stand development, and prescribed burning guidelines for regeneration of mixed-oak forests. Delayed mortality and growth rate dynamics may change in coming years, and more definitive conclusions can eventually be made about how the shelterwood-burn sequence affected the competitive position of oak. Brose (2010) presented promising results in a follow-up to an early shelterwood-burn study (Brose and Van Lear 1998) that show the increased density of oak stems and decreased red maple and yellow-poplar stems found 2 years post-fire, persisted after 11 years. Additionally, the number of dominant oak increased with fire intensity, while the number of red maple and yellow-poplar dominant stems decreased (Brose 2010). Monitoring stand development in coming years will help contribute to knowledge about effects of the shelterwood-burn method in mesic mixed-oak stands on good sites (SI 21 m) in the central Appalachian Mountains. With the ability to predict outcomes of seedbed preparation and release burns more reliably, developing silvicultural treatments will be easier to navigate.

LITERATURE CITED

Adams, M.B.; Edwards, P.J.; Ford, W.M. [and others]. 2012. Fernow Experimental Forest: Research History and Opportunities. Experimental Forests and Ranges EFR-2. Washington, DC: U.S. Department of Agriculture Forest Service. 26 p.

Arthur, M.A.; Alexander, H.D.; Dey, D.C. [and others]. 2012. Refining the fire-oak hypothesis for management of oak-dominated forests of the eastern United States. Journal of Forestry. 110: 257-266.

Barnes, T.A.; Van Lear, D.H. 1998. Prescribed fire effects on advanced regeneration in mixed hardwood stands. Southern Journal of Applied Forestry. 22(3): 138-142.

Brashears, M.B.; Fajvan, M.A.; Schuler, T.M. 2004. An assessment of canopy stratification and tree species diversity following clearcutting in Central Appalachian hardwoods. Forest Science. 50(1): 54-64.

Brose, P.H. 2010. Long-term effects of single prescribed fires on hardwood regeneration in oak shelterwood stands. Forest Ecology and Management. 260: 1516-1524.

Brose, P.H.; Van Lear, D.H. 1998. Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. Canadian Journal of Forest Research. 28: 331-339.

Brose, P.H.; Van Lear, D.H. 2004. Survival of hardwood regeneration during prescribed fires: The importance of root development and root collar location. In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Ashville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 123-127.

Brose, P.H.; Waldrop, T.A. 2014. Making sense out of confusion: a review of fire-oak papers published in the past 50 years. In: Waldrop, T. A., ed. Proceedings, wildland fire in the Appalachians: discussions among managers and scientists. Gen. Tech. Rep. SRS-199. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 12-24.

Brose, P.H.; Dey, D.C.; Waldrop, T.A. 2014. The fire-oak literature of eastern North America: synthesis and guidelines. Gen. Tech. Rep. NRS-135. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 98p.

Brose, P.H.; Dey, D.C.; Phillips, R.J.; Waldrop, T.A. 2013. A meta-analysis of the fire-oak hypothesis: Does prescribed burning promote oak reproduction in eastern North America. Forest Science. 59(3): 322-334.

Brose, P.H.; Gottschalk, K.W.; Horsley, S.B. [and others]. 2008. Prescribing regeneration treatments for mixed-oak forests in the Mid-Atlantic region. Gen. Tech. Rep. NRS-33. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 100 p.

Brose, P.H.; Schuler, T.M.; Ward, J.S. 2006. Responses of oak and other hardwood regeneration to prescribed fire: what we know as of 2005. In: Dickinson, M.B., ed. 2006. Fire in eastern oak forests: delivering science to land managers, proceedings of a conference; 2005. Columbus, OH. Gen. Tech. Rep. NRS-P-1. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 123-135.

Brose, P.H.; Van Lear, D.H; Cooper, R. 1999. Using shelterwood harvests and prescribed fire to regenerate oak stands on productive upland sites. Forest Ecology and Management. 113: 125-141.

Dey, D.C.; Hartman, G. 2005. Returning fire to Ozark Highland forest ecosystems: effects on advance regeneration. Forest Ecology and Management. 217: 37-53.

Fei, S.; Kong, N.; Steiner, K.C. [and others]. 2011. Change in oak abundance in the eastern United States from 1980 to 2008. Forest Ecology and Management. 262: 1370-1377.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2009. The ecology and silviculture of oaks. 2nd edition. New York: CABI Publishing. 580 p.

Loftis, D.L. 2004. Upland oak regeneration and management. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 163-167.

Madarish, D.M.; Rodrigue, J.L.; Adams, M.B. 2002. Vascular flora and macroscopic fauna on the Fernow Experimental Forest. Gen. Tech. Rep.NE-291. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station. 37p.

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

Moser, W.K.; Hansen, M.; McWilliams, W.; Sheffield, R. 2006. Oak composition and structure in the Eastern United States. In: Dickinson, M.B., ed. Fire in eastern oak forests: delivering science to land managers, proceedings of a conference; 2005 November 15-17; Columbus, OH. Gen. Tech. Rep. NRS-P-1. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 49-61.

Rentch, J.S.; Fajvan, M.A.; Hicks, Jr., R.R. 2003. Oak establishment and canopy accession strategies in five old-growth stands in the central hardwood forest region. Forest Ecology and Management. 184: 285-297.

Sander, I.L. 1971. Height growth of new oak sprouts depends on size of advance reproduction. Journal of Forestry. 69(11): 809-811.

Sander, I.L. 1972. Size of oak advance reproduction: key to growth following harvest cutting. Res. Pap. NC-79. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 6 p.

Sander, I.L.; Clark, F.B. 1971. Reproduction of upland hardwood forests in the central states. Agric. Handb. 405. Washington, DC: U.S. Department of Agriculture Forest Service. 25 p.

Schnur, G.L. 1937. Yield, stand, and volume tables for even-aged upland oak forests. Tech. Bull. No. 560. Washington, DC: U.S. Department of Agriculture Forest Service. 88 p.

Schuler, T.M. 2004. Fifty years of partial harvesting in a mixed mesophytic forest: composition and productivity. Canadian Journal of Forest Research. 34: 985-997.

Schuler, T.M.; Gillespie, A.R. 2000. Temporal patterns of woody species diversity in a central Appalachian forest from 1856 to 1997. Journal of the Torrey Botanical Society. 127(2): 149-161.

Schuler, T.M.; Miller, G.W. 1995. Shelterwood treatments fail to establish oak reproduction on mesic forest sites in West Virginia - 10-year results. In: Gottschalk, K.W.; Fosbroke, S.L., eds. Proceedings of the 10th central hardwood forest conference. Gen. Tech. Rep. NE-197. Radnor, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station: 375-387.

Schuler, T.M.; Thomas-Van Gundy, M.; Adams, M.B.; Ford, W.M. 2010. Seed bank response to prescribed fire in the central Appalachians. Res. Pap. NRS-9. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 9 p.

Schuler, T.M.; Thomas-Van Gundy, M.A.; Adams, M.B.; Ford, W.M. 2013. Analysis of two pre-shelterwood prescribed fires in a mesic mixed-oak forest in West Virginia. In: Miller, G.W., Schuler, T.M., Gottschalk, K.W., eds. Proceedings of the 18th central hardwoods forest conference. Gen. Tech. Rep. NRS-P-117. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 430-446.

Shearin, A.T.; Buner, M.H.; Goebel, N.B. 1972. Prescribed burning stimulates natural regeneration of yellow-poplar. Journal of Forestry. 70: 482-484.

Smith, H.C. 1993. Regenerating oaks in the central Appalachians. In: Loftis, D.L., McGee, C.E. (eds.). Oak regeneration: serious problems practical recommendations (symposium proceedings). Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 319 p.

Spetich, M.A. 2013. Regeneration dynamics during oak decline with prescribed fire in the Boston Mountains of Arkansas. In: Guldin, J.M., ed. 2013. Proceedings of the 15th biennial southern silvicultural research conference. SRS- Gen. Tech. Rep. 175. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 255-259.

Trimble, Jr., G.R. 1970. 20 years of intensive uneven-aged management: effect on growth, yield, and species composition in two hardwood stands in West Virginia. Res. Pap. NE-154. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 12 p.

Ward, J.S.; Brose, P.H. 2004. Mortality, survival, and growth of individual stems after prescribed burning in recent hardwood clearcuts. In: Yaussy, D.A.; Hix, D.M.; Long, R.P.; Goebel, P., eds. Proceedings of the 14th central hardwood forest conference. Gen. Tech. Rep. NE-316. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station: 193-199.

Yaussy, D.A.; Waldrop, T.A. 2010. Delayed mortality of eastern hardwoods after prescribed fire. In: Stanturf, J.A., ed. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS–121. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 609-612.

THE ESTABLISHMENT OF SHORTLEAF PINE FOLLOWING REPEATED PRESCRIBED BURNS AT CATOOSA WMA

John Bowers, Wayne Clatterbuck, Mike McCloy, Ben Royer, and Stephen Peairs[1]

Abstract—A mature shortleaf pine (*Pinus echinata*) stand on the Cumberland Plateau in Tennessee at the Catoosa Wildlife Management Area was harvested in 2001 in response to a regional southern pine beetle outbreak and converted into a savannah through periodic prescribed burns in 2005, 2010, and 2013. Following the harvest and series of burns, the stand was occupied by shortleaf pine seedlings and saplings of different sizes (<5 feet, 6 to 10 feet, and >10 feet) at a rate of approximately 400 stems per acre. Given the re-sprouting capability of shortleaf pine, the objective of this study was to assess the age of the shortleaf pine regeneration to determine if establishment occurred progressively over time in conjunction with known prescribed burn dates or during a single event prior to, during, or after the timber harvest in 2001. Shortleaf pines from each height class were aged above-ground level and below-ground level at or just above the basal crook to determine when they initiated and if and when they re-sprouted. An analysis of variance and post-ANOVA mean separation were used to determine differences amongst mean ages of each height class. Shortleaf pines from all three height classes had similar below-ground ages of approximately 13 years, indicating that they were of a single cohort initiating around the time of the 2001 timber harvest ($p=0.4104$). While shortleaf pines in the 6- to 10-foot height class and the >10-foot height class had similar above-ground ages of 12 and 10 years respectively, shortleaf pines currently less than five feet tall were significantly younger above-ground, averaging 6 years in age ($p<0.001$). Shortleaf pines currently less than five feet in height were more than likely top-killed in the 2005 prescribed burn and have since re-sprouted, while those currently greater than five feet tall were more than likely not top-killed in the 2005 burn. The fluctuations in age and growth of regenerating stems of shortleaf pine in this study are indicative of the mosaic of stand burns and their impact across the stand.

INTRODUCTION

Shortleaf pine (*Pinus echinata*) is a fire-adapted species, in that it possesses the ability to re-sprout following top-kill. The J-shaped basal crook, a physiological trait unique to shortleaf pine amongst most other southern pine species, occurs just below the ground surface and contains numerous dormant buds that are capable of sprouting when top-kill from a disturbance, such as fire, occurs (Mattoon 1915, Guldin 1986). This conveys shortleaf pine a competitive advantage over species that do not re-sprout. Shortleaf pine historically occupied the widest range of southern pine species, stretching across approximately 16.5 million acres of the eastern United States (Smith 1986).

Over the past 50 years, however, shortleaf pine populations have declined up to 50 percent due to a combination of active fire suppression, land conversion, beetle outbreaks, and selective timber harvesting (Guldin and others 1999, South and Buckner 2003). Today's shortleaf pine dominated forests are older and of larger diameter classes, with comparatively low occurrences of early successional shortleaf pine forests (Oswalt 2012). This signals further difficulties of sustaining shortleaf pine across its native range. Efforts to restore the shortleaf pine ecosystem have largely focused on prescribed fire as a beneficial management tool, because of the species' re-sprouting capabilities, the fire-adapted requirements of associated vegetation, and historical wildfire regimes (Guldin and others 2004, Bukenhofer and Hedrick 2013). The overall impact of prescribed fire on shortleaf pine establishment, however, is not fully understood. Research has shown that bark thickness, seedling age, seedling size, and fire intensity can all impact the re-sprout success and survival of shortleaf pine following prescribed burning (Lilly and others 2012, Clabo 2014).

In order to investigate shortleaf pine establishment and re-sprouting in response to disturbances, such as overstory harvesting and prescribed burns, a case study was conducted at the Catoosa Wildlife Management Area on the Cumberland Plateau in East TN during the fall of 2014. The study site was a mature shortleaf pine stand until harvest in 2001, and is currently an open-savannah type community with

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various heights of shortleaf pine regeneration. Given the re-sprouting capability of shortleaf pine, the objective of this study was to assess the age of the shortleaf pine regeneration to determine if establishment occurred progressively with time in conjunction with known prescribed burn dates in 2005, 2010, and 2013 or during a single event prior to, during, or after overstory removal in 2001.

METHODS

Sites

The study site for this research was located at the Catoosa Wildlife Management Area (WMA) in Cumberland County, TN, within the Cumberland Plateau physiographic region. The study was conducted in Unit S41-North at Catoosa WMA, which is presently an open savannah-type community, last prescribed burn in February/March of 2013, with a minimal overstory occupied by scattered residual oaks and a midstory dominated by varying heights of shortleaf pine regeneration at a rate of approximately 400 stems/acre. Three general height classes characterize the shortleaf pine regeneration: one to five feet, six to ten feet, and greater than ten feet. All shortleaf pines were generally shorter than 16 feet. Woody vegetation accounted for 10 to 20 percent of the area, composed of shortleaf pine, red maple (*Acer rubrum*), southern red oak (*Quercus falcata*), scarlet oak (*Quercus coccinea*), post oak (*Quercus stellata*), sourwood (*Oxydendrum arboreum*). Sweetgum (*Liquidambar styraciflua*), American beech (*Fagus grandifolia*), black cherry (*Prunus virginiana*), hickory (*Carya spp.*), viburnum (*Viburnum spp.*) and flowering dogwood (*Cornus florida*) occurred infrequently in the area. Annuals and perennials such as forbs, composites, and legumes, composed 5 to 10 percent of the plot cover, primarily represented by *Rubus spp*. Various grasses composed the remaining 75 percent of the plot cover.

Unit S41-North was harvested in 2001 in response to a southern pine beetle outbreak. Shortleaf pine dominated the pre-harvest stand, with an average DBH of 16 inches and basal area of 100 feet^2/acre, with scattered Virginia pine (*Pinus virginiana*) and various oak species (*Quercus* spp.). All shortleaf pines were removed in 2001, while the majority of oaks were left standing. An intense late-growing season burn, deemed the "Halloween Burn", was performed in the stand on October 31, 2005. The "Halloween Burn" was particularly intense and burned hotter than anticipated, likely due to the higher fuel loads of leftover woody debris from the 2001 timber harvest and the drier conditions normally associated with late growing season burns. The majority of leaf litter and duff were consumed, exposing mineral soil across the stand. Subsequent early-growing season burns of lower intensity were conducted in February/March of 2010 and 2013. These fires were cooler and did not expose the mineral soil, suggesting that they had a much lesser impact. The three prescribed fires conducted over this time period probably impacted the development of shortleaf pine regeneration across Unit S41-North to varying degrees.

PROCEDURES

Data Collection—To determine the age of establishment and sprouting in relation to disturbance history, shortleaf pine regeneration from each height class were destructively sampled and aged just above-ground level and below-ground level at or just above the basal crook. Within Unit S41-North, three parallel transects were established two chains apart. Three points, each two chains apart, were sampled along each transect line for a total of nine points. All distances were measured using standard pacing techniques. At each point along the transects, the closest height class one, Ht1 (one to five feet); height class two, Ht2 (six to ten feet); and height class three, Ht3 (greater than ten feet) shortleaf pines were collected. This procedure produced nine total samples from each height class and 27 samples overall (table 1).

When more than one sprout occurred on a single sample, the most dominant sprout was selected.

Table 1—Height class distribution and sample sizes of shortleaf pine regeneration in Unit S41-North of Catoosa WMA in Cumberland County, TN, 2014

Height Class	Stems per acre	Sample Size (n)
Ht1 (1-5ft)	98	9
Ht2 (6-10ft)	168	9
Ht3 (10ft+)	127	9
Total	393	27

Samples were cut to a more manageable size, both above-ground and below the basal crook. Each sample was labeled according to its specific transect point and height class, then taken to the lab for further analysis. To determine the density of shortleaf pine by height class, 26 1/20th acre plots were allocated equidistantly along three additional transects. Shortleaf pines were counted within each plot and categorized into the same three height classes used for age classification. Sample collection and field measurements occurred on November 7, 2014.

In the lab, the samples were cut with a band saw to reveal clean cross-sectional faces just above-ground level and below-ground level at or just above the basal crook. Each sample face was sanded with a belt sander until the growth rings were clear enough that the sample could be properly aged both above- and below-ground level. Using a magnifying lens and a 50x stereo microscope, the growth rings of both sides of each sample were counted. Due to the possibility of observer bias, multiple researchers independently aged each sample. An above-ground age and a below-ground age was determined for each sample, to differentiate between the sample's effective age, or sprout age, and its true age, or rootstock age. The sample preparation and aging took place during the week of December 1 - 5, 2014. Shortleaf pine sample ages were referenced to known dates of the overstory harvest and the prescribed burns.

Statistical Analysis

Analysis of variance (ANOVA) was employed to test for differences in above- and below-ground ages between height classes. A significance level of 5 percent was used. Following a significant ANOVA test, mean separation procedures were conducted utilizing Bonferonni's method to determine which height classes significantly differed.

RESULTS

Shortleaf Pine Density

Across Unit S41-North, shortleaf pines in Ht1 occurred at a rate of 98 stems per acre. Ht2 occurred at a rate of 168 stems per acre, and Ht3 occurred at a rate of 127 stems per acre. Overall shortleaf pine density equaled 393 stems per acre (table 1).

Shortleaf Pine Aging

The average ages of Ht1 were 6 years above-ground and 12 years below-ground (table 2). The average ages of Ht2 were 12 years above-ground and 14 years below-ground. The average ages of Ht3 were 10 years above-ground and 12 years below-ground.

The mean below-ground ages for all three height classes did not significantly differ ($p = 0.4104$). Ht1, Ht2, and Ht3 all had statistically similar below-ground ages (table 2). The mean above-ground ages for all three height classes did significantly differ ($p < 0.001$). The above-ground ages for Ht2 and Ht3 did not differ. However, Ht1 had a significantly younger above-ground age than Ht2 and Ht3.

The below-ground ages for all three height classes, 12 years, 14 years, and 12 years respectively, all occurred around the time of the 2001 overstory harvest 13 years ago. The mean above-ground ages for Ht2 and Ht3, 12 years and 10 years respectively, date to prior to the intense prescribed burn in 2005. The mean above-ground age for Ht1, 6 years, dates to after the 2005 prescribed burn occurred.

DISCUSSION

The similarity in below-ground ages for all three height classes reveals that the majority of shortleaf pine regeneration in Unit S41-North is of a single cohort, established 12 to 13 years ago, shortly after the timber harvest in 2001. The conditions following the timber

Table 2—Mean above- and below-ground ages and ANOVA results for three height classes of shortleaf pine regeneration in Unit S41-North of Catoosa WMA in Cumberland County, TN, 2014

Height Class	Below-ground age (yrs.)	Std. Dev.	Above-ground age (yrs.)	Std. Dev.
Ht1 (1-5ft)	12a[1]	4.6	6a	2.7
Ht2 (6-10ft)	14a	3.9	12b	3.7
Ht3 (10ft+)	12a	2.9	10b	1.9

[1] Means within each column not followed by same letter differ significantly at P = 0.05.

harvest would have been conducive for shortleaf pine re-establishment, with a residual seed source from the recently removed mature trees, an open canopy, and exposed mineral soil from the harvesting operations.

The differences in above-ground ages suggest that Ht1 pines were significantly younger than Ht2 and Ht3 pines. In relation to the 2005 prescribed burn, Ht1 had above-ground ages younger than the burn, while Ht2 and Ht3 had above-ground ages older than the burn. This occurrence is representative of the mosaic of impact that prescribed fire displays across a landscape. Shortleaf pines currently in Ht2 and Ht3 were more than likely not top-killed in the 2005 prescribed burn. This would explain why these pines are in the taller height classes. However, shortleaf pines currently occupying Ht1 were more than likely top-killed in the 2005 prescribed burn and have since re-sprouted, explaining why these pines are in the shortest height class. A large percentage of the initial cohort that regenerated after the 2001 timber harvest likely were killed entirely by the 2005 prescribed burn. Other studies have concluded that only 40 percent of shortleaf pine seedlings will re-sprout after top-kill and that the likelihood of re-sprouting decreases with increasing age and size (Lilly and others 2012, Clabo 2014). A similar scenario probably occurred on this study site.

The results of this study suggest that the 2005 prescribed fire was of varying intensity and impacted shortleaf pines differently across the landscape. Many shortleaf pine seedlings that regenerated after the 2001 timber harvest were likely killed entirely by the 2005 burn, while other seedlings were only top-killed and re-sprouted or were minimally impacted and not top-killed. This “mosaic” of fire impact across the landscape could be explained by variations in fuel loads and environmental conditions, specifically micro-topography and moisture levels, between microsites. Shortleaf pines occurring on more favorable, less-exposed, wetter, concave microsites had a greater chance of withstanding the impacts of fire than those occurring on less favorable, more-exposed, drier, convex microsites. The burns occurring in 2010 and 2013 seemed to have a much less dramatic impact on the shortleaf seedlings, but likely produced a similar mosaic of impact across Unit S41-North. Another important observation from this study is that periodic burning did not create new cohorts of shortleaf pine. A viable seed source was not present in this stand or adjacent stands once the shortleaf pine overstory was removed.
The results of this case study stem from a small sample size and reflect only a fairly small portion of the 80,000 acre Catoosa WMA with unique circumstances of a sanitation harvest prior to southern pine beetle and the resulting creation of a savannah through a series or prescribed fires. Only current vegetation was available to sample, making it difficult to characterize the vegetation and shortleaf pine density before the 2005, 2010, and 2013 burns.

SILVICULTURAL IMPLICATIONS

Prescribed burning can be a useful tool when managing shortleaf pine stands for site preparation, hardwood competition control, and the maintenance of fire-dependent, associated species, such as bluestem (*Andropogon spp.*). Site preparation is perhaps the most applicable use of prescribed fire in shortleaf pine stands, as a well-conducted burn can expose mineral soil and improve regeneration conditions. However, burning does not create new cohorts in the absence of an active shortleaf pine seed source. For this reason, it can be assumed that burning too frequently in young stands could lead to a decline in the shortleaf pine component. If shortleaf pine restoration is a goal in stands where no seed source exists, planting may become necessary. Burning promotes a mosaic of vegetational structures across a stand or landscape. Microsite variations have unavoidable influence on prescribed burning. The results of this study suggest that burning prior to 5 years of age encourages top-kill and re-sprouting, but the overall re-sprout rate of shortleaf pine remains unknown. Prescribed burning can play a key role in shortleaf pine management regimes, but the results of the practice are highly variable with an array of impacts on the structure of the vegetation.

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LITERATURE CITED

Bukenhofer, G.A.; Hedrick, L.D. 2013. Shortleaf pine/bluestem renewal. U.S. Department Agriculture Forest Service, Ouachita National Forest Arkansas. http://www.fs.usda.gov/detailfull/ouachita/home/?cid=fsm9_039689&width=full>. [Date accessed: May 4, 2015].

Clabo, David C. 2014. Shortleaf pine sprout production capability in response to disturbances. Knoxville, TN: University of Tennessee. 76 p. M.S. thesis.

Guldin, J.M. 1986. Ecology of shortleaf pine. In: Murphy, P.A., ed. Symposium on the shortleaf pine ecosystem. Monticello, AR: Arkansas Cooperative Extension Service: 25-40.

Guldin, J.M; Strom, J.; Montague, W.; Hedrick, L.D. 2004. Shortleaf pine-bluestem habitat restoration in the interior highlands: Implications for stand growth and regeneration. In: Shepperd, W.D.; Eskew, L.G., comps. Silviculture in special places: proceedings of the national silviculture workshop. RMRS-P-34. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 182-190.

Guldin, J.M.; Thompson, F.R.; Richards, L.L.; Harper, K.C. 1999. Chapter 3: Status and trends of vegetation. In: Ozark-Ouachita highland assessment, terrestrial vegetation and wildlife. Gen. Tech. Rep. SRS-35. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 21-70.

Lilly, C.J.; Will, R.E.; Tauer, C.G. [and others]. 2012. Factors affecting the sprouting of shortleaf pine root stock following prescribed fire. Forest Ecology and Management. 265: 13-19.

Mattoon, W.R. 1915. Life History of Shortleaf Pine. Bull. 244. Washington, DC: US Department of Agriculture. 46 p.

Oswalt, C.M. 2012. Spatial and temporal trends of the shortleaf pine resource in the eastern United States. In: Kush, J.; Barlow, R.J.; Gilbert, J.C., eds. Proceedings of the shortleaf pine conference: east meets west, bridging the gap with research and education across the range. Special Report No. 11. Auburn, AL: Alabama Agricultural Experiment Station: 33-37

South, D.B.; Buckner, E.R. 2003. The decline of southern yellow pine timberland. Journal of Forestry. 101: 30-35.

Smith, K.L. 1986. Historical perspective. In: Proceedings of the symposium on the shortleaf pine ecosystem. Monticello, AR: Arkansas Cooperative Extension Service: 1-8.

Biometrics

Moderator:

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QUANTIFYING AND MAPPING SPATIAL VARIABILITY IN SIMULATED FOREST PLOTS

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Abstract—We used computer simulations to test the efficacy of multivariate statistical methods to detect, quantify, and map spatial variability of forest stands. Simulated stands were developed of regularly-spaced plantations of loblolly pine (*Pinus taeda* L.). We assumed no affects of competition or mortality, but random variability was added to individual tree characteristics. The purpose of simulating stands without these complex interactions was to provide a controlled situation to measure the efficacy of our methods. We examined redundancy analysis, partial redundancy analysis, and spatially constrained cluster analysis for detecting spatial patterns and found that redundancy analysis and partial redundancy analysis were reliable methods to quantify and test spatial dependence, respectively. Spatially constrained cluster analysis had moderate success in mapping variability, but its application to more complex situations may be limited.

INTRODUCTION

Information is lacking on reliable methods to detect, quantify, and map spatial heterogeneity in small-scale forest plots. This study focused on individual plots of even-aged, regularly-spaced plantation stands. Increased efforts to improve stand uniformity have lead to questions as to why spatial patterns of tree growth emerge. Spatially recognizable growth patterns of trees may be caused by many factors, in particular the effects of genotype and microsite variation are thought to greatly influence tree characteristics within a plot. Tree growth within a stands of different genetic stocks is an important topic (Buford and Burkhart 1987, Magnusson and Kremer 1993, Tang and others. 2001). Moreover, Oliver and Larson (1996) noted that polymorphism is likely to occur due to variations in microsite productivity. Both conceptually and mathematically, there exist, to some extent, a confounding of genetic and microsite effects on tree growth. A logical first step in understanding the underlying causes of structural dissimilarities in tree growth is to better understand how spatial patterns emerge and to quantify those effects. In order to investigate spatial effects on tree growth we first simulated then tested the efficacy of statistical methods to detect, quantify and map spatial heterogeneity.

METHODS

The study material was made up of five simulated plots of planted loblolly pine (*Pinus taeda L.*) (fig. 1). Plots consisted of 25 rows and 25 columns of even-aged loblolly pine. Each plot was assigned 1 of 5 microsite patterns. Microsite patterns are distinct formations of high or low areas of productivity within each stand. The patterns of microsites were chosen to represent a range of possible site conditions and, most importantly, to induce spatial dependence of tree structure. By creating areas of high and low productivity, we make the characteristics (DBH and height) of trees dependent on their location in space. Microsite patterns consisted of 1 to 5 microsites per plot: a control (uniform productivity), biplot (2 microsites, triplot (3 microsites), quad plot (4 microsites), and the free plot (5 microsites). All five site patterns were established without the complexities of competition and mortality for an initial assessment of sensitivity analysis of statistical methods for assessing patterns before progressing to increasingly more complex situations.

Site index (SI) and diameter at breast height (DBH) values were drawn from normal distributions. Site index and DBH distributions were unique to microsites within plots. Microsites with higher SI values have higher DBH values. Height and diameter relationship equations from loblolly pines were used to allocate heights. Lastly we used grid coordinates to assign spatial variables to each tree. Our model with covariates was then:

$$\mathbf{Y}_{625x2} = \mathbf{X}_{625x1} + \mathbf{W}_{625x2} \rightarrow [\text{DBH, Height}] = [\text{SI}] + [X_{\text{coordinate}}, Y_{\text{coordinate}}]$$

Redundancy analysis which is synonymous with explained variance (Legendre & Legendre 2012) was implemented using R software's "VEGAN" package. The first step to RDA is to center all variables. The variables mentioned hereafter are considered centered.

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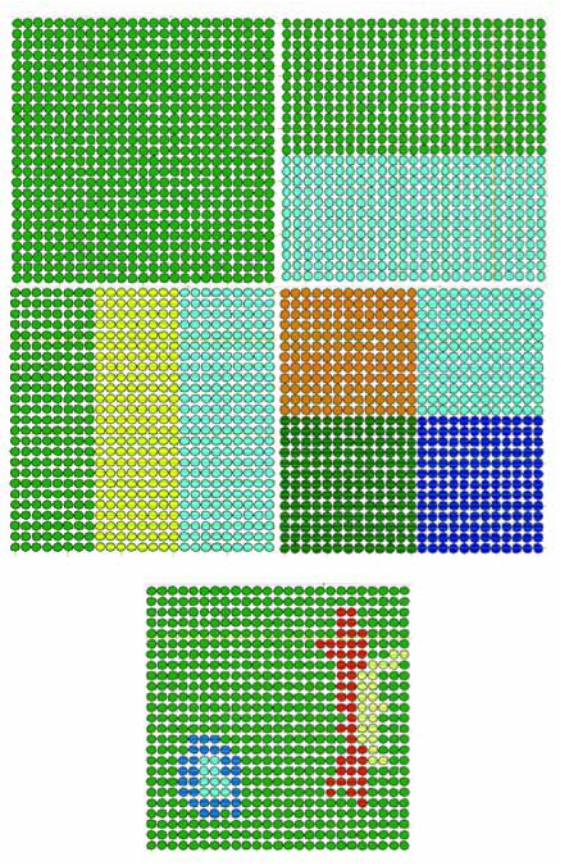


Figure 1—Illustrates the 5 spatial patterns used in this study Microsites are distinguished by color. For left to right: Control, biplot, triplot, quadplot, and freeplot.

This symmetric form of analysis utilizes a response matrix $\mathbf{Y}_{625x2}$ (hereafter **Y**) with explanatory vector $\mathbf{X}_{625x1}$ (hereafter **X**) and covariables $\mathbf{W}_{625x2}$ (hereafter **W**). Three different simple RDA's were performed to estimate the pure spatial, pure environmental, spatially structured environmental, and residual variability. The pure spatial variability could be isolated and tested for significance by partial redundancy analysis (pRDA).

Partial redundancy analysis was done using R software's "VEGAN" package, is the partial canonical redundancy analysis of our response matrix **Y** on matrix **X** while controlling for the linear effect of matrix **W** of covariables. For our specific objectives, we tracked the significance testing of **Y**~**W**|**X**. This is the hypothesis of spatial dependence and tests the significance of a pure spatial component, representing a hypothesis of spatial dependence.

The clustering algorithms we used come from the R software package "const.clust". This clustering algorithm is an agglomerative approach with a constraint of spatial contiguity. The spatial contiguity constraint allows only trees that are neighbors to each other in space to be grouped together. There are two important steps that must be performed in order to execute this algorithm. The first is to determine a dissimilarity metric and build a dissimilarity matrix; the second is to pick a connection method and build a contiguity matrix. We used Euclidean distance to build our dissimilarity matrix and Delaunay triangulation to define neighbors.

For each of our spatial patterns we imposed a range of scenarios (table 1). Each scenario specifies a difference in mean diameter at breast height (DBH) values among microsites and the value of coefficient of variation used to draw diameter values from normal distributions for individual trees.

RESULTS

Using RDA we performed a variation partitioning and estimate of the pure spatial variability. Table 2 contains the estimated partial R^2 value associated with the pure spatial component. As discussed earlier, there are 15 scenarios per spatial pattern. The scenarios are a combination of difference in mean DBH and a specified coefficient of variation (CV). Table 1 contains all possible combinations of CV and difference in mean DBH. Notably, the partial R^2 values for the control plot

Table 1—Illustrates 15 possible scenarios for each spatial pattern with k>1 microsites. Each scenario will be simulated for each spatial pattern. The control plot will be simulated with all 5 levels of CV

Difference in mean DBH (inches)	Coefficient of Variation 5%	10%	15%	20%	25%
0.5	(0.5,5%)	(0.5,10%)	(0.5,15%)	(0.5,20%)	(1,25%)
1	(1,5%)	(1,10%)	(1,15%)	(1,20%)	(1,25%)
2	(2,5%)	(2,10%)	(2,15%)	(2,20%)	(2,25%)

Table 2—Estimated partial R2 for each spatial pattern and all scenarios

Coefficient of Variation	BIPLOT	TRIPLOT	QUADPLOT	FREE	Control	
5%	29%	12%	5%	1%	0%	
10%	11%	7%	3%	1%	0%	0.5 inch
15%	5%	4%	2%	1%	0%	
20%	3%	2%	1%	0%	0%	
25%	2%	1%	1%	0%	0%	
Coefficient of Variation	BIPLOT	TRIPLOT	QUADPLOT	FREE	Control	
5%	47%	18%	6%	1%	0%	1 inch
10%	27%	14%	5%	1%	0%	
15%	16%	10%	4%	1%	0%	
20%	10%	8%	3%	1%	0%	
25%	6%	5%	2%	1%	0%	
Coefficient of Variation	BIPLOT	TRIPLOT	QUADPLOT	FREE	Control	
5%	56%	20%	7%	1%	0%	2inch
10%	45%	19%	7%	1%	0%	
15%	33%	16%	6%	1%	0%	
20%	24%	13%	5%	1%	0%	
25%	17%	11%	0%	0%	0%	

of 0 percent reflected the lack of spatial dependence. The largest R^2 values are in the biplot where spatial dependency was more pronounced in the data.

Partial redundancy analysis was used to detect spatial dependence. The results we report are the probability of detecting spatial dependence for all combinations of spatial patterns and scenarios. Table 3 contains the results from our pRDA simulations. The highest probabilities for detection are for the biplot and with larger differences in DBH among microsites. The probability for detecting spatial dependence in the control plot is consistently at about 5 percent. This reflects the type 1 error rate of **α=0.05**, which is the probability of rejecting the null hypothesis when the null hypothesis is true.

Cluster analysis was performed to examine the efficacy in detecting the number of microsites in simulated plots and to correctly allocate each tree to the correct microsite. Table 4 illustrates the probability of detecting the correct number of microsites for all combinations of spatial pattern and scenario. The control spatial pattern is excluded because it is homogeneous and the clustering algorithm does not test for k=1 groups in the data. Homogeneous stands are tested with spatial dependency tests such at the pRDA method described earlier. Much like the other methods, cluster analysis worked best in the biplot and where DBH differences were greatest.

Table 4 displays the probabilities for detecting the correct number of microsites. We used a threshold for reliability is 80 percent success. Therefore, in all cells of table 4 where the probability is greater than or equal to 80 percent we ran misclassification simulations. Table 5 shows the probability of misclassifying any given tree when using constrained cluster analysis. Values are generally low, but range from approximately 0 percent to 32 percent.

DISCUSSION

The variation partitioning and quantification of spatial variability performed as expected. Referring to table 2, we see that there is a general decrease in the amount of spatial variation captured as complexity of spatial pattern increases. This confirms that spatial pattern complexity and perhaps the shape of microsites

Table 3—Illustrates the results from pRDA. The values represent the probability of detecting spatial dependence

Coefficient of Variation	*BIPLOT*	*TRIPLOT*	*QUADPLOT*	*FREE*	*Control*	
5%	100%	100%	100%	94%	5%	
10%	100%	100%	100%	64%	5%	0.5 inch
15%	100%	99%	91%	36%	7%	
20%	98%	92%	67%	24%	6%	
25%	88%	72%	49%	19%	5%	
Coefficient of Variation	*BIPLOT*	*TRIPLOT*	*QUADPLOT*	*FREE*	*Control*	
5%	100%	100%	100%	99%	6%	
10%	100%	100%	100%	97%	4%	1 inch
15%	100%	100%	100%	89%	5%	
20%	100%	100%	100%	78%	6%	
25%	100%	100%	98%	64%	4%	
Coefficient of Variation	*BIPLOT*	*TRIPLOT*	*QUADPLOT*	*FREE*	*Control*	
5%	100%	100%	100%	99%	6%	
10%	100%	100%	100%	99%	4%	2inch
15%	100%	100%	100%	95%	5%	
20%	100%	100%	100%	93%	6%	
25%	100%	100%	100%	85%	4%	

Table 4—Illustrates the probability of correctly identifying the number of microsites in all combinations of spatial patterns and scenarios

Coefficient of Variation	*BIPLOT*	*TRIPLOT*	*QUADPLOT*	*FREE*	
5%	90%	52%	28%	0%	
10%	80%	22%	0%	0%	0.5 inch
15%	68%	26%	4%	4%	
20%	58%	28%	4%	2%	
25%	50%	10%	0%	0%	
Coefficient of Variation	*BIPLOT*	*TRIPLOT*	*QUADPLOT*	*FREE*	
5%	100%	100%	100%	0%	
10%	97%	84%	14%	27%	1 inch
15%	80%	12%	0%	14%	
20%	74%	12%	0%	2%	
25%	70%	10%	0%	0%	
Coefficient of Variation	*BIPLOT*	*TRIPLOT*	*QUADPLOT*	*FREE*	
5%	100%	100%	100%	0%	2 inch
10%	100%	100%	100%	0%	
15%	96%	34%	62%	10%	
20%	96%	4%	6%	6%	
25%	74%	0%	0%	0%	

Table 5—Illustrates the probability of misclassifying trees for all combinations of spatial patterns and scenarios where the probability of correctly identifying the correct number of microsites (Table 4) was above 80 percent. (*) indicates a combination where the probability from table 4 was less than 80 percent

Coefficient of Variation	*BIPLOT*	*TRIPLOT*	*QUADPLOT*	
5%	0.068	*	*	
10%	0.316	*	*	0.5 inch
15%	*	*	*	
20%	*	*	*	
25%	*	*	*	
Coefficient of Variation	***BIPLOT***	***TRIPLOT***	***QUADPLOT***	
5%	0.004	0.012	0.009	
10%	0.080	0.276	*	1 inch
15%	0.159	*	*	
20%	*	*	*	
25%	*	*	*	
Coefficient of Variation	***BIPLOT***	***TRIPLOT***	***QUADPLOT***	
5%	0.000	0.000	0.000	
10%	0.007	0.013	0.036	2 inch
15%	0.084	*	*	
20%	0.136	*	*	
25%	*	*	*	

can influence the efficacy of RDA to capture spatial variability. The results from this portion of the experiment were very promising. The values from table 2 are partial R^2 values range to a high of 56 percent. The largest portions of variation explained by the spatial component were found in the scenarios with the greatest difference in mean DBH.

In pRDA we measured the probability of successfully detecting spatial dependency with permutation F-tests which looked at the significance of the "pure spatial" component. Generally, the results are promising with many scenarios > 80 percent success. Not surprising, an increase in spatial pattern complexity decreases the probability of successful detection; however, the decreases tend to be small. More influential than pattern complexity is difference in mean DBH.

The success of our clustering algorithm depends on how distinct each microsite is and how similar each tree is to others within a single microsite. For example, when we simulate the biplot we are looking at trees from two different distributions. The greater the difference in means and the smaller the CV, the more similar trees are within a microsite and the more dissimilar they are to trees from other microsites. As the mean difference in DBH decreases and/or CV increases the more overlap there is in the distributions of the microsites. This overlap can create false groups and the clustering algorithm may identify these overlapped trees as unique groups (given they are also neighbors) and affect the accuracy of estimates.

Misclassification simulations were done for scenarios where the probability of successful detection was greater than 80 percent. For scenarios where probability of successful detection was greater than 80 percent, the misclassifications of trees ranged from about 0 percent-32 percent. This indicates that when cluster analysis works with high reliability the probability of misclassification is low. The cluster analysis was moderately successful. Sharp decreases in probability of success leave some lack of confidence in the ability of these methods to detect microsites patterns accurately.

LITERATURE CITED

Buford, M.A.; Burkhart, H.E. 1987. Genetic improvement effects on growth and yield of loblolly pine plantations. Forest Science. 33(3): 707-724.

Legendre, P.; Legendre, L.F. 2012. Numerical ecology. Vol. 20. The Netherlands: Elsevier. 989 p.

Magnussen, S.; Kremer, A. 1993. Selection for an optimum tree growth curve. Silvae Genetica. 42: 322-322.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. Updated ed. New York: Wiley and Sons. 520 p.

Tang, S.; Meng, F.R.; Bourque, C.P.A. 2001. Analyzing parameters of growth and yield models for Chinese fir provenances with a linear mixed model approach. Silvae Genetica. 50(3-4): 140-14.

A COLLECTION OF SPROUTING PARAMETERS FOR SIMULATING REGENERATION ESTABLISHMENT IN THE MISSOURI OZARKS

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Abstract—Predicting the effects of silvicultural choices on tree regeneration has traditionally been difficult with the tools currently available to foresters. In an effort to improve this, we have developed a simulation framework based on hypotheses of stand dynamics for several species found in the Missouri Ozarks. This framework includes separate modules for establishment, growth, and mortality. Within the establishment module, empirical parameters are used to stochastically simulate regeneration establishment following a variety of harvest-based silvicultural manipulations. A pre-disturbance inventory is used to account for existing conditions and previously published sprouting parameters define the potential contributions of stump-sprouts to regeneration. The collection of previously published sprouting parameters selected for use and their role in simulating regeneration establishment in the Missouri Ozarks are discussed.

INTRODUCTION

Forest regeneration is a dynamic process involving the establishment, growth, and mortality of trees. Complex interactions between these three components continually shape the regeneration process, yet their outcomes may not manifest until years or decades have passed (Quero and others 2011). Given this timeframe and the crucial role of regeneration to sustainable forestry (Dey 2014), the need for computer simulated projections of regeneration to expedite silvicultural diagnoses and improve the likelihood of achieving desired outcomes at the end of regeneration is clear.

Many existing strategies for regeneration simulation were designed for specific silvicultural systems in specific regions. Recently, there have been dramatic shifts in management objectives in favor of more structurally and compositionally diverse forests with continuous canopy cover (Puetmann and others 2008). Accordingly, many of the previous strategies for simulating regeneration are not readily adaptable to the more structurally diverse forest conditions embodied by current management objectives. Moreover, many existing strategies combine the outcomes of regeneration and recruitment into a single point estimate of certain attributes (density, composition, structure). Combining regeneration and recruitment is biologically sound and likely has little impact for most applied objectives. However, non-process base modeling approaches may incur an opportunity cost of limited ability to simulate novel floristic and management scenarios. In addition, non-process based models are typically incapable of generating output in incremental time steps throughout the regeneration period. Consequently, they lack the ability to support adaptive management.

In an effort to improve upon the current capacity of foresters to simulate regeneration in more structurally diverse conditions, a research project was initiated to develop a simulation framework to examine the impact of overstory density on the establishment, growth, and mortality of reproduction in the Missouri Ozarks (Vickers 2015). This framework allows for output in incremental time steps, which contrasts to models that produce point estimates of structure, stocking and composition only at the end of the regeneration period. In addition, this framework permits evaluating the impact of varying residual overstory density resulting from differing silvicultural regeneration methods on regeneration and thus, is a more robust modeling approach. Under this approach, various silvicultural methods are viewed simply as manipulations that result in varying overstory density that is retained long enough to effect regeneration, and whose affect can vary spatially throughout the stand. The increased generalization offered by this approach greatly increases the breadth of harvest and natural gap based disturbance scenarios that can be examined for their impact on regeneration.

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Forest regeneration is a defining example of secondary succession (Horn 1974). A fundamental concept of secondary succession is the development of and reliance upon regeneration sources disseminated prior to or immediately following disturbance. Therefore, the composition of future forests is largely a function of the composition of previous forests (Egler 1954), though the ultimate identity of the regenerating stand will be influenced by the magnitude and timing of disturbances and their biological impacts on stand development along with other stochastic influences (Gleason 1917). Accordingly, a commonality among many regeneration models is the requirement of some metric of advance reproduction as input data. Weiskittel and others (2011, p157) described this as the "established seedling" approach to regeneration modeling. Our simulation framework largely follows this approach and requires an inventory of certain attributes of advance reproduction and potential sprouting sources prior to a disturbance event. The reliance on disturbance to initiate the regeneration process is consistent with leading hypotheses of forest stand development (Oliver and Larson 1996).

Sprouting is often critical to regeneration success in the oak-hickory dominated forests of the Missouri Ozarks (Johnson and others 2009), and is an important component of regeneration establishment for many species in other regions as well (Bond and Midgley 2003). Thus, several probabilistic models of stump sprouting have been developed to quantify possible contributions of this regeneration source to forest composition following disturbance. The objective of this manuscript is to describe a collection of existing empirical models of stump sprouting and stump clumping (number of sprouts per stump) that were identified for use in the establishment module of a regeneration simulator for the Missouri Ozarks. Establishment is defined herein as reproduction (trees ≤ 5cm d.b.h.) present three years following disturbance.

METHODS

Sprouting parameters were obtained from a review of existing literature and analyses. Existing parameters for both stump sprouting probability and stems per sprouted stump (clumping) were sought for the following species (or species groups) that are commonly found in Missouri Ozark forests: 1) ashes (*Fraxinus americana* L., *F. pennsylvanica* Marsh.), 2) blackgum (*Nyssa sylvatica* Marsh.), 3) black cherry (*Prunus serotina* Ehrh.), 4) dogwood (*Cornus florida* L.), 5) elms (*Ulmus alata* Michx., *U. rubra* Muhl., *U. americana* L.), 6) hickories (*Carya tomentosa* Sarg., *C. glabra* Mill., *C. ovata* (Mill.) K. Koch., *C. texana* Buckley, *C. cordiformis* (Wangenh.) K. Koch.), 7) red maple (*Acer rubrum* L.), 8) red oaks (*Q. rubra* L., *Q. velutina* Lam., *Q. coccinea* Münchh, *Q. marilandica* Münchh.), 9) sassafras (*Sassafras albidum* J. Presl.), 10) shortleaf pine (*Pinus echinata* Mill.), 11) sugar maple (*Acer saccharum* Marsh.), and 12) white oaks (*Q. alba* L., *Q. stellata* Wangenh, *Q. muehlenbergii* Engelm).

Species specific parameters were preferred when available. In cases where species specific parameters were not found, parameters for closely related species were accepted. Otherwise, parameters for a species may have been obtained from publications that reported parameters for a collection of species grouped according to the various publication specific criteria. When multiple publications were available for an individual species, those developed in forest communities that most closely resembled the Missouri Ozarks were preferred. Additionally, attributes such as the strength and range of data, as well as model form and complexity were considered when multiple publications were found for a single species. The potential impact of differences in sprouting parameters due to varying stand ages reported among the respective publication studies was ignored.

In order for these equations to be utilized, individual stems with the potential to sprout must first be identified from a provided inventory. For the purposes of the establishment module, the parent stem (all stems for multi-stemmed parents) must be identified as a tree to be removed in the pre-disturbance inventory. For multi-stemmed parents, attributes of the largest stem are used to apply the probabilistic sprouting parameters. These probabilities are used in a binomial random number generator to stochastically determine the sprouting outcome (success or failure) of each inventoried stem that has the potential to sprout. Although some species produce basal sprouts under an intact parent stem, this regeneration source along with root sprouts (suckers) were not directly considered. However, it is assumed that root sprouts are indirectly accounted for in the establishment estimates for reproduction that is not from obvious sprout origin (Vickers 2015). The clumping parameters are used in a Poisson random number generator to stochastically determine the number of new stems to be established per sprouted stump.

RESULTS AND DISCUSSION

Existing models for estimating the probability of establishment via stump sprouting were utilized for the species groups included in the establishment module. The models used for estimating stump-sprouting, along with their origin, are provided in table 1. Petrice and Haack (2011) reported approximately 96 percent sprouting for ash (*F. americana*, *F. pennsylvanica*, *F. nigra* Marsh.) stumps 1 year after midspring harvest in Michigan. Keyser and Zarnoch (2014) reported that breast height diameter influenced sprouting of blackgum one year following harvest in the Southern Appalachians. Wendel (1975) reported that 100 percent

Table 1—Existing stump sprouting parameters identified for use in the establishment module of a regeneration simulator for the Missouri Ozarks

Species Group		Sprouting Probability	Source	Source Locale
ashes		0.96	Petrice & Haack 2011	MI
blackgum		$\{1+e^{-(5.5798-0.2011 \cdot X_1)}\}^{-1}$	Keyser & Zarnoch 2014	S. Appalachians
black cherry		0.98	Wendel 1975	WV
dogwood		0.88	Keyser & Zarnoch 2014	S. Appalachians
elms		$\{1+e^{-(1.6323-0.0608 \cdot X_1)}\}^{-1}$	Olson, M.G. unpublished	MO
hickories		0.77	Keyser & Zarnoch 2014	S. Appalachans
red maple		0.95	Keyser & Zarnoch 2014	S. Appalachians
red oaks	black oak	$\{1+e^{-(-0.7523-0.2691[1.2703(0.3937 \cdot X_1)^{1.0105}]+0.0615[3.2808 \cdot X_2])}\}^{-1}$	Dey 1991	MO Ozarks
	blackjack oak	$\{1+e^{-(2.6481-0.3005[1.3322(0.3937 \cdot X_1)^{0.9955}])}\}^{-1}$	Dey 1991	MO Ozarks
	n red oak	$\{1+e^{-(-7.4793-0.0005[(0.3937 \cdot X_1)X_3]+3.1679 \cdot \ln X_2)}\}^{-1}$	Weigel & Peng 2002	IN
	scarlet oak	$\{1+e^{-(5.7710-0.3513[1.2154(0.3937 \cdot X_1)^{0.9798}])}\}^{-1}$	Dey 1991	MO Ozarks
sassafras		0.91	Keyser & Zarnoch 2014	S. Appalachians
shortleaf pine		0.67	Clabo 2014	TN
sugar maple		0.81	MacDonald & Powell 1983	NB, Canada
white oaks	post oak	$\{1+e^{-(2.6481-0.3005[1.3322(0.3937 \cdot X_1)^{0.9955}])}\}^{-1}$	Dey 1991	MO Ozarks
	white oak	$\{1+e^{-(1.7378-0.5476[1.5443(0.3937 \cdot X_1)^{0.9532}]-0.0546 \cdot X_3+0.0033[X_3(1.5443(0.3937 \cdot X_1)^{0.9532})]+0.0705[3.2808 \cdot X_2])}\}^{-1}$	Dey 1991	MO Ozarks
other species		0.89	Keyser & Zarnoch 2014	S. Appalachians

Where: X_1 = diameter at breast height (cm), X_2 = site index (m), X_3 = tree age (yrs). Values for black cherry were modified slightly from those reported (100%, Wendel 1975). The sprouting equations developed by Dey (1991) were originally fit using stump diameter rather than diameter at breast height. Because stump diameter is not commonly inventoried, these equations were modified to include a diameter at breast height to stump diameter conversion (Dey 1991). The parameters designated for the other species group were used for any sprouting species not included in the previous groups.

of black cherry harvested in a study in West Virginia produced a sprout within 1 year of harvesting. This value was modified slightly to ninety-eight percent to foster stochastic establishment in the regeneration simulator. Eighty-eight percent of dogwood stumps sprouted 1 year following harvest in the Southern Appalachians (Keyser and Zarnoch 2014). The sprouting probability of elm (*U. americana*) was influenced by breast height diameter one year post harvest in Missouri (unpublished data: Olson, M.G., Resource Scientist-Silviculturist, Missouri Department of Conservation). Seventy-seven percent of hickories sprouted 1 year following harvest in a Southern Appalachian study (Keyser and Zarnoch 2014). Ninety-five percent of red maple sprouted in the same study (Keyser and Zarnoch 2014).

Dey (1991) reported that the sprouting probabilities for black, blackjack, scarlet, post, and white oaks in the Missouri Ozarks were influenced by size (table 1). The sprouting equations developed by Dey (1991) were originally fit using stump diameter rather than breast height diameter. Because stump diameter is not commonly inventoried, these equations were modified to include the breast height to stump height conversions that were also developed by Dey (1991). Dey (1991) reported that the sprouting probability of black oak and white oak were also influenced by site index. The sprouting probability of white oak was also influenced by tree age (Dey 1991). Weigel and Peng (2002) reported that the sprouting probability of red oaks (*Q. rubra*, *Q. coccinea*) were influenced by breast height diameter, site index, and tree age.

Keyser and Zarnoch (2014) reported the sprouting probability of sassafras (ninety-one percent) one year post harvest as part of a species group that included shade tolerant species (table 1). Sixty-seven percent of clipped shortleaf pine seedlings sprouted in a study in Tennessee reported by Clabo (2014). The reported sprouting probabilities for shortleaf pine seedlings (Clabo 2014) were not used within the simulator for trees with a breast height diameter > 5cm. MacDonald and Powell (1983) reported sprouting probabilities for sugar maple one year following harvest in New Brunswick. The probability used for sugar maple (eighty-one percent) was an average calculated from the results reported by MacDonald and Powell (1983, table 1 column 3, rows 1-4). The sprouting probability (eighty-nine percent) designated for the other species group were used for any sprouting species not included in the previous groups and was the average of the tolerant and intolerant species group probabilities reported by Keyser and Zarnoch one year post harvest (2014).

Because a stump that successfully sprouts often produces more than one new stem per stump, existing models of stems per sprouted stump were used to account for multi-stemmed stump-clumps. The models used for estimating the number of stems per sprouted stump, along with their origin, are provided in table 2. Kays and Canham (1991) reported a mean of 5.2 stems per sprouted ash stump 3 years following harvesting in New York. Atwood and others (2009) reported a mean of 4.3 stems per sprouted stump 9 years following harvesting in the Southern Appalachians for a group of predominately midstory species that included dogwood and sassafras among others. This value was applied to the blackgum and other species group as well. An average of 4.3 stems per sprouted stump was also reported for black cherry 3 years following harvesting in New York (Kays and Canham 1991). In Missouri, 7.3 new stems per sprouted stump were reported for elm (*U. americana*) 3 years following harvest (unpublished data: Olson, M.G., Resource Scientist-Silviculturist, Missouri Department of Conservation). Atwood and others (2009) reported a mean of 3.2 stems per sprouted stump 9 years post harvest in the Southern Appalachians for a species group that included hickories and white oak. This value was used only for hickories. Three years following harvest in New York, red maple averaged 6.2 stems per sprouted stump (Kays and Canham 1991). Clabo (2014) reported an average of 2.1 stems per sprouted stump for clipped shortleaf pine seedlings in Tennessee. An average of 10 sugar maple stems per sprouted stump was reported 1 year following harvest in a study in New Brunswick (MacDonald and Powell 1983).

Dey and Jensen (2002) reported that the number of stems per sprouted stump one year following harvest was influenced by stump diameter and tree age for black, scarlet, and white oaks in the Missouri Ozarks (table 2). The parameters identified for blackjack oak and post oak were taken from the equations reported by Dey and Jensen (2002) for black oak and white oak respectively. Johnson (1975) reported that the number of northern red oak stems per sprouted stump in Wisconsin, Michigan, and Iowa was influenced by breast height diameter and stand age.

Ecological classification systems have been suggested for regeneration research (Dey and others 2009, Kabrick and others 2008). Accordingly, an ecological classification system was used to delineate site differences in the regeneration simulator framework (Vickers 2015). Consequently, it was assumed that site index will often be unknown. For sprouting equations that required site index (black oak, northern red oak, white oak), empirical mean and standard deviation site index values for each site class (exposed backslopes: 21.0 ± 1.3m, protected backslopes: 22.0 ± 1.1m; black oak, base age 50) are used in a Gaussian random number generator to stochastically assign a site index value for a plot (Vickers 2015).

Table 2—Existing stump-clump parameters identified for use in the establishment module of a regeneration simulator for the Missouri Ozarks

Species Group		Sprouts per stump	Source	Source Locale
ashes		5.2	Kays & Canham 1991	NY
blackgum		4.3	Atwood et al. 2009	S. Appalachians
black cherry		4.3	Kays & Canham 1991	NY
dogwood		4.3	Atwood et al. 2009	S. Appalachians
elms		7.3	Olson, M.G. unpublished	MO
hickories		3.2	Atwood et al. 2009	S. Appalachians
red maple		6.2	Kays & Canham 1991	NY
red oaks	black oak	$21.028+(-0.0310[1.2703(0.3937 \cdot X_1)^{1.0105}])+(-0.1537 \cdot X_2)$	Dey & Jensen 2002	MO Ozarks
	scarlet oak	$20.9773+(-0.0310[1.2154(0.3937 \cdot X_1)^{0.9798}])+(-0.1537 \cdot X_2)$	Dey & Jensen 2002	MO Ozarks
	blackjack oak	$21.028+(-0.0310[1.3322(0.3937 \cdot X_1)^{0.9955}])+(-0.1537 \cdot X_2)$	Dey & Jensen 2002	MO Ozarks
	n red oak	$3.7+8.82([0.3937 \cdot X_1]/X_3^2)$	Johnson 1975	WI,MI,IA
sassafras		4.3	Atwood et al. 2009	S. Appalachians
shortleaf pine		2.1	Clabo 2014	TN
sugar maple		10	MacDonald & Powell 1983	NB, Canada
white oaks	white oak	$17.0967+(-0.0310[1.5443(0.3937 \cdot X_1)^{0.9532}])+(-0.1537 \cdot X_2)$	Dey & Jensen 2002	MO Ozarks
	post oak	$17.0967+(-0.0310[1.3322(0.3937 \cdot X_1)^{0.9955}])+(-0.1537 \cdot X_2)$	Dey & Jensen 2002	MO Ozarks
other species		4.3	Atwood et al. 2009	S. Appalachians

Where: X_1 = diameter at breast height (cm), X_2 = tree age (yrs), and X_3 = post disturbance stand age (yrs). A post disturbance stand age of 3 is used for stem establishment in the simulator. The equations developed by Dey and Jensen (2002) were originally fit using stump diameter rather than diameter at breast height. Because stump diameter is not commonly inventoried, these equations were modified to include a diameter at breast height to stump diameter conversion (Dey 1991). The parameters designated for the other species group were used for any sprouting species not included in the previous groups.

Some sprouting and clumping equations required tree age, but tree age is often not included in forest inventories. Thus, it was assumed that tree age will often be unknown and published equations for converting diameter at breast height to tree age (Lowenstein and others 2000) were used when required for stump sprouting probabilities (northern red oak, white oak) and clumping (black oak, scarlet oak, blackjack oak, white oak, post oak). Due to a positive interaction between tree age and diameter in the equation used for the sprouting probability of white oak, increasing tree age substantially improved sprouting probability with increasing diameter once age exceeded certain thresholds. In contrast, Johnson and others (2009) suggest that it is unlikely that large diameter oaks will sprout if harvested. Thus, an algorithm was required to ensure that spurious sprouting probabilities were not produced as a result of age estimates that fell outside the range of the fitted dataset. Similarly, an algorithm was required to ensure that the clumping equations that required tree age did not produce spurious estimates when age estimates fell outside the range of the original dataset.

Most of the available models of stump sprouting probabilities were developed from data following clearcut harvesting. Foresters have long recognized differences in life history traits among species and the influence of overstory density and crown cover on regeneration dynamics. Despite this longstanding recognition, relatively little progress has been made toward simulating the impact of varied overstory conditions on the development of reproduction, particularly in naturally regenerated mixed species stands. Evidence of reductions in stump sprouting probabilities with increasing residual overstory density in oak-dominated forests has been reported 9-11 years after harvesting (Atwood and others 2009), but a separate study conducted 1-3 years after harvesting did not report reductions in stump sprouting probabilities with increasing residual overstory density (Keyser and Zarnoch 2014). Keyser and Zarnoch (2014) suggested that this discrepancy could be due to accumulated mortality in the older stands analyzed by Atwood and others (2009). This suggestion is supported by the report of Dey and others (2008), that increasing residual overstory density lowered survival and growth of oak stump-sprouts 10 years after harvesting, but the proportion of sprouting stumps in the first year after harvest was not significantly affected (Dey and Jensen 2002). Keyser and Zarnoch (2014) also found that the heights of sprouts were reduced with increasing residual overstory density. In light of these reports, it is apparent that residual overstory density influences stump sprouting dynamics in oak dominated forests. However, this influence appears to be primarily a reduction in growth rates which, in turn, reduces stump-sprout survival during stem exclusion. This hypothesis is followed in the regeneration simulator. Dey and Jensen (2002) also reported that the number of oak sprouts per stump was significantly affected by stump diameter, but not by residual overstory density. Consequently, models of stump sprouting and clumping probabilities following clearcutting may provide reasonable estimates of third-year regeneration establishment following a variety of harvest-based silvicultural manipulations. However, the prolonged influence of residual overstory density on sapling survival which, in turn, influences the abundance of potential sprouting sources should be considered. Additional research into the influence of residual overstory density on stump sprouting dynamics is warranted but beyond the scope of this effort.

Many studies suggest that the sprouting probability of small diameter trees is high after harvesting, but the minimum diameter included in sprouting models vary (e.g., Johnson and others 2009). The probability of advance reproduction being damaged or top-killed during a disturbance has not been well defined empirically. Additional research to provide empirical parameters for the probability of damage/topkill and subsequent sprouting probabilities of advance reproduction is warranted.

CONCLUSION

Regeneration is a dynamic process involving the establishment, growth, and mortality of individual trees and their neighbors. The capability to stochastically simulate reproduction establishment across a gradient of residual overstory density allows foresters to examine the potential outcomes of proposed regeneration treatments. The collection of sprouting parameters identified in this report provide an empirical foundation for estimating the potential contributions of this, often critical, source of reproduction to post disturbance composition in Missouri Ozark forests. Areas in need of additional work have been suggested. It is possible that alternative or, perhaps, more appropriate parameters than those included in this report currently exist or may be developed with future research. Periodic literature reviews are suggested to ensure the best available science is incorporated in future regeneration simulations.

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LITERATURE CITED

Atwood, C.J.; Fox, T.R.; Loftis, D.L. 2009. Effects of alternative silviculture on stump sprouting in the southern Appalachians. Forest Ecology and Management. 257: 1305-1313.

Bond, W.J.; Midgley, J.J. 2003. The evolutionary ecology of sprouting in woody plants. International Journal of Plant Science. 164(3 Suppl.): S103-S114.

Clabo, D.C. 2014. Shortleaf pine sprout production capability in response to disturbance. Knoxville, TN: University of Tennessee. 76 p. M.S. thesis.

Dey, D.C. 1991. A comprehensive Ozark regenerator. Columbia, MO: University of Missouri. 283 p. Ph.D. dissertation.

Dey, D.C. 2014. Sustaining oak forests in Eastern North America: regeneration and recruitment, the pillars of sustainability. Forest Science. 60(5): 926-942.

Dey, D.C.; Jensen, R.G. 2002. Stump sprouting potential of oaks in Missouri Ozark forests managed by even- and uneven-aged silviculture. In: Shifley, S. R.; Kabrick, J. M., eds. Proceedings of the 2nd Missouri Ozark forest ecosystem project symposium: post-treatment results of the landscape experiment. Gen. Tech. Rep. NC-227. St. Paul, MN: U.S. Dept. of Agriculture Forest Service, North Central Forest Experiment Station: 102-113

Dey, D.C.; Jensen, R.G.; Wallendorf, M.J. 2008. Single-tree harvesting reduces survival and growth of oak stump sprouts in the Missouri Ozark Highlands. In: Jacobs, D.F; Michler, C.H., eds. Proceedings of the 16th central hardwood forest conference. Gen. Tech. Rep. NRS-P-24. St. Paul, MN: U.S. Dept. of Agriculture Forest Service, North Central Forest Experiment Station: 26-37.

Dey, D.C.; Spetich, M.A.; Weigel, D.R. [and others]. 2009. A suggested approach for design of oak (*Quercus* L.) regeneration research considering regional differences. New Forests. 37: 123-135.

Egler, F.E., 1954. Vegetation science concepts. I. initial floristic composition – a factor in old-field vegetation development. Vegetatio. 4: 412-417.

Gleason, H.A. 1917. The structure and development of the plant association. Bulletin of the Torrey Botanical Society. 43: 463-481.

Horn, H.S. 1974. The ecology of secondary succession. Annual Review of Ecological Systems. 5: 25-37.

Johnson, P.S. 1975. Growth and structural development of red oak sprout clumps. Forest Science. 21: 413-418.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2009. The ecology and silviculture of oaks. New York: CABI Publishing. 600 p.

Kabrick, J.M.; Zenner, E.K.; Dey, D.C. [and others]. 2008. Using ecological land types to examine landscape-scale oak regeneration dynamics. Forest Ecology and Management. 255: 3051-3062.

Kays, J.S.; Canham, C.D. 1991. Effects of time and frequency of cutting on hardwood root reserves and sprout growth. Forest Science. 37: 524-539.

Keyser, T.L.; Zarnoch, S.J. 2014. Stump sprout dynamics in response to reductions in stand density for nine upland hardwood species in the southern Appalachian Mountains. Forest Ecology and Management. 319: 29-35.

Loewnstein, E.F.; Johnson, P.S.; Garrett, H.E. 2000. Age and diameter structure of a managed uneven-aged oak forest. Canadian Journal of Forest Research. 30: 1060-1070.

MacDonald, J.E.; Powell, G.R. 1983. Relationships between stump sprouting and parent-tree diameter in sugar maple in the 1st year following clear-cutting. Canadian Journal of Forest Research. 13: 390-394.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. 2nd ed. New York: John Wiley & Sons, Inc. 520p.

Petrice, T.R.; Haack, R.A. 2011. Effects of cutting time, stump height, and herbicide application on ash (Fraxinus spp.) stump sprouting and colonization by emerald ash borer (Agrilus planipennis). Northern Journal of Applied Forestry. 28(2): 79-83.

Puettmann, K.J.; Coates, K.D.; Messier, C.C. 2008. A critique of silviculture: managing for complexity. Washington, DC: Island Press. 206 p.

Quero, J.L., Herrero, A., Zamora, R. 2011. Linking stochasticity to determinism of woody plant recruitment in a mosaic landscape: a spatially explicit approach. Basic and Applied Ecology. 12: 161-171.

Vickers, L.A. 2015. Modeling the regeneration and early stand dynamics of Missouri Ozark forests. Columbia, MO: University of Missouri, Columbia, Missouri. 271 p. Ph.D. dissertation.

Weigel, D.R.; Peng, C.-Y.J. 2002. Predicting stump sprouting and competitive success of five oak species in southern Indiana. Canadian Journal of Forest Research. 32: 703-712.

Weiskittel, A.R.; Hann, D.W.; Kershaw, J.A. Jr.; Vanclay, J.K. 2011. Forest growth and yield modeling. New York: Wiley-Blackwell. 415p.

Wendel, G.W. 1975. Stump sprout growth and quality of several Appalachian hardwood species after clearcutting. Res. Pap. NE-329. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 9 p.

SIMULATING THE EFFECTS OF SITE INDEX VARIATION WITHIN LOBLOLLY PINE PLANTATIONS USING AN INDIVIDUAL TREE GROWTH AND YIELD MODEL

Ralph L. Amateis and Harold E. Burkhart[1]

Abstract—Site index is the most common metric of site productivity in loblolly pine plantations. Generally applied as a constant for a particular stand, it provides an overall measure of a site's ability to grow trees. It is well known, however, that even the most uniform stands can have considerable variation in site index due to soil factors that influence microsite, variation in genetics from tree to tree, or the uneven application of silvicultural treatments. To better account for such variability, input options to the PTAEDA (version 4.1), an individual tree growth and yield model, were expanded to allow groups of trees at time of planting to be assigned to different site index classes and the variability within those classes to be specified by the user in different ways. This capability allows comparison of alternative methods of introducing site variability into individual tree simulators such as PTAEDA. Preliminary results suggest that the individual tree distance dependent growth and yield model architecture is a useful platform for defining site productivity patterns within stands and evaluating the impact of those patterns on growth and yield.

INTRODUCTION

Characterizing the productive capacity of a site to grow trees is one of the first and most important assessments land managers must make. Site index, defined as the height of the dominant portion of the stand at a standardized (index) age, has long been accepted as the most direct, common measure of overall productivity for loblolly pine plantations. It has the advantages of being quantifiable, easy to measure and relatively insensitive to changes in stand density (Clutter and others 1983; Burkhart and Tomé 2012). While usually treated as a constant for a given stand, site index is actually an average height whose value depends on the sampling scheme used to obtain an estimated dominant stand height (Smith and Burkhart 1983) as well as the definition of dominant stand height itself (Sharma and others 2002). Further, it can vary due to changes in edaphic and climatic factors over time and the influence of management treatments.

Although the term implies a measure of productivity, it is only meaningful for the particular trees growing on that site. That is, site index is really a composite measure of a site's productive potential to grow a specific population of trees. For example, the same acre of land might be excellent for growing loblolly pine with a high loblolly pine site index value but poor for growing yellow poplar with a correspondingly low value for yellow poplar.

Ameliorative treatments that alter the productive potential of a site must also be weighed when assigning a site index value to a stand of trees. Draining and/or bedding of inherently wet sites and adding phosphorus to phosphorus deficient sites are examples of early treatments that can have a large impact on the site index of loblolly pine (Allen 1987; Allen and others 1990). Mechanical site preparation prior to stand establishment can also affect site index. For a cutover site-prepared loblolly pine plantation established in the North Carolina Piedmont following clear cutting, Fox and others (1989) found that piling logging slash into windrows following harvest resulted in a 23 percent reduction in volume yield compared to an adjacent site that received a broadcast burn site preparation treatment following harvest. On the same tract, they found the site index between the windrows to be 11 feet less than the broadcast burn area. Within the windrowed area, the height of the trees decreased rapidly as distance from the windrow increased so that the average height of windrow-adjacent trees was 10 feet taller than windrow-interior trees.

Stand conditions such as those just described produce variations in site index that are visually obvious from the height growth of the trees on the site. However, even the most uniform sites have microsite and genetic variation across the landscape and from tree to tree that affect growth. These sources of variation and their interactions

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may not be visually obvious but can have an impact on overall stand productivity and product distributions at harvest. Growth and yield models that can account for variation in site productivity due to microsite and genetic influences and unevenness in treatment applications should be useful for assessing the impact of these important factors on the productivity of loblolly pine.

The loblolly pine individual tree distance dependent (IDD) model PTAEDA (Daniels and Burkhart 1975) for old field loblolly pine plantations has a general model structure that appears suitable for studying these factors. In PTAEDA juvenile mortality is assigned at random, trees are set out on the landscape in an x-y grid and the initial heights and diameters at time of initiation of intraspecific competition are obtained from the Weibull distribution. The core individual tree growth equations are comprised of a potential height increment equation based on a site index equation and a potential dbh increment equation defined by the open grown diameter increment equation of Daniels and Burkhart (1975). These potential equations are then modified by a competition index (Hegyi 1974; Daniels 1976) that reflects the intraspecific competitive pressure exerted by neighboring trees. Random components obtained from the fitted growth equations are then added to the height and dbh increment to account for the variation in tree growth from year to year.

While the general model structure of PTAEDA has remained foundational since its inception, it has shown itself to be quite flexible and able to accommodate additional enhancements and features incorporated through the years. Major enhancements have included re-estimation of core equations using a large region-wide set of data from cutover site-prepared plantations (Burkhart and others 1987) (version 2.0), addition of mid-rotation fertilization response functions (Hynynen and others 1998), juvenile growth equations (Westfall and others 2004) reflecting the effects of alternative site preparation treatments (version 2.1), addition of individual-tree thinning capabilities, diameter distribution and stem quality input capabilities by diameter class (version 3.1), linkage with the Stand Visualization System developed by the Forest Service, additional merchandizing and economic evaluation options (version 4.0), and the capability of modeling so called “flex” stands comprised of two populations (version 4.1) (Amateis and Burkhart 2012). Table 1 summarizes much of the evolution of PTAEDA through the years.

The purpose of this paper is to show how PTAEDA 4.1 has been adapted to account for variation in site index due to factors that are not constant across the landscape such as microsite influences, variation in genetic potential from tree to tree, and the uneven application of silvicultural treatments.

METHODS

In order for PTAEDA to be useful for studying the impacts of variation in site index, additions to the input options were made to allow the user more control over how initial tree characteristics are defined in the simulation plot. First, a graphical tool was installed to allow definition of groups of trees that are determined by mean site index and coefficient of variation (CV) about that mean under the assumption of a normal distribution. Groups can be circumscribed in the plot as rectangles, ovals or drawn freeform. The number of groups must be at least one, there must be at least two trees in every group, and all trees must be assigned to a group. This relaxes the usual assumption that site index is a constant stand attribute applying to all the trees. Instead, site index is a tree variable with value for any given tree depending on the mean and the CV of the group in which it resides:

$$S = \bar{S} + \bar{S} * CV/100 * st_nor_dev$$

where S is the assigned site index for each tree of a group, $\bar{S}$ and CV are the mean and CV for the group, and *st_nor_dev* is a standard normal deviate. When this is done, each tree in the group has its own site index value, or growth potential, that will be somewhat more or less than the mean for the group. The choice of $\bar{S}$ and CV reflect the overall productivity and the variation of that productivity for each group.

Each group of the stand is advanced to age 8, the end of the juvenile period, and the mortality by group, is assigned at random using the stand-level survival equations in PTAEDA. The minimum and average dbh of each group is determined by $\bar{S}$ and the trees surviving for the group. Initial dbh values are assigned to each live tree assuming a Weibull distribution. Initial height and crown ratio are assigned as in Burkhart and others (1987). After assigning all initial tree sizes, a competition index is calculated for each tree based on the size of the tree and its qualifying neighbors who may reside within or outside the subject tree’s group. Information for every tree is then passed to the stand growth algorithm for projection. Mid-rotation management treatments are handled in the usual way. Output options that include exporting the list of tree attributes along with group characteristics have been added to facilitate post-processing of simulation results.

CASE STUDY

The unreplicated case study of Fox and others (1989) was used to test the usefulness of the site index modifications to the PTAEDA simulator. The study consisted of a three acre loblolly pine stand in the North

Table 1—Overview of the evolution of the major data, models and software used in the PTAEDA simulator from its inception to version 4.1

Characteristic	PTAEDA (1975)	PTAEDA2 (1987)	PTAEDA4.1 (2015)
		Data	
Plot	Old field (Burkhart and others 1972)	Cutover site-prepared (Region-wide)	Cutover site-prepared (Region-wide)
Tree Growth	Limited mapped studies	Cutover site-prepared (Region-wide)	Cutover site-prepared (Region-wide)
Potential dbh growth	81 open-grown	81 open-grown	81 open-grown
		Models	
Juvenile mortality	Random	Random	Random
End of juvenile period	CCF = 100	Age 8	Variable or age 8
Site Index	Anamorphic (Burkhart and others 1972)	Polymorphic (Amateis and Burkhart 1985)	Polymorphic (Diéguez-Aranda and others 2006)
Crown Ratio	Linear	Non-linear (Dyer and Burkhart 1987)	Non-linear (Dyer and Burkhart 1987)
Potential Dbh Increment	Linear	Linear	Linear
Competition Index	Weighted Size Ratio (Hegyi 1974)	Weighted Size Ratio (Hegyi 1974)	Weighted Size Ratio (Hegyi 1974)
Height Increment Adjustment	Non-linear	Non-linear	Non-linear
Dbh Increment Adjustment	Non-linear	Non-linear (optional hardwood competition)	Non-linear (optional hardwood competition)
Mortality	Non-linear	Non-linear	Non-linear
Simulated Trees	Fixed at 100	Variable between 25 and 400	Fixed at 625
Planting Patterns	Regular	Regular or Irregular	Regular or Irregular
Silvicultural treatments	Thinning, fertilization	Thinning, fertilization, pruning	Thinning, fertilization, pruning, site preparation
		Software	
Implementation	Mainframe (Fortran)	PC DOS (Fortran)	PC Windows (C++)
Input	Stand variables	Stand variables	Stand, diameter distribution variables
Output	Stand tables	Stand tables, tree list	Stand tables, tree structure, SVS visualization, financial NPV

Carolina Piedmont that was harvested in the fall of 1953. In early 1954 the logging slash was sheared and piled into parallel windrows about 200 feet apart. The windrows were neither burned nor planted leaving about 11 percent of the area unutilized. The area between windrows was planted on 6 feet x 6 feet spacing and grown to age 25 where a mid-rotation thinning removed 1204 cubic feet per acre volume inside bark leaving a standing residual volume of 2783 cubic feet per acre. The residual stand was grown to age 31 when plots were established across the non-windrowed portion of the stand. Approximately twenty-five feet out from the edge of the windrows (about 4 rows) was considered to be "windrow-adjacent" and the center portion between windrows was "windrow-interior". Height and dbh measurements were collected and the stand was clearcut. Plot summaries were compiled and results reported (Fox and others 1989). In the windrow-interior portion of the stand heights were shorter than the windrow-adjacent areas leading to an estimated site index of 56 feet for the windrow-interior areas and 66 feet for the windrow-adjacent areas. This "windrow effect" resulted in a correspondingly higher total volume per acre production in the windrow-adjacent areas compared to the windrow-interior areas.

To test whether PTAEDA could reproduce the growth of this stand, conditions at time of stand establishment were inputted to the PTAEDA simulator. Three groups were defined using the new site productivity pattern capability: (1) an unutilized windrowed area, (2) a windrow-adjacent area, and, (3) a windrow-interior area. (fig. 1). From establishment records and the Fox and others (1989) data, the site index and CV for the windrow-adjacent group were set at 66 feet and 12, respectively, and 56 feet and 10, respectively, for the windrow-interior group. The differentiation in site index between the two groups is apparent by age 8 at the end of the juvenile period (fig. 2). The stand was projected to age 25 where the mid-rotation thinning treatment was applied. Based on assumptions that the thinning was a low thinning with removals of selected larger trees that were unsuitable for a final sawtimber harvest, the basal area of the stand was reduced to 95 square feet per acre resulting in 1265 cubic feet per acre volume inside bark removed. The residual stand was projected to final harvest at age 31.

For comparative purposes, a second simulation was conducted exactly as the first ascribing one overall site index to the stand. Thus, the windrow-adjacent and windrow-interior groups were combined into one group and an area-based weighted average site index of 60 feet with CV set to zero was used. Table 2 compares the volume estimation from the two simulation results against the observed plot data from Fox and others (1989). The results of this comparison suggest that for this case, grouping the trees according to distance from

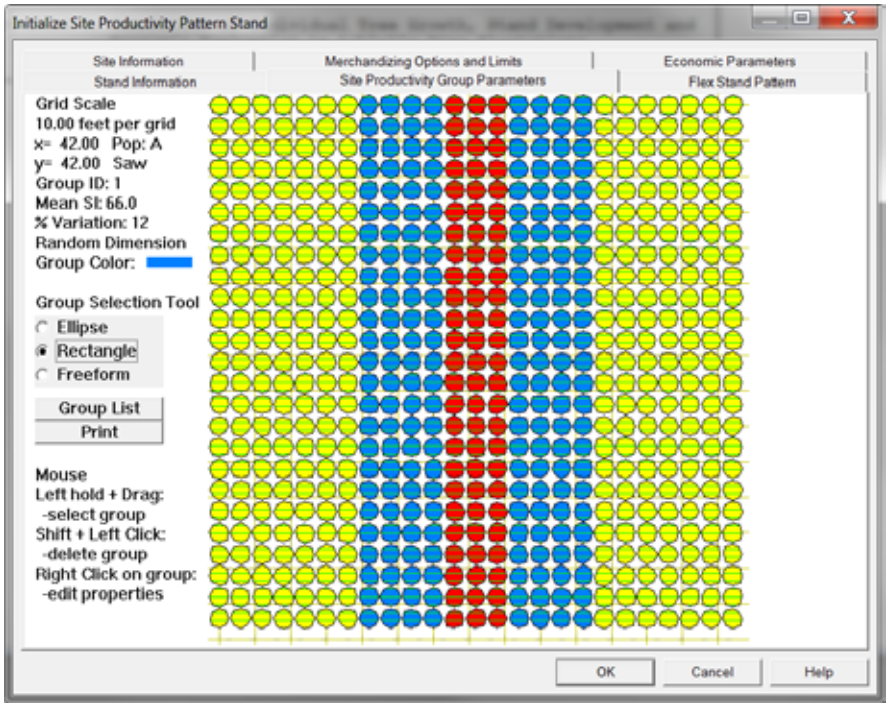


Figure 1—Example site productivity pattern definition comprised of three groups consisting of an unutilized windrow in the center (red) flanked by a windrow-adjacent group of trees (blue with mean site index of 66 feet and CV of 12) and a windrow-interior group of trees (yellow with mean site index of 56 feet and CV of 10).

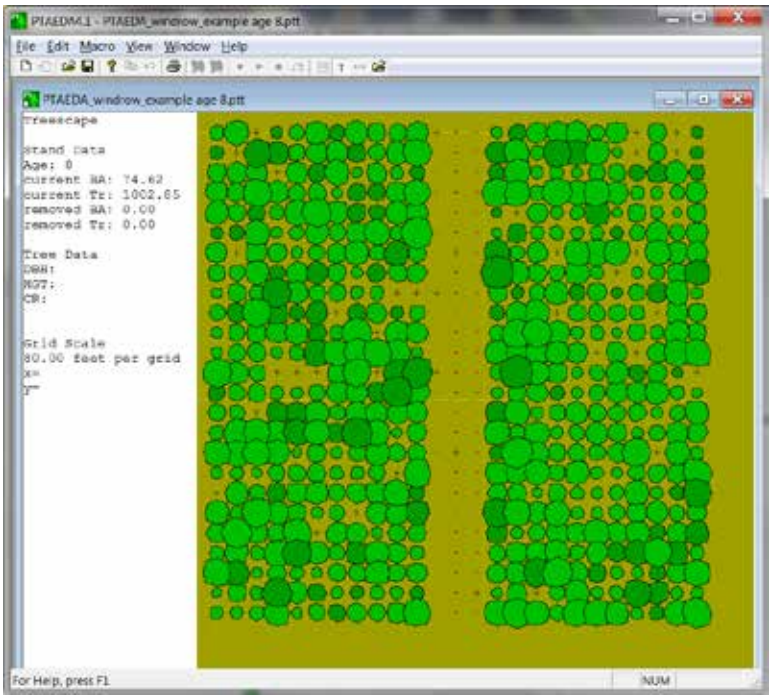


Figure 2—Example stand shown at the end of the juvenile growth period at the initiation of intraspecific competition (age 8).

Table 2—Observed total volume inside bark (cubic feet per acre) through 31 years of a windrowed loblolly pine plantation in the Piedmont of North Carolina compared to two PTAEDA simulations, one using the site productivity pattern capability with site index and coefficient of variation specified by group according to distance from the windrow (PTAEDA Grouped) and a second simulation where no grouping of trees occurs and only a mean site index with no variation is defined (PTAEDA Not Grouped)

Data	Standing Age 31	Total Harvested
Observed	2783	3987
PTAEDA Grouped	2730	3995
PTAEDA Not Grouped	2385	3487

the windrow and ascribing different site index values for each group rather than assigning one overall site index value to the stand will give more realistic yield predictions.

DISCUSSION

As noted in Fox and others (1989) the effect of windrowing on site index is actually a gradient with trees closest to the windrow being the tallest and gradually decreasing in height as distance from the windrow increases approaching some lower plateau toward the windrow-interior area. For simplicity, the area between the windrow was divided into two groups, the boundaries of which were set somewhat arbitrarily. But with the IDD modeling system of PTAEDA, the analysis could be expanded. For example, if data were available, each row could be assigned to a separate group based on distance from the windrow and each group assigned a mean site index and CV. This would expand the number of groups and could account more completely for the height gradient seen in the transect data collected on the site. This level of detail should improve estimates of yield for various portions of the stand and for the stand as a whole.

The case study examined here shows how shearing and piling has impacted the microsite variability within a stand. Other silvicultural treatments such as bedding, fertilization and weed control can also affect microsite variation especially when applied unevenly. If patterns of variation can be identified from soil maps, historical records, ground-based measurements or remotely sensed imaging, PTAEDA should be a useful tool for modeling the growth and development of such stands.

Variation due to planting stock can also affect stand growth. PTAEDA could be used to model two identical sites where one has been established with an elite varietal with a higher exhibited site index value and lower CV than the other site established with open pollen stock. From data, or perhaps assumptions about the effects of genetics on the mean and variation of site index and by holding all other site, stand and management treatment variables constant, it should be possible to simulate the impact of different genetic stock on stand growth using PTAEDA.

By relaxing the usual assumption that site index is a constant for a particular stand and allowing the user to specify groups of trees with a mean site index and associated CV, the IDD modeling system of PTAEDA should be helpful for evaluating the effects of site and tree characteristics that impact stand development. The system is very flexible and able to simulate a wide variety of site, stand and treatment conditions given alternative assumptions about how such conditions affect site index making it ideal for “what if” scenarios.

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LITERATURE CITED

Allen, H.L. 1987. Forest fertilizers: nutrient amendment, stand productivity, and environmental impact. Journal of Forestry. 85: 37-46.

Allen, H.L.; Dougherty, P.M.; Campbell, R.G. 1990. Manipulation of water and nutrients – practice and opportunity in southern U.S. pine forests. Forest Ecology and Management. 30: 437-453.

Amateis, R.L.; Burkhart, H.E. 1985. Site index curves for loblolly pine plantations on cutover site-prepared lands. Southern Journal of Applied Forestry. 9: 166-169.

Amateis, Ralph L.; Burkhart, Harold E. 2012. Extending the capabilities of an individual tree growth simulator to model non-traditional loblolly pine plantation systems for multiple products. In: Butnor, J.R., ed. 2012. Proceedings of the 16th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 7-9.

Burkhart, H.E.; Parker, R.C.; Strub, M.R.; Oderwald, R.G. 1972. Yields of old-field loblolly pine plantations. FWS-3-72. Blacksburg, VA: Virginia Polytechnic Institute and State University, Division of Forestry and Wildlife Resources. 51 p.

Burkhart, H.E.; Tomé, M. 2012. Modeling Forest Trees and Stands. New York: Springer Science and Business Media. 457 p.

Burkhart, H.E.; Farrar, K.D.; Amateis, R.L.; Daniels, R.F. 1987. Simulation of individual tree growth and stand development in loblolly pine plantations on cutover, site-prepared areas. FWS-1-87. Blacksburg, VA: Virginia Polytechnic Institute and State University, Division of Forestry and Wildlife Resources. 47 p.

Clutter, J.L.; Fortson, J.C.; Pienaar, L.V. [and others]. 1983. Timber management: A quantitative approach. New York: John Wiley & Sons, Inc. 333 p.

Daniels, R.F. 1976. Simple indices and their correlation with annual loblolly pine tree growth. Forest Science. 22: 454-456.

Daniels, R.F.; Burkhart, H.E. 1975. Simulation of individual tree growth and stand development in managed loblolly pine plantations. FWS-5-75. Blacksburg, VA: Virginia Polytechnic Institute and State University, Division of Forestry and Wildlife Resources. 69 p.

Diéguez-Aranda, U.; Burkhart, H.E.; Amateis, R.L. 2006. Dynamic site model for loblolly pine (*Pinus taeda* L.) plantations in the United States. Forest Science. 52: 262-272.

Dyer, M.E.; Burkhart, H.E. 1987. Compatible crown ratio and crown height models for loblolly pine. Canadian Journal of Forest Research. 17: 572-574.

Fox, T.R.; Morris, L.A.; Maimone, R.A. 1989. The impact of windrowing on the productivity of a rotation age loblolly pine plantation. In: Miller, J.H., comp. Proceedings of the 5th biennial southern silvicultural research conference. Gen. Tech. Rep. SO-74. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 133-140

Hegyi, F. 1974. A simulation model for managing jack-pine stands. In: Growth models for tree and stand simulation. Stockholm, Sweden: Royal College of Forestry: 74-90.

Hynynen, J.; Burkhart, H.E.; Allen, H.L. 1998. Modeling tree growth in fertilized midrotation loblolly pine plantations. Forest Ecology and Management. 107: 213-229.

Sharma, M.; Amateis, R.L.; Burkhart, H.E. 2002. Top height definition and its effect on site index determination in thinned and unthinned loblolly pine plantations. Forest Ecology and Management. 168: 163-175.

Smith, J.L.; Burkhart, H.E. 1983. A simulation study assessing the effect of sampling for predictor variable values on estimates of yield. Canadian Journal of Forest Research. 14: 326-330.

Westfall, J.A.; Burkhart, H.E.; Allen, H.L. 2004. Young stand growth modeling for intensively-managed loblolly pine plantations in southeastern U.S. Forest Science. 50: 823-835.

MODELING INDIVIDUAL TREE SURVIVAL

Quang V. Cao[1]

Abstract—Information provided by growth and yield models is the basis for forest managers to make decisions on how to manage their forests. Among different types of growth models, whole-stand models offer predictions at stand level, whereas individual-tree models give detailed information at tree level. The well-known logistic regression is commonly used to predict tree survival probability. In addition to the maximum likelihood approach, a new approach called CDF regression was introduced here to estimate parameters of the tree survival equation.

Each of the two above approaches was evaluated as follows: (1) unadjusted, (2) disaggregated from the whole-stand model, and (3) disaggregated from the combined estimator. Results from this study showed that the tree survival model, when adjusted from the combined estimator, produced the best-ranked two alternatives. The new method, CDF Regression, coupled with the combined estimator, was better than the Maximum Likelihood method in estimating parameters of the logistic regression equation.

INTRODUCTION

Among many different types of growth and yield models, individual-tree simulation models provide the most flexible outputs because growth of an individual tree is the basis for this type of models. Predicting tree survival is an important component of tree-level models. The probability that a tree survives a growing period has been modeled by use of logistic regressions (Hamilton 1974, Hamilton and Edwards 1976, Monserud 1976, Buchman 1979, 1983, Zhang and others 1997, Monserud and Sterba 1999) or other methods (Glover and Hool 1979, Amateis and others 1989, Guan and Gertner 1991a, 1991b).

Maximum likelihood estimation is the most common method for estimating the parameters of a logistic regression model. An alternative method, called CDF Regression, is introduced in this paper.

Stand-level prediction of survival can be predicted directly from a stand survival model, or indirectly by summing individual tree survival probabilities. The predictions could be improved by use of a combined estimator (Yue and others 2008, Zhang 2010), which is a weighted average of outputs from both types of models.

Disaggregation method is a method that links a tree-level model and a stand-level model (Ritchie and Hann 1997). In this method, outputs from the tree-level model are adjusted such that the resulting stand summary matches prediction from a stand-level model.

The objective of this study was to evaluate two methods of estimating parameters of the logistic regression model for predicting tree survival probabilities, Maximum Likelihood and CDF Regression. The evaluation was conducted under the following scenarios:

- Unadjusted tree survival model,
- Tree survival model adjusted by disaggregation from the stand survival outputs, and
- Tree survival model adjusted by disaggregation from the combined estimator.

METHODS

Data

Data used in this study were from 200 plots randomly selected from the Southwide Seed Source Study, which include 15 loblolly pine (*Pinus taeda* L.) seed sources planted at 13 locations across 10 southern states (Wells and Wakeley 1966). Each 0.0164 ha plot consisted of 49 trees, planted at a 1.8 m × 1.8 m spacing. Tree diameters and survival were recorded at ages 10, 15, 20, and 25 years, resulting in a total of 600 growth periods.

The data were randomly divided into two groups of 100 plots each (table 1). The leave-one-out evaluation scheme was applied in this study. Parameters of the tree survival model were estimated from one group, and then used to predict for the other group. The predictions

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Table 1—Stand and tree attributes for the 200 plots used in this study, by group

Group	Age	Stand attributes				Tree attribute	
		# obs.	*Hd*	*TPH*	*BA/ac*	*# obs.*	*Avg. DBH*
1	10	100	9.0	1987	21.52	3257	11.4
	15	100	13.2	1750	31.63	2868	14.7
	20	100	16.4	1303	33.63	2135	17.6
2	10	100	9.2	1977	22.08	3237	11.6
	15	100	13.4	1702	32.03	2788	15.0
	20	100	16.7	1243	33.22	2037	17.9

from both groups were used to compute evaluation statistics for the different methods.

Stand survival equation

The model developed by Cao (2006) was used in this study to predict stand-level survival:

$$\hat{N}_{2i} = \frac{N_{1i}}{1+exp(b_0+b_1 RS_{1i}+b_2 H_{1i}+b_3 N_{1i}/A_{1i}+b_4/A_{1i})}, \quad (1)$$

where $\hat{N}_{2i}$ = predicted number of trees per hectare for plot i at the end of the 5-year growth period, A_{1i} = stand age in years for plot i at the beginning of the growth period, N_{1i} = number of trees per hectare at age A_{1i}, H_{1i} = dominant height in meters at age A_{1i}, $RS_{1i} = \frac{\sqrt{10000/N_{1i}}}{H_{1i}}$ = relative spacing at age A_{1i}.

Tree survival equation

The following logistic regression model was employed to predict tree survival probability ($\hat{p}_{1j}$) of tree j in plot i during the 5-year growth period:

$$\hat{p}_{ij} = \frac{1}{1+exp(b_0+b_1 H_{1i}+b_2 RS_{1i}+b_3 d_{1ij}/Dq_{1i})}, \quad (2)$$

where Dq_{1i} = quadratic mean diameter of plot i at age A_{1i}, and d_{1ij} = dbh of tree j in plot i at age A_{1i}.

CDF Regression

Tree diameters in each plot were sorted from smallest to largest. Let $d_{1i(j)}$ be the j^{th} order statistic for tree dbh at age A_{1i} for plot i, with $d_{1i(j)}$ being the minimum dbh in plot i at age A_{1i}. The empirical CDF (cumulative distribution function) or tree survival, $F_{1i(j)}$, is defined for diameter $d_{1i(j)}$ as follows:

$$F_{i(j)} = F_{i(j-1)} + \delta_{i(j)}/n_{2i}. \quad (3)$$

where $F_{i(0)} = 0$, $\delta_{i(j)} = 0$ if the tree having diameter d_{1ij} is dead and 1 if it survives the 5-year growth period, and n_{2i} = total number of surviving trees in plot i at the end of the growth period. Subscript j varies from 1 to n_{1i}, where n_{1i} = total number of trees in plot i at age A_{1i}.

In the CDF Regression method, the parameters of the tree survival equation (2) were solved to minimize

$$z = \sum_i \sum_j \left(F_{i(j)} - \hat{F}_{i(j)}\right)^2, \quad (4)$$

where $\hat{F}_{i(j)} = \hat{F}_{i(j-1)} + \hat{p}_{i(j)}/n_{2i}$, $\hat{p}_{i(j)}$ = predicted tree survival probability for a tree having diameter $d_{1i(j)}$, and $\hat{F}_{i(0)} = 0$.

Figure 1 shows observed survival CDF for a sample plot and predicted CDF's from the Maximum Likelihood and CDF Regression methods.

Combined estimator

The combined estimator for plot i (N_i^C) is the weighted average of predictions from the stand-level model (N_i^S) and the tree-level model (N_i^T):

$$N^C = wN^T + (1-w)N^S, \quad (5)$$

where w were computed according to the least-squares method described by Tang (1992, 1994) and applied by Zhang and others (2010).

Disaggregation

The predicted tree survival probability ($\hat{p}_{ij}$) of tree j in plot i was adjusted as follows so that the resulting stand survival matched either the prediction from the stand survival model or the combined estimator:

$$\tilde{p}_{ij} = \hat{p}_{ij}^{\alpha_i}, \quad (6)$$

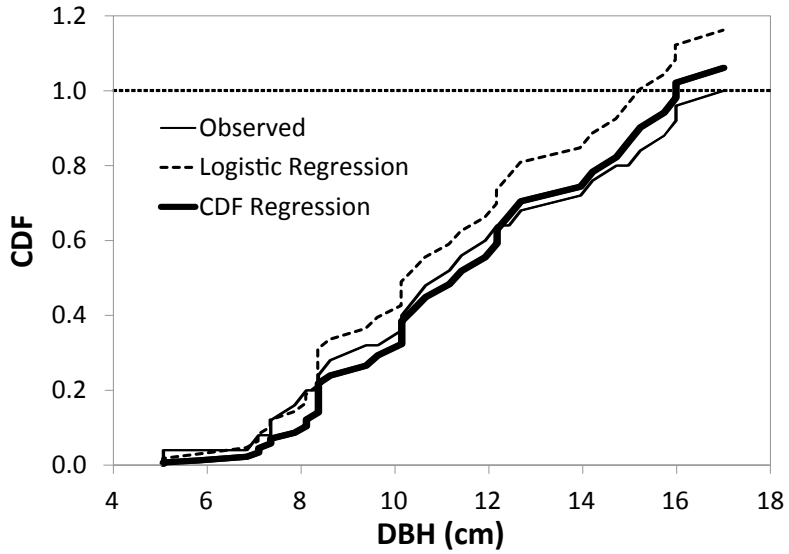


Figure 1—Observed survival CDF for a sample plot and predicted CDF's from the Maximum Likelihood and CDF Regression methods.

where $\tilde{p}_{ij}$ = adjusted tree survival probability, and a_i = coefficient for plot i to adjust probabilities.

Evaluation Criteria

The performance of the different methods was evaluated at the stand level based on the following statistics.

Mean difference: $MD = \sum_i (N_{2i} - \hat{N}_{2i})/m$ (7)

Mean absolute difference: $MAD = \sum_i |N_{2i} - \hat{N}_{2i}|/m$ (8)

Fit index: $R^2 = \sum_i (N_{2i} - \hat{N}_{2i})^2 / \sum_i (N_{2i} - \bar{N}_2)^2$ (9)

where N_{2i} and $\hat{N}_{2i}$ = observed and predicted number of trees per hectare in plot i at the end of the growth period, $\bar{N}_2$ = the average number of trees per hectare at the end of the growth period, and m = total number of plots.

The stand-level evaluation included the whole-stand model, the individual-tree model (with two parameter estimation methods), and the combines estimator (also with two parameter estimation methods).

The tree-level evaluation statistics were:

Mean difference: $MD = \sum_i \sum_j (\delta_{ij} - \hat{p}_{ij}) / \sum_i n_{1i}$ (10)

where $\delta_{ij} = 0$ if tree j in plot i was dead and 1 if it was alive, and n_i = number of trees in plot i.

Mean absolute difference:

$MAD = \sum_i \sum_j |\delta_{ij} - \hat{p}_{ij}| / \sum_i n_{1i}$ (11)

Log-likelihood:

$-2lnL = -2[\sum_i \sum_j \hat{p}_{ij} \ln(\hat{p}_{ij}) + \sum_i \sum_j (1 - \hat{p}_{ij}) \ln(1 - \hat{p}_{ij})]$ (12)

AUC: area under the ROC (Receiving Operating Characteristic) curve. The range for AUC is between 0.5 and 1. The higher the AUC value, the better the fit.

The tree-level evaluation involved six methods. Each of the two parameter estimation methods included three alternatives: the unconstrained model, the Disaggregation method in which outputs from the tree-level model was adjusted from the stand-level predictions, and the Combination method that adjusted the tree-level outputs to match the combined estimator.

The relative rank, developed by Poudel and Cao (2013), was used in this study to display the relative position of each method. The relative rank of method i is defined as

$R_i = 1 + \frac{(k-1)(S_i - S_{min})}{S_{max} - S_{min}}$ (13)

where R_i = the relative rank of method i (i = 1, 2, ..., k), k = number of methods evaluated, S_i = the goodness-of-fit statistic produced by method i, S_{min} = the minimum value of S_i, and S_{max} = the maximum value of S_i.

RESULTS AND DISCUSSION

The individual-tree survival model, with parameters estimated by either the Maximum Likelihood or CDF Regression method, produced stand-level outputs that were inferior to the whole-stand survival model (table 2). This was expected because stand-level outputs from individual-tree models typically suffer

from accumulation of errors (Qin and Cao 2006). On the other hand, the combined estimators did outperform the whole-stand model based on all three evaluation statistics (Table 2). The combined estimator from the CDF Regression method was slightly better than the one from the Maximum Likelihood method.

Table 3 shows the tree-level evaluation statistics for the six methods. The relative ranks for these methods are presented in Table 4.

Unconstrained models

For the unadjusted model, the Maximum Likelihood method clearly outperformed the CDF Regression method in predicting tree survival. It produced better statistics for all evaluation criteria.

Disaggregation

Based on the sum of the relative ranks, disaggregation was better for the CDF Regression than for the Maximum Likelihood method. This was true when disaggregation was either from the whole-stand model or from the combined estimator. The CDF Regression method was a compromise between optimizing for tree-level and stand-level survival prediction. As such, it made sense that this method performed well in conjunction with the combined estimator.

Radar plot

The radar plot based on the relative ranks of four criteria from all methods is shown in figure 2. The method resulting in the smallest area inside the box represents the best method. Figure 3 presents the relative ranks for the best three methods. The CDF Regression method, coupled with the combined estimator, overall ranked best in predicting tree survival, followed by the Maximum Likelihood method, unconstrained and disaggregated by the Combination method.

CONCLUSIONS

Results from this study showed that the Combination approach, in which outputs from the tree survival model were adjusted to match the combined estimator, was the best approach to predict both tree- and stand-level survival. The new method, CDF Regression, when disaggregated from the combined estimator, was better than the Maximum Likelihood method in estimating parameters of the logistic regression equation.

Table 2—Stand-level evaluation statistics for different types of models. Underlined, italic numbers denote the best statistic among the methods

Type	Parameter estimation	MD	MAD	R^2
Whole-stand model		-21.77	160.37	0.8295
Individual-tree model	Maximum Likelihood	***-0.05***	176.28	0.7963
	CDF Regression	55.77	180.36	0.7866
Combined estimator	Maximum Likelihood	-17.11	159.86	0.8322
	CDF Regression	-3.91	***159.40***	***0.8337***

Table 3—Tree-level evaluation statistics for different parameter estimation methods. Underlined, italic numbers denote the best statistic among the methods

Parameter estimation	Method	MD	MAD	-2lnL	AUC
Maximum Likelihood	Unconstrained	***0.0001***	0.2276	***0.7384***	0.7929
	Disaggregation	-0.0134	0.2141	0.7631	0.7821
	Combination	-0.0106	0.2166	0.7391	0.7932
CDF Regression	Unconstrained	0.0337	0.2315	0.7659	0.7907
	Disaggregation	-0.0134	***0.2057***	0.7788	0.7901
	Combination	-0.0026	0.2109	0.7480	***0.7997***

Table 4—Relative rankings for tree-level evaluation statistics. Underlined, italic numbers denote the highest rank among the methods

Parameter estimation	Method	Rank MD	Rank MAD	Rank -2lnL	Rank AUC	Total	Overall Rank
Maximum Likelihood	Unconstrained	***1.00***	5.24	***1.00***	2.93	10.16	2.33
	Disaggregation	3.00	2.61	4.07	6.00	15.68	4.39
	Combination	2.58	3.10	1.09	2.84	9.61	2.13
CDF Regression	Unconstrained	6.00	6.00	4.41	3.56	19.98	6.00
	Disaggregation	3.00	***1.00***	6.00	3.72	13.72	3.66
	Combination	1.40	2.01	2.20	***1.00***	6.60	***1.00***

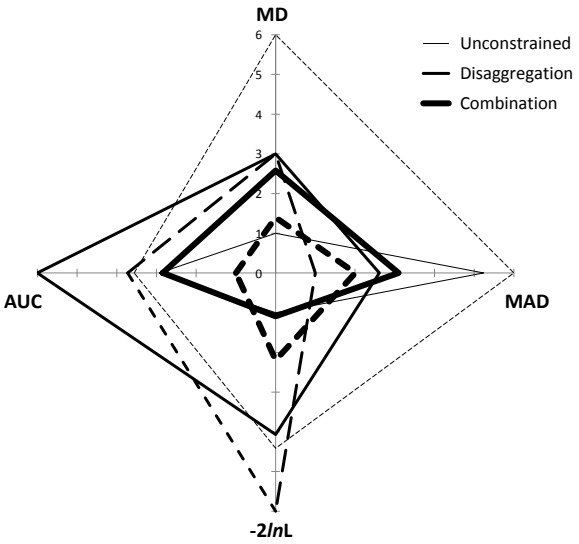


Figure 2—Overall comparison for the Maximum Likelihood (continuous lines) and CDF Regression (dashed lines) parameter estimation methods. Each method was evaluated unadjusted or disaggregated from either the stand model or the combined estimator. The method resulting in the smallest area inside the box represents the best method.

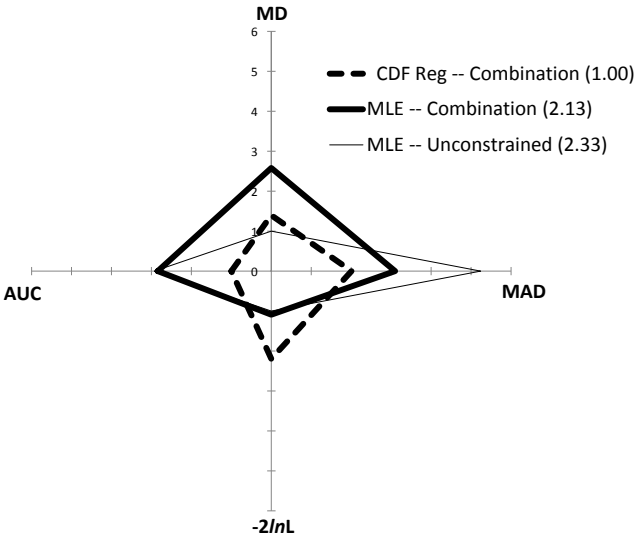


Figure 3—Relative ranks of the three best methods.

LITERATURE CITED

Amateis, R.L.; Burkhart, H.E.; Walsh, T.A. 1989. Diameter increment and survival equations for loblolly pine trees growing in thinned and unthinned plantations on cutover, site-prepared lands. Southern Journal of Applied Forestry. 13: 170-174.

Buchman, R.G. 1979. Mortality functions. In: A generalized forest growth projection system applied to the Lake States region. Gen. Tech. Rep. NC-49. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment: 47-55.

Buchman, R.G. 1983. Survival predictions for major Lake States tree species. Res. Pap. NC-233. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 10 p.

Cao, Q.V. 2006. Predictions of individual-tree and whole-stand attributes for loblolly pine plantations. Forest Ecology and Management. 236: 342-347

Glover, G.R.; Hool, J.N. 1979. A basal area ratio predictor of loblolly pine plantation mortality. Forest Science. 25: 275-282.

Guan, B.T.; Gertner, G. 1991a. Using a parallel distributed processing system to model individual tree mortality. Forest Science. 37: 871-885.

Guan, B.T.; Gertner, G. 1991b. Modeling red pine tree survival with an artificial neural network. Forest Science. 37: 1429-1440.

Hamilton, D.A. 1974. Event probabilities estimated by regression. Res. Pap. INT-152. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 18 p.

Hamilton, D.A.; Edwards, B. M. 1976. Modeling the probability of individual tree mortality. Res. Pap. INT-185. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 22 p.

Monserud, R.A. 1976. Simulation of forest tree mortality. Forest Science. 22: 438-444.

Monserud R.A.; Sterba, H. 1999. Modeling individual tree mortality for Austrian forest species. Forest Ecology and Management. 113: 109-123.

Poudel, K.; Cao, Q.V. 2013. Evaluation of methods to predict Weibull parameters for characterizing diameter distributions. Forest Science. 59: 243-252.

Qin, J.; Cao, Q. V. 2006. Using disaggregation to link individual-tree and whole-stand growth models. Canadian Journal of Forest Research. 36: 953-960.

Ritchie, M.W.; Hann, D.W. 1997. Implications of disaggregation in forest growth and yield modeling. Forest Science. 43: 223-233.

Tang, X. 1992. Forecasting theory with its applications (in Chinese with English abstract). Chengdu, China: University of Electronic Science and Technology of China: 288.

Tang, X. 1994. Some new results of the efficient frontier of portfolio investment. J. UEST China. 23:301-305.

Wells, O.O.; Wakeley, P. C. 1966. Geographic variation in survival, growth, and fusiform rust infection of planted loblolly pine. Forest Science Monograph. 11(Supplement to Number 3). 40 p.

Yue, C.F.; Kohnle, U; Hein, S. 2008. Combining tree- and stand-level models: a new approach to growth prediction. Forest Science. 54: 553-566.

Zhang, S.; Amateis, R.L.; H.E. Burkhart, H.E. 1997. Constraining individual tree diameter increment and survival models for loblolly pine plantations. Forest Science. 43: 414-423.

Zhang, X.; Lei, Y.; Cao, Q.V. 2010. Compatibility of stand basal area predictions based on forecast combination. Forest Science. 56: 552-557

AN *UPDATED* WHOLE STAND GROWTH AND YIELD SYSTEM FOR PLANTED LONGLEAF PINE IN SOUTHWEST GEORGIA

John R. Brooks and Steven B. Jack[1]

Abstract—An updated whole stand growth and yield system for planted longleaf pine (*Pinus palustris*) was developed from permanent plot data collected annually over a 13 to 16 year period. The data set consists of 15 intensively managed longleaf pine plantations that are located in Lee, Worth, Mitchell, and Baker counties in southwest Georgia. Stand survival, dominant height, basal area and cubic foot volume yield models were developed for both low and high planting densities. Model prediction error remained low for both planting density classes. Yield models are an improvement over those published in 2006 (Brooks and Jack, 2006), as eight additional growth remeasurements were added which improved projection accuracy for stands older than 10 years. Models are designed for application in unthinned stands (prior to onset of self thinning) in this region between stand age 2 and 25 years.

INTRODUCTION

An updated whole stand growth and yield system was developed for unthinned longleaf (*Pinus palustris*) plantations located in the Flint River Basin of southwest Georgia. The data is from a long term growth and yield study initiated in 1996 based on annual remeasurement data from 15 managed longleaf pine plantations located in Lee, Worth, Mitchell, and Baker counties in southwest Georgia. A system of whole stand projection models were developed for stand survival, dominant height, basal area per acre, and cubic foot volume outside bark (ob) per acre. This system of models updates the previously published models by Brooks and Jack (2006) for the same dataset, however the new models are based on approximately four times the number of growth intervals reported in 2006 and extends plantation prediction age from 18 to 24 years.

METHODS

Study Description

Rectangular fixed area plots were established at different dates and have been remeasured annually; thus the number of measurements available per plot ranges from 7 to 14. Sample plots are approximately 0.1 (mean 0.10585) acre in size with tree age ranging from 2 to 24 years old. Planting densities ranged from 338 to 940 trees per acre. Stand level description of the major variables are shown in table 1. A total of 184 unique non-overlapping growth intervals were available for modeling. This dataset contains two subpopulations, those with planting densities over 750 trees per acre (798 to 940 trees per acre) noted as high density plantings and those with planting densities lower than 450 trees per acre (338 to 445 trees per acre) noted as low density plantings. The groups were separated due to distinct differences in stand survival and dominant height growth patterns. The high density plantings were predominantly cutover sites that were mechanically and chemically site prepared while the low density plantings were old field sites that were mechanically site prepared. All locations were predominantly loamy sands except for two plantings in the high density group which were classified as sands.

At each measurement date, diameter at breast height (d.b.h.) was measured with a diameter tape and recorded for every tree to the nearest 0.01 inch. Total tree height was measured with a height pole or an Impulse laser (depending upon tree size) and recorded to the nearest 0.1 foot. Trees < 15 feet were measured with a height pole, while taller trees were measured with an Impulse 200 laser. Crown class was recorded for every sample tree. The traditional definition of crown class was slightly modified in order to assign crown class to the younger aged stands. The younger plantations generally have wider initial planting spacing, and thus all trees receive full sunlight. The codominant crown class was defined as those trees that make up the average crown canopy, while intermediate and suppressed classes were assigned to those

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Table 1—Range of stand level variables for longleaf plantations in southwest Georgia

	Age (yr)	TPA	BA (ft^2/ac)	HD (ft)	VOL (ft^3/ac)
Low Density					
Min	7	516	36.7	13.7	258.6
Max	24	877	153.5	55.4	4,173.9
High Density					
Min	3	263	4.7	3.9	20.0
Max	20	372	117.1	55.9	3,215.9

TPA = trees per acre, BA = basal area per acre, HD = dominant height and VOL = total cubic foot volume (ob) per acre.

trees that were visually shorter (and usually less vigorous) than the trees that constitute the average crown height. Cubic foot volumes are based on a taper function developed by Brooks and others (2002), utilizing trees sampled from these same plantations.

Survival Projection

Several survival models, commonly employed for loblolly (*Pinus taeda*) and slash pine (*Pinus ellottii*) plantations throughout the South, were tested for their ability to accurately predict stand survival. Models were initially fit independently from other stand level models to test which model forms performed best, based on root mean square error (RMSE) and visual examination of the residuals. Mortality for the high density plantings was greater than the low density plantings and exhibited a typical decrease in survival with age while survival in the low density plantings exhibited negligible mortality by age 20. To provide the best estimates, separate models were fit to the two density classes.

Dominant Height Projection

An algebraic difference equation of a modified Chapman-Richards height/age projection function was tested to model dominant height. This equation is of the form:

$$HD_2 = HD_1\left[\frac{1-Exp\left(\beta_1 * A_2\right)}{1-Exp\left(\beta_1 * A_1\right)}\right]^{\beta_2} \quad (1)$$

where

A_1 = initial plantation age (yr) at time 1,
A_2 = future plantation age (yr) at time 2,
HD_2 = projected dominant height (ft) at A_2,
HD_1 = current dominant height (ft) at A_1, and
β_1, β_2 = parameters to be estimated from the data.

This equation form has been used successfully in loblolly (Pienaar and Shiver 1980), slash (Pienaar and Shiver 1984) and longleaf (Brooks and Jack 2006) pine plantations. Initial model forms were initially established independent of other stand level models. Evaluation of these independent model forms was based on RMSE and visual evaluation of residuals. Since average dominant height growth differed by planting density group, separate parameter estimates were established for each group. Through age 20, the lower density planting exhibited a higher dominant height development pattern than the higher density planting group.

Basal Area Projection

Several algebraic difference models, commonly employed for loblolly and slash pine plantations throughout the south, were tested for their ability to accurately predict future stand basal area per acre. These model forms were fit as a function of the change in age, trees per acre and dominant height. Again, models were initially fit independently from other stand level models to test which model forms performed best, based on root mean square error (RMSE) and visual examination of the residuals. It became apparent that the same model form would not predict future basal area equally well for both planting density groups.

Volume Projection

Several algebraic difference models, commonly employed for loblolly and slash pine plantations throughout the south, were tested for their ability to accurately predict future stand total cubic foot volume per acre. Again, models were initially fit independently from other stand level models to test which model forms performed best, based on root mean square error (RMSE) and visual examination of the residuals. As found with the individual basal area projection models, the same model form for both planting density groups

would not predict future basal area equally well so individual models were developed.

Whole Stand Model Projection Systems

Once the best form of each independent equation was determined, the four models were then fit as a family of projection equations using seemingly unrelated regressions (SUR) using SAS MODEL (SAS 2010). Model forms for each stand parameter were adjusted to provide the best family of projection equations, removing non-significant variables in each individual model form. Final model forms were based on minimum RMSE and visual examination of the residuals for each model. Because of the exhibited differences in growth patterns between the two planting density groups, models were developed separately for the low and high density planting groups.

RESULTS

Parameter estimates and fit statistics are presented by planting density group.

Low Density Planting Group

The final family of projection equations for the low planting density group is of the form:

$$N_2 = Exp\left(Ln\left(N_1\right)+\alpha_1\left(A_2-A_1\right)\right)$$

$$HD_2 = HD_1 * \left(\frac{\left(1-Exp\left(\beta_1 A_2\right)\right)}{\left(1-Exp\left(\beta_1 A_1\right)\right)}\right)^{\beta_2} \quad (2)$$

$$B_2 = Exp\left(Ln\left(B_1\right)+\sigma_1\left(\frac{1}{A_2}-\frac{1}{A_1}\right)+\sigma_2\left(Ln\left(HD_2\right)-Ln\left(HD_1\right)\right)+\sigma_3\left(Ln\left(N_2\right)-Ln\left(N_1\right)\right)\right)$$

$$V_2 = Exp\left(Ln\left(V_1\right)+\gamma_1\left(Ln\left(HD_2\right)-Ln\left(HD_1\right)\right)+\gamma_2\left(Ln\left(B_2\right)-Ln\left(B_1\right)\right)\right)$$

where

N_1 = initial stand trees per acre at A_1,
N_2 = future trees per acre at A_2,
B_1 = initial basal area (ft^2/ac) at A_1,
B_2 = future basal area (ft^2/ac) at A_2,
V_1 = initial total cubic feet per acre at A_1,
V_2 = future total cubic feet per acre at A_2,
$\alpha_i, \beta_i, \sigma_i, \lambda_i$ = parameters to be estimated from the data,

All other variables as previously defined.

Model fit statistics are displayed in table 2 and parameter estimates and confidence intervals are displayed in table 3. Residual analysis (fig. 1) indicated a slight positive bias in trees per acre, however most predictions errors were less than 10 trees per acre. Basal area residuals exhibited some negative bias for stands less than 10 years old, although this error was less than 5 square feet per acre. No other residual irregularities were noted. Due to the limited size of the dataset, no independent verification of this prediction system was tested.

High Density Planting Group

The final family of projection equations for the high planting density group is of the form:

$$N_2 = Exp\left(Ln\left(N_1\right)+\alpha_1\left(A_2-A_1\right)\right)$$

$$HD_2 = HD_1 * \left(\frac{\left(1-Exp\left(\beta_1 A_2\right)\right)}{\left(1-Exp\left(\beta_1 A_1\right)\right)}\right)^{\beta_2} \quad (3)$$

$$B_2 = Exp\left(Ln\left(B_1\right)+\sigma_1\left(Ln\left(HD_2\right)-Ln\left(HD_1\right)\right)+\sigma_2\left(Ln\left(N_2\right)-Ln\left(N_1\right)\right)\right)$$

$$V_2 = Exp\left(Ln\left(V_1\right)+\gamma_1\left(Ln\left(HD_2\right)-Ln\left(HD_1\right)\right)\right)$$

Where all variables as previously defined.

Model fit statistics are displayed in table 4 and parameter estimates and confidence intervals are displayed in table 5. Residual analysis (fig. 2) indicated a slight positive bias in trees per acre, however most predictions errors were less than 10 trees per acre. No other residual irregularities were noted. Due to the limited size of the dataset, no independent verification of this prediction system was tested.

DISCUSSION

A new set of whole stand models were developed for longleaf pine plantations in southwest Georgia. Previous models were published (Brooks and Jack 2006), however these earlier models were based on very young stands with limited remeasurement data. The current proposed models are based on unthinned stands from age 2 to 24 with almost 4 times the number of unique growth intervals for model fitting. The dataset consists of both low density plantings, similar to those created under several Conservation Reserve Programs, and high planting densities, similar to densities used in conventional reforestation techniques. To minimize prediction error for both planting densities, separate models or parameter estimates were necessary for

each planting density class. These differences were not apparent during the development of the initial models for younger stands in 2006 (Brooks and Jack 2006), but were obvious as stands developed to older ages. Model prediction errors for the fitted dataset were within acceptable ranges and most parameters estimates were significant. The exceptions were with the basal area projection models for both low and high density plantings, where the single parameter estimates representing stand density, trees per acre (TPA), were significant at the 0.18 and 0.07 probability level. These variables were highly scrutinized and finally included in the final models since the prediction models that included these variables were highly significant and that inclusion of the stand density variable (TPA) greatly improved prediction accuracy relative to the stand densities represented in this dataset for basal area projection. Initial comparison with models for west gulf planted longleaf (Lohrey and Bailey 1976) provided less than acceptable results, especially for prediction of dominant height.

Table 2—Nonlinear summary of residual errors for the low density plantation family of projection equations

Equation	SSE	MSE	RMSE	R^2	Adj-R^2
N_2	3,016.2	51.41	7.17	0.9292	0.9300
HD_2	30.4	0.52	0.72	0.9920	0.9920
B_2	661.6	11.71	3.42	0.9619	0.9608
V_2	380,908.0	6,989.10	83.60	0.9852	0.9842

Table 3—Nonlinear seemingly unrelated regressions (SUR) parameter estimates and fit statistics for the low density plantation family of projection equations

Parameter	Estimate	Approx. Std Err	t-value	Approx Pr>|t|
b_1	-0.13122	0.0135	-9.69	<.0001
b_2	2.86595	0.3242	8.84	<.0001
a_1	-0.01180	0.0030	-3.93	0.0002
s_1	-97.90100	30.8709	-3.17	0.0025
s_2	-5.48162	2.0454	-2.68	0.0097
s_3	4.20403	3.1079	1.35	0.1817
l_1	1.20799	0.1686	7.16	<.0001
l_2	0.79831	0.1713	4.66	<.0001

Table 4—Nonlinear summary of residual errors for the high density plantation family of projection equations

Equation	SSE	MSE	RMSE	R^2	Adj-R^2
N_2	9,025.3	139.90	11.83	0.9886	0.9887
HD_2	39.8	0.62	0.79	0.9923	0.9924
B_2	659.5	10.67	3.27	0.9835	0.9829
V_2	394,021.0	6,221.40	78.88	0.9928	0.9927

Table 5—Nonlinear seemingly unrelated regressions (SUR) parameter estimates and fit statistics for the high density plantation family of projection equations

Parameter	Estimate	Approx. Std Err	t-value	Approx Pr>|t|
a_1	-0.01207	0.0022	-5.56	<.0001
b_1	-0.10104	0.0084	-12.09	<.0001
b_2	2.65133	0.2191	12.10	<.0001
s_1	1.16647	0.0706	16.51	<.0001
s_2	0.56564	0.3023	1.87	0.066
l_1	2.08282	0.0733	28.43	<.0001

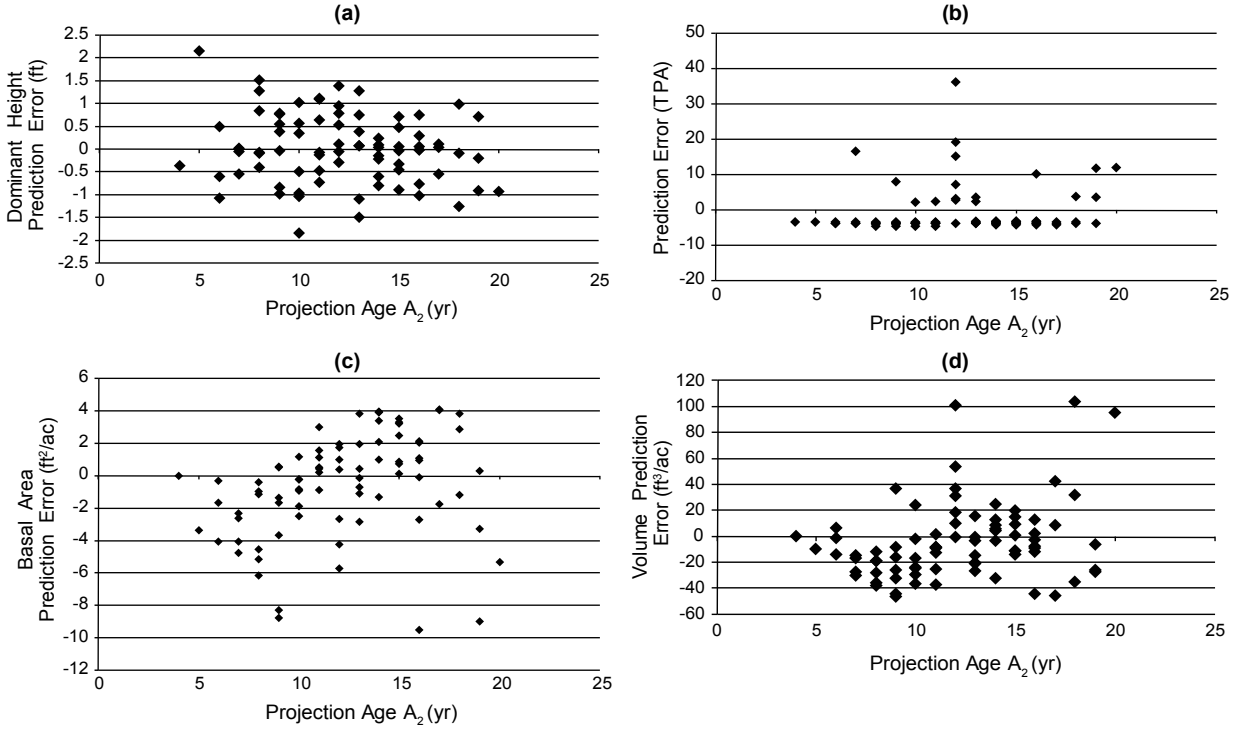


Figure 1—Model residuals for dominant height (a), tree per acre (b), basal area per acre (c), and cubic foot volume per acre (d) for low density plantings.

Comparisons were also made to newly developed whole stand models published by Gonzalez-Benecke and others (2012) for planted longleaf pine plantations in the west gulf region. Using the existing southwest Georgia dataset, their whole stand density (TPA) model compared favorably for both planting density classes while their basal area model tended to over predict this variable. In addition, considerable error and bias was apparent for the whole stand volume models for both planting densities. In an attempt to isolate the source of this difference, the individual tree volume model (Gonzalez-Benecke and others 2014) was compared to the volume models developed for southwest Georgia (Brooks and others 2002). No large differences were found in this comparison. In addition to testing tree volume, the individual tree height model was tested against the measured tree heights from the Brooks and others (2002) study. Differences in average tree height by diameter class was exhibited for all diameter classes, with the west gulf model over predicting tree height from 10 to 15 feet between the 2 and 12-inch class represented in this study.

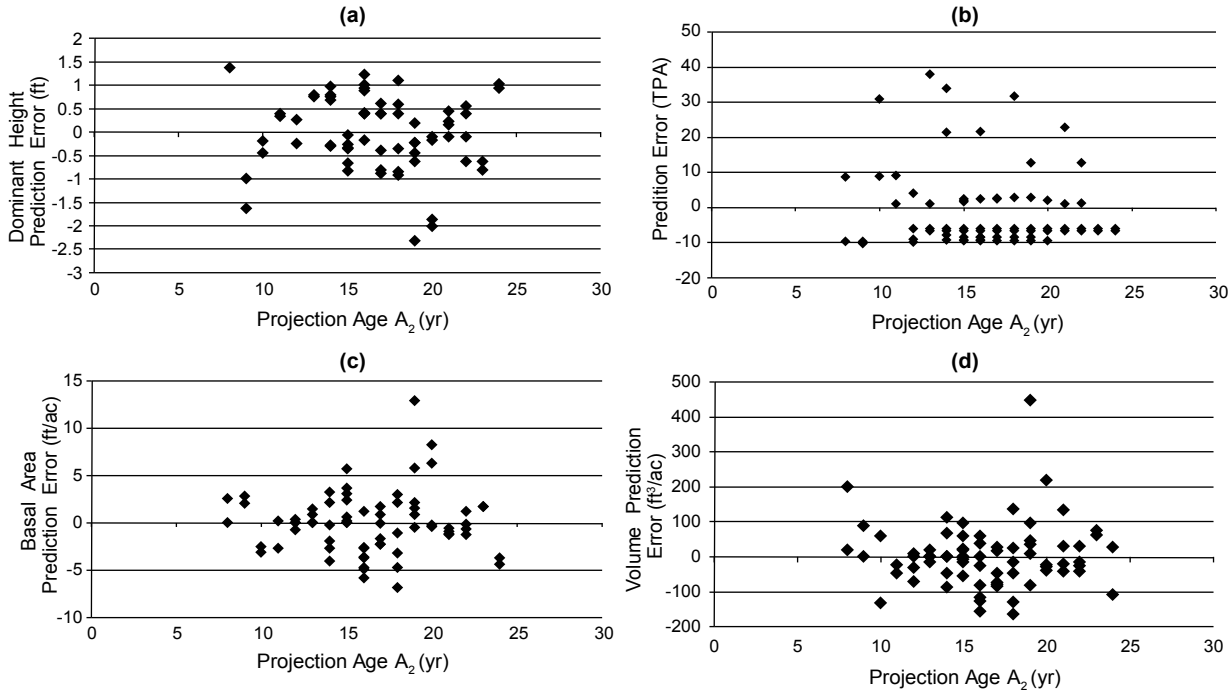


Figure 2—Model residuals for dominant height (a), tree per acre (b), basal area per acre (c), and cubic foot volume per acre (d) for high density plantings.

These models should perform well for planted longleaf in southwest Georgia but caution should be employed if extrapolating to stands outside this region or if heavy self-thinning mortality is evident.

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LITERATURE CITED

Brooks, J.R.; Martin, S.; Jordan, J.; Sewell, C. 2002. Interim taper and cubic-foot volume equations for young longleaf pine plantations in southwest Georgia. In: Outcalt, K.W., ed. Proceedings of the 11th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-48. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 467-470.

Brooks, J.R.; Jack, S.B. 2006. A whole stand growth and yield system for young longleaf pine plantations in southwest Georgia. In: Connor, Kristina F., ed. 2006. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 317-318.

Gonzalez-Benecke, C.A.; Gezan, S.A.; Martin, T.A. [and others]. 2014. Individual tree diameter, height, and volume functions for longleaf pine. Forest Science. 60(1): 43-56.

Gonzalez-Benecke, C.A.; Gezan, S.A.; Leduc, D.J. [and others]. 2012. Modeling survival, yield, volume partitioning and their response to thinning for longleaf pine plantations. Forests. 2012(3): 1104-1132.

Lohrey, R.E.; Bailey, R.L. 1976. Yield tables and stand structure for unthinned longleaf pine plantations in Louisiana and Texas. Res. Pap. SO-133. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 53 p.

Pienaar, L.V.; Shiver, B.D. 1980. Dominant height growth and site index curves for loblolly pine plantations in the Carolina flatwoods. Southern Journal of Applied Forestry. 4(1): 54-59.

Pienaar, L.V.; Shiver, B.D. 1984. The effect of planting density on dominant height in unthinned slash pine plantations. Forest Science. 30(4): 1059-1066.

SAS Institute Inc. 2010. SAS/ETS user's guide. Version 9.3. Cary, NC: SAS Institute Inc. 3,582 p.

A WEB BASED GROWTH AND YIELD SYSTEMS FOR SOUTHEASTERN PINE AND HARDWOOD STANDS

Jacob R. Beard, Thomas G. Matney, Emily B. Schultz, Stephen C. Grado, and Charles O. Sabatia[1]

Abstract—Faced with the challenges of maintaining customer-specified objectives and meeting deadlines, forest land managers must integrate various mensurational information sources for developing management scenarios. Cost efficient measures are a consideration to save both time and money for a client. While inventory data collection is time and money intensive, data processing does not have to be costly. This paper presents an open access, non-fee website that will host several common growth and yield models. The growth and yield web-based system was developed by scientists at Mississippi State University. Software programs are also available in standalone versions with additional capabilities. Users may project future yields by either implementing combinations of thinning procedures based on a specified residual stand basal area, or projecting the stand to a desired age. Initial stand data and management options for growth and yield models are described in a downloadable user manual. Forest land managers should find this website to be a valuable tool for estimating the growth and yield needs of their clients. Other decision support utilities included on the website are TProfile©, TVolume©, and calculators for estimating required sample size, confidence limits and sampling error from stratified and unstratified random samples (Matney 1996a, Matney 1996b).

INTRODUCTION

The main goal of forest biometrics is to apply mathematical and statistical estimators and models for assisting land managers in decision support. The average reader of forest biometrics literature is often overwhelmed by the task of making sense of what is being presented to them. Everyone can recall the first time they looked at a profile equation article; it was obvious that one would either have to ask a lot of questions or study some higher mathematics texts. This, however, should not be the way that a typical reader should feel when presented with an exciting new method, or finding, regarding forest biometrics. It is uncommon to find published forest biometrics/mensurational literature that include digital access to the source code, or programs, to implement the proposed new procedure. While this is not always true, the availability of a centralized source for growth and yield models and other mensurational information is severely lacking. The website being developed at Mississippi State University (MSU) will allow users to select growth models applicable to their needs. This website repository for software is created with the forest manager in mind to promote the understanding and use of forest mensuration and models by concentrating the information into a central depository.

Availability of Growth and Yield Models

MSU and several other institutions have developed local/regional growth and yield models for important commercial species, stand origins, and regions. Both industry and research data have been analyzed in producing the models. MSU has been instrumental in the development of computer models for natural longleaf (*Pinus palustris* Mill.), plantation loblolly pine (*Pinus taeda* L.), plantation slash pine (*Pinus elliottii* Engelm.), and red oak-sweetgum bottomland forests (Farrar and Matney 1994, Matney and Farrar 1992, Matney and others 1987, Schultz and others 2010). Although these models are already available as standalone applications, current computer technologies permit dissemination of information in both web-based and standalone format. The initial website being implemented will include all of the MSU models and MSU's web-based and standalone implementation of the widely used PTAEDA model (Burkhart and others 2008). In addition, general biometrics applications, such as sample size estimators, and software such as TVolume© and TProfile©, will be made available (Matney 1996a, Matney 1996b). For ease of accessibility, online versions of models will be present with the option to download standalone versions. Some growth and yield models, particularly those of

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the past, have authors who are no longer active in their respective institutions. These models would possibly need to have missing model components fitted to the original data, in order to permit the development of a stable computer model.

Third-Party Growth and Yield Model Case Study

The implementation of PTAEDA 4.0 was a case study for determining the feasibility of creating a growth and yield model from publicly-available literature (Burkhart and others 2008). PTAEDA 4.0 was chosen because it is a well documented growth and yield model that has a standalone computer program. For clarification, the emulated model developed from this article will be hereafter described as "Gateway to PTAEDA," to distinguish it from the original computer model developed by Virginia Tech, PTAEDA 4.0 (Burkhart and others 2008). The results obtained from MSU's version of PTAEDA and the actual PTAEDA correspond very well.

MATERIALS AND METHODS

The case study version, Gateway to PTAEDA, employs a five step process of: 1) generate a Weibull diameter at breast height (DBH) distribution, 2) randomly assign trees to a tree-spacing lattice, 3) employ establishment management regimes, 4) grow to crown closure, and 5) process competitive growth models. PTAEDA 4.0 as established by Virginia Tech presents a stochastic process. Stochastic models provide a picture of the variability of yields that might occur. They are not ideal for practical use. In Gateway to PTAEDA, a constant random number and tree list of 3000+ trees are employed for the generation of the Weibull distribution to insure consistent answers identical to averaged stochastic trials by the user. Users may also elect to implement the stochastic model. Gateway to PTAEDA was programmed with C++ (ISO/IEC, 2014). As it is intended for both online and offline use, the solution was debugged in Microsoft Visual Studio 2010© prior to implementation (Microsoft 2010a).

The third step within the Gateway to PTAEDA process, 'employ establishment management regimes', involves a user choice between three different data entry scenarios. In the first scenario, the user provides existing conditions for a newly established, one to two year old loblolly plantation. The second scenario allows the user, who has an existing stand of loblolly pine, to enter a 'DBH'-'total height' tree list for the purpose of analysis. The third available scenario lets the user depict a hypothetical stand via stand totals. Based on the management stage of a user's property, this allows for a dynamic approach from which optimal management can be ascertained. Thinning models for Gateway to PTAEDA were taken from the MSU loblolly pine cooperative model. For the existing stand scenario, regional mean tree arithmetic and quadratic diameter prediction from the MSU loblolly pine cooperative model were used to generate the initial stand Weibull DBH distribution.

RESULTS

Gateway to PTAEDA as a standalone program provides the user with the three scenario windows at startup. Each window represents a separate growth scenario (described in the methods). Figure 1 displays the three scenario windows. The online version of Gateway to PTAEDA allows for the user to select a scenario and returns the appropriate HTML page for the selected scenario. In addition to the online reporting of results, users are able to download a Microsoft Excel© spreadsheet of the output (Microsoft 2010b). Figures 2 through 4 show user entry, online output, and the Microsoft Excel© output for the online Gateway to PTAEDA version (Microsoft 2010b).

DISCUSSION

Gateway to PTAEDA allows users to project future yields by implementing combinations of thinning procedures based on a specified residual stand basal area, and/or projecting a stand to a desired age. Growth scenario choice allows for management flexibility and provides a valuable tool for estimating growth and yield needs. This case study demonstrates the feasibility of literature adaption for growth and yield programming. The web-based version of Gateway to PTAEDA can be found at http://www.consultingforester.net/GrowthAndYield/PTAEDA.htm.

The growth and yield, and mensurational data web site being developed at MSU when completed will provide a central depository for forest biometric information support. MSU is committed to populating the website with MSU's biometrical store of information that it has developed over the past forty years. MSU will also select published models which have sufficient information to develop a web/standalone computer model that is stable. Anyone interested in contributing information for inclusion on the website that MSU is developing should contact MSU at jacob.beard@cfr.msstate.edu. Full decision support implementation of other growth and yield models will most likely not be possible because many of them do not have all of the required component equations. In some cases the missing equations can possibly be developed from the original source data or another similar data source.

LITERATURE CITED

Burkhart, H.E.; Amateis, R.L.; Westfall, J.A.; Daniels, R.F. 2008. PTAEDA 4.0: Simulation of individual tree growth, stand development and economic evaluation in loblolly pine plantations. Blacksburg, Virginia, 24061.

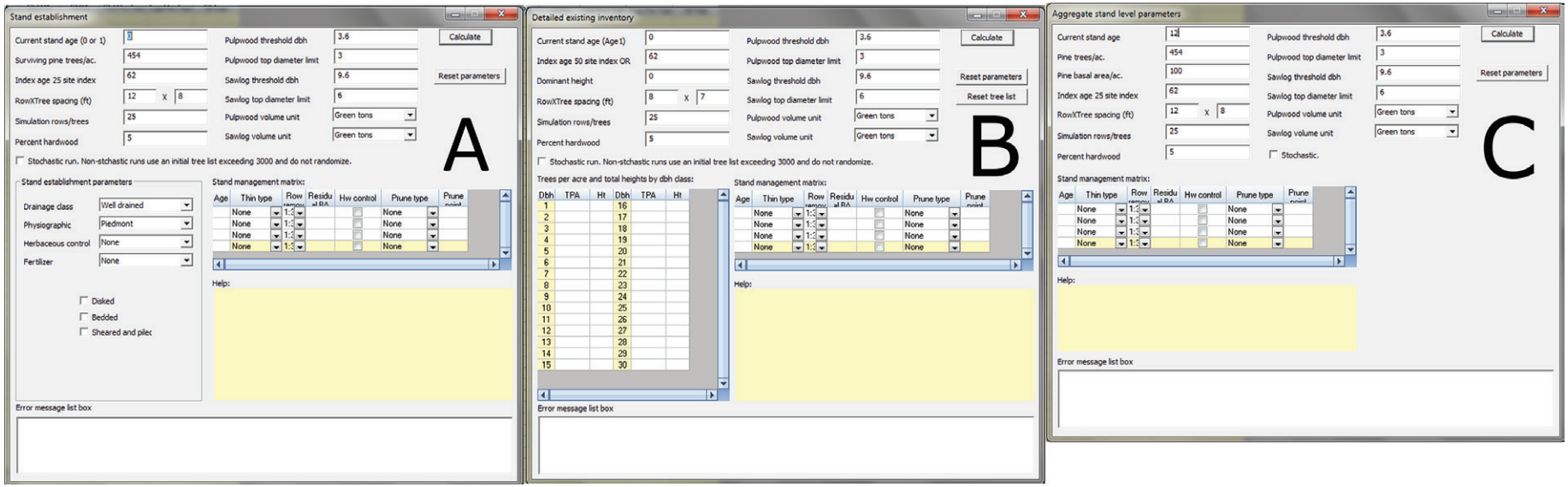
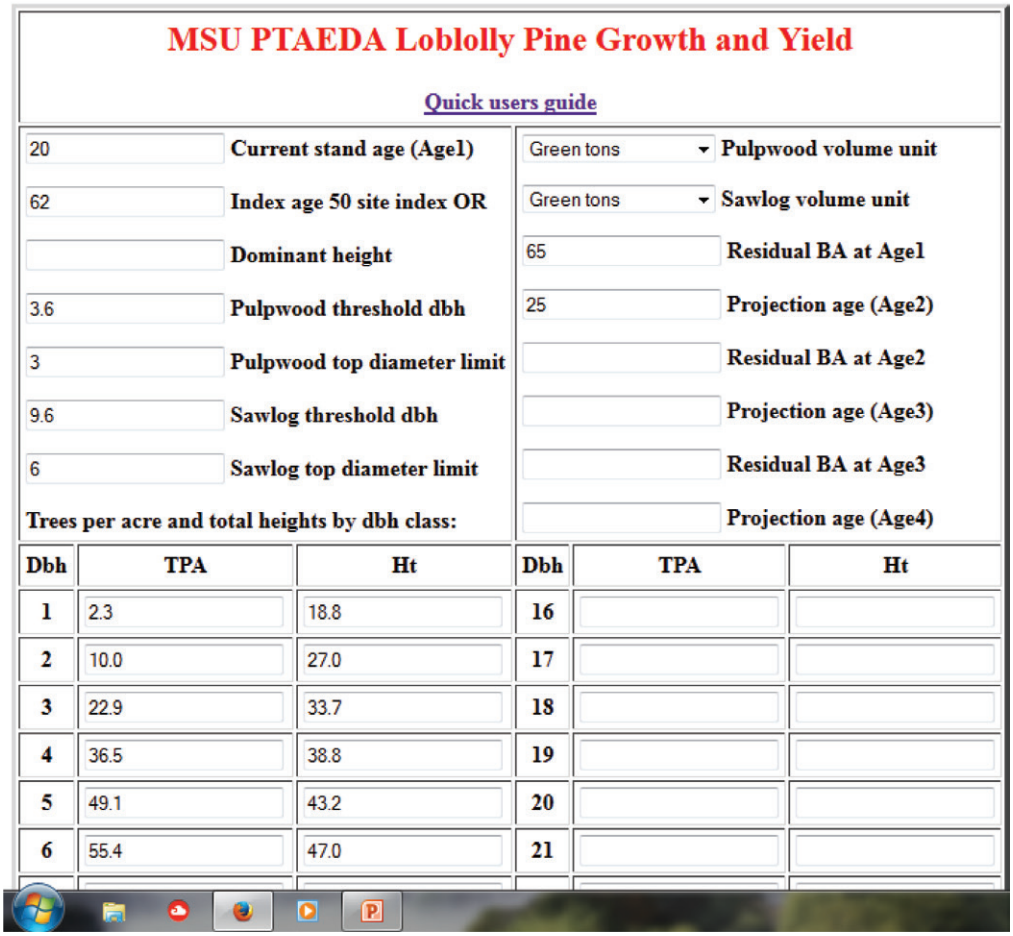


Figure 1—(A) Established stand input scenario screen; (B) Tree list input scenario screen; (C) Hypothetical stand input scenario screen.



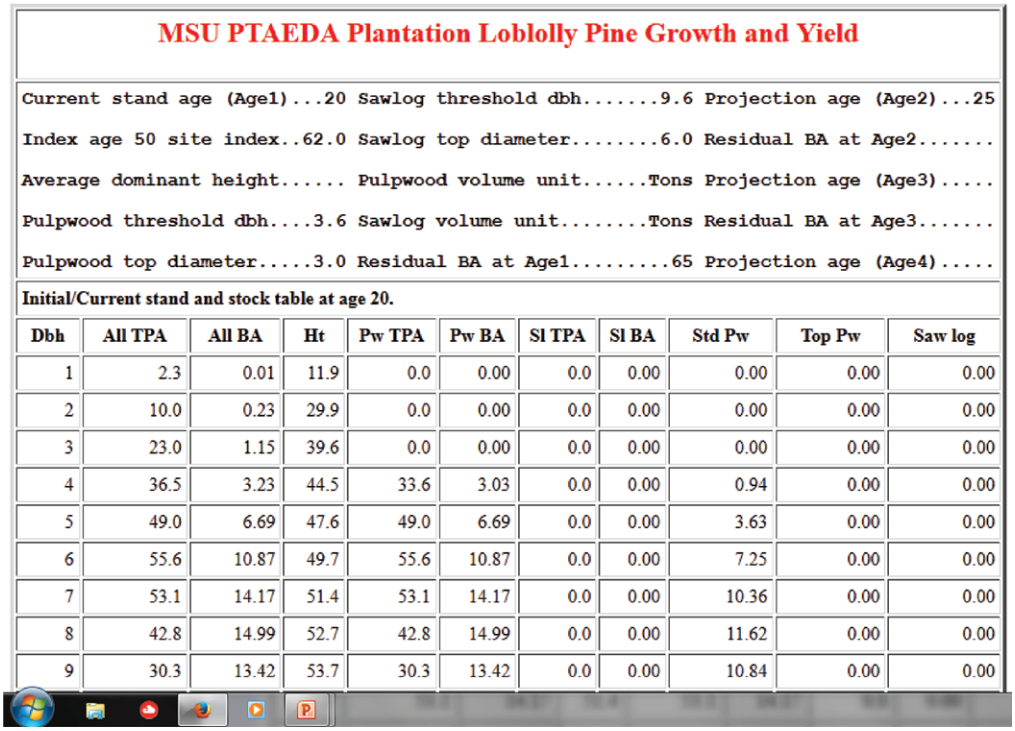
MSU PTAEDA Loblolly Pine Growth and Yield

Quick users guide

20	Current stand age (Age1)	Green tons	Pulpwood volume unit
62	Index age 50 site index OR	Green tons	Sawlog volume unit
	Dominant height	65	Residual BA at Age1
3.6	Pulpwood threshold dbh	25	Projection age (Age2)
3	Pulpwood top diameter limit		Residual BA at Age2
9.6	Sawlog threshold dbh		Projection age (Age3)
6	Sawlog top diameter limit		Residual BA at Age3
Trees per acre and total heights by dbh class:			Projection age (Age4)

Dbh	TPA	Ht	Dbh	TPA	Ht
1	2.3	18.8	16		
2	10.0	27.0	17		
3	22.9	33.7	18		
4	36.5	38.8	19		
5	49.1	43.2	20		
6	55.4	47.0	21		

Figure 2—Online Gateway to PTAEDA user entry page.



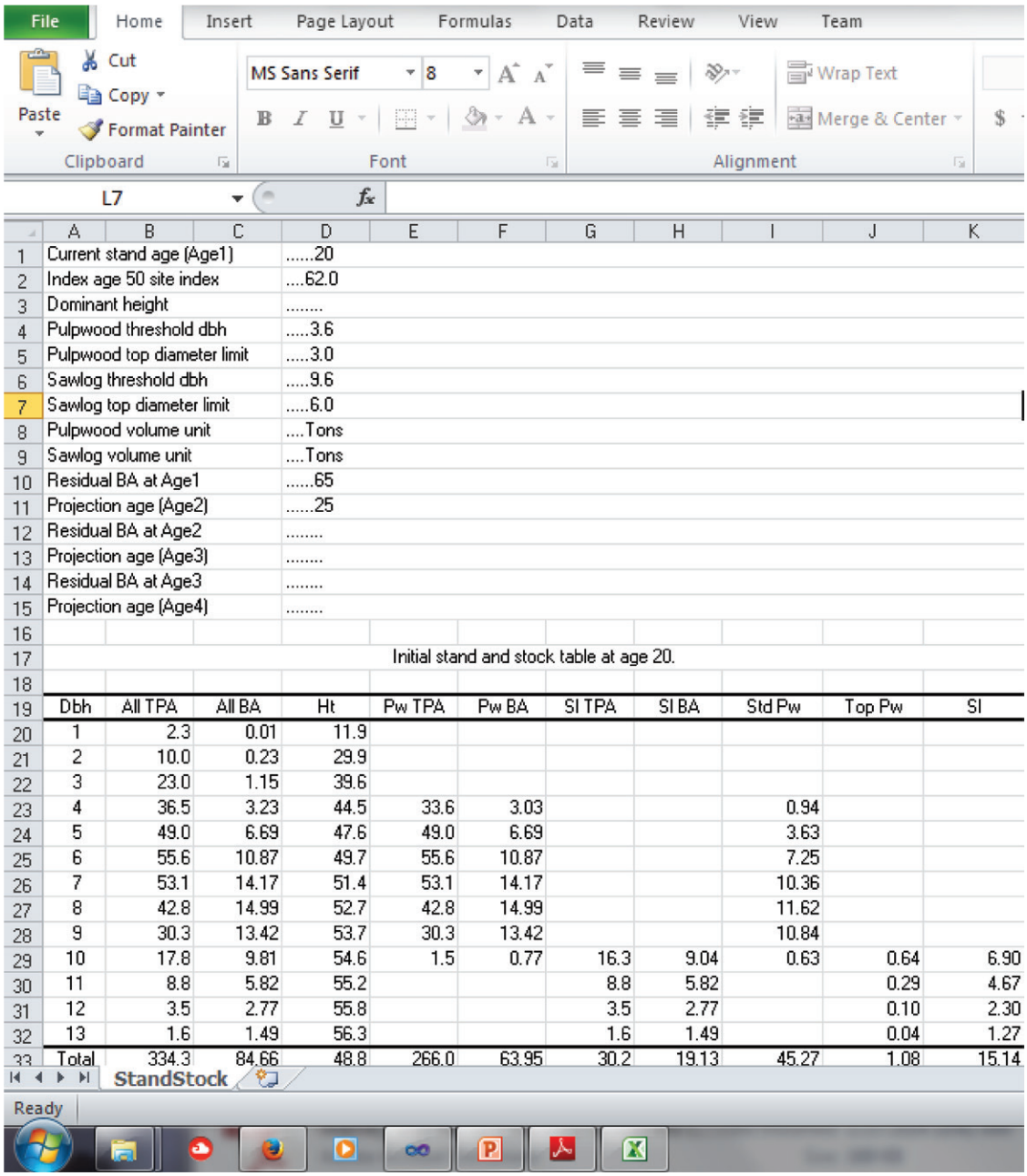
MSU PTAEDA Plantation Loblolly Pine Growth and Yield

```
Current stand age (Age1)...20 Sawlog threshold dbh.......9.6 Projection age (Age2)...25
Index age 50 site index..62.0 Sawlog top diameter........6.0 Residual BA at Age2.......
Average dominant height...... Pulpwood volume unit......Tons Projection age (Age3).....
Pulpwood threshold dbh....3.6 Sawlog volume unit........Tons Residual BA at Age3.......
Pulpwood top diameter.....3.0 Residual BA at Age1.........65 Projection age (Age4).....
```

Initial/Current stand and stock table at age 20.

Dbh	All TPA	All BA	Ht	Pw TPA	Pw BA	Sl TPA	Sl BA	Std Pw	Top Pw	Saw log
1	2.3	0.01	11.9	0.0	0.00	0.0	0.00	0.00	0.00	0.00
2	10.0	0.23	29.9	0.0	0.00	0.0	0.00	0.00	0.00	0.00
3	23.0	1.15	39.6	0.0	0.00	0.0	0.00	0.00	0.00	0.00
4	36.5	3.23	44.5	33.6	3.03	0.0	0.00	0.94	0.00	0.00
5	49.0	6.69	47.6	49.0	6.69	0.0	0.00	3.63	0.00	0.00
6	55.6	10.87	49.7	55.6	10.87	0.0	0.00	7.25	0.00	0.00
7	53.1	14.17	51.4	53.1	14.17	0.0	0.00	10.36	0.00	0.00
8	42.8	14.99	52.7	42.8	14.99	0.0	0.00	11.62	0.00	0.00
9	30.3	13.42	53.7	30.3	13.42	0.0	0.00	10.84	0.00	0.00

Figure 3—Online Gateway to PTAEDA output example pa



	A	B	C	D	E	F	G	H	I	J	K
1	Current stand age (Age1)			20							
2	Index age 50 site index			....62.0							
3	Dominant height			.....							
4	Pulpwood threshold dbh			.....3.6							
5	Pulpwood top diameter limit			.....3.0							
6	Sawlog threshold dbh			.....9.6							
7	Sawlog top diameter limit			.....6.0							
8	Pulpwood volume unit			....Tons							
9	Sawlog volume unit			....Tons							
10	Residual BA at Age1			65							
11	Projection age (Age2)			25							
12	Residual BA at Age2			.....							
13	Projection age (Age3)			.....							
14	Residual BA at Age3			.....							
15	Projection age (Age4)			.....							
16											
17	Initial stand and stock table at age 20.										
18											
19	Dbh	All TPA	All BA	Ht	Pw TPA	Pw BA	Sl TPA	Sl BA	Std Pw	Top Pw	Sl
20	1	2.3	0.01	11.9							
21	2	10.0	0.23	29.9							
22	3	23.0	1.15	39.6							
23	4	36.5	3.23	44.5	33.6	3.03			0.94		
24	5	49.0	6.69	47.6	49.0	6.69			3.63		
25	6	55.6	10.87	49.7	55.6	10.87			7.25		
26	7	53.1	14.17	51.4	53.1	14.17			10.36		
27	8	42.8	14.99	52.7	42.8	14.99			11.62		
28	9	30.3	13.42	53.7	30.3	13.42			10.84		
29	10	17.8	9.81	54.6	1.5	0.77	16.3	9.04	0.63	0.64	6.90
30	11	8.8	5.82	55.2			8.8	5.82		0.29	4.67
31	12	3.5	2.77	55.8			3.5	2.77		0.10	2.30
32	13	1.6	1.49	56.3			1.6	1.49		0.04	1.27
33	Total	334.3	84.66	48.8	266.0	63.95	30.2	19.13	45.27	1.08	15.14

Figure 4—Downloaded Microsoft Excel© output from online output example page.

Farrar, R.M.; Matney, T.G. 1994. A dual growth simulator for natural even-aged stands of longleaf pine in the South's East Gulf region. Southern Journal of Applied Forestry. 18: 147-155.

ISO/IEC. (2014). *ISO International Standard ISO/IEC 14882:2014(E) – Programming Language C++.* Geneva, Switzerland: International Organization for Standardization (ISO).

Matney, T.G. 1996a. TProfile©. Starkville, MS: World Wide Heuristic Solutions.

Matney, T.G. 1996b. TVolume©. Starkville, MS: World Wide Heuristic Solutions.

Matney, T.G.; Farrar, R.M. 1992. A thinned/unthinned loblolly pine growth and yield simulator for planted cutover site-prepared land in the Mid-Gulf South. Southern Journal of Applied Forestry. 16: 70-75.

Matney, T.G.; Ledbetter, J.R.; Sullivan, A.D. 1987. Diameter distribution yield systems for unthinned cutover site-prepared slash pine plantations in Southern Mississippi. Southern Journal of Applied Forestry. 11: 32-36.

Microsoft. 2010a. Visual Studio 2010©. Redmond, WA: Microsoft Corporation.

Microsoft. 2010b. Excel©. Redmond, WA: Microsoft Corporation.

Schultz, E.B.; Iles, C.J.; Matney, T.G. [and others]. 2010. Stand-level growth and yield component models for red oak-sweetgum forests on mid-south minor stream bottoms. Southern Journal of Applied Forestry. 34: 161-175.

Growth and Development

Moderator:

Gordon Holley
Louisiana Tech University

HYDROLOGIC EFFECTS ON DIAMETER GROWTH PHENOLOGY FOR *CELTIS LAEVIGATA* AND *QUERCUS LYRATA* IN THE FLOODPLAIN OF THE LOWER WHITE RIVER, ARKANSAS

Scott T. Allen, J. Wesley Cochran, Ken W. Krauss, Richard F. Keim, and Sammy L. King[1]

Abstract—Bottomland hardwood (BLH) forests represent an extensive wetland system in the Mississippi Alluvial Valley and southeastern USA, and it is currently undergoing widespread transition in species composition. One such transition involves increased establishment of sugarberry (*Celtis laevigata*), and decreased establishment of overcup oak (*Quercus lyrata*). The ecological mechanisms that control this transition are not well understood. We measured monthly diameter growth with dendrometer bands on 86 sugarberry and 42 overcup oak trees at eight sites in the floodplain of the White River (AR, USA) with differing hydrologic regimes. For both species, growth attenuated earlier at drier sites compared to wetter sites. Overcup oak grew slightly longer through late August, suggesting its growth period extends across both wet and dry periods. In contrast, sugarberry growth rate decreased substantially by mid-July. While these results did not necessarily indicate a mechanism for increased prominence of sugarberry, they suggest sugarberry growing season does not as much coincide with the typically drier period of late summer and may be less affected by these conditions. Overcup oak grows later into the dry season and water table conditions during this period may determine if overcup oak benefits from this relatively extended growth period.

INTRODUCTION

Floodplain forests are valued highly for their ecological functions, which depend upon hydrologic conditions (Sparks 1995). Bottomland hardwood forests (BLH) of the southeast are commonly wetlands, defined by often having hypoxic soil conditions driven by frequently saturated or flooded soils (Patrick 1981). Connectivity to rivers and river flow rates largely control the timing of flooding and soil saturation in many BLH floodplain wetlands. Given the seasonality of river flows and climate, floodplain BLH forests typically transition between wet to dry states. In the BLH forests of the Lower Mississippi Alluvial Valley (LMAV) flooding usually occurs in winter and the early growing season, and soils become drier throughout the summer (Dewey and others 2006). This results in a complex stress regime typical of many non-tidal forested floodplains (Parolin and others 2010), in which saturated conditions transition to water limited conditions through the growing season.

Superimposed upon the natural hydrologic regime, anthropogenic developments have altered the hydrologic cycle leading to changes in BLH forests (King and others 2012). Few large rivers in the United States have natural flow regimes because levees, dams, and other engineering structures affect river flows and exchange with associated floodplains (Merritt and others 2010); these alterations affect tree growth. Studies have demonstrated relationships between BLH tree growth and hydrology (Anderson and Mitsch 2008, Megonigal and others 1997, Mitsch and others 1991), so large alterations in hydrology often affect growth patterns (Gee and others 2014, Reily and Johnson 1982).

Parallel to these changes, species composition has also changed in BLH, tending towards species associated with drier or wetter conditions (Gee and others 2014, Hanberry and others 2012). Multiple mechanisms may be responsible for this change in species composition. One likely mechanism is a reduced disturbance regime, whether from reduced flooding or reduced disturbance from other sources, such as fire and timber harvesting (Oliver and others 2005). Shade tolerant species are generally less flood tolerant (Battaglia and Sharitz 2006), and reduced flooding enables establishment of trees that would otherwise die from flooding (e.g.,

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sugarberry, *Celtis laevigata*; Conner and others 2002). Another possible mechanism is that the shift towards drier conditions creates more favorable growing conditions for some species compared to others. In this study, we investigate this latter hypothesis.

We examined timing of intra-annual diameter growth of wetter-association and drier-association BLH species (overcup oak, *Quercus lyrata* and sugarberry, respectively) with regard to floodplain hydrology. Dendrometer bands are useful for measuring fine-scale growth increments throughout the growing season to examine timing of growth and how growth responds to intra-annual variation in environmental conditions (Keeland and Sharitz 1993). This method was used to determine how growth varied throughout the growing season and how timing of growth was related to site hydrology and species.

METHODS

Study Site

The study was conducted at eight plots in the Dale Bumpers White River National Wildlife Refuge on the White River floodplain in southeastern Arkansas, USA. The White River is 1162 km long and drains a 7.2×10^4 km^2 area of northern Arkansas and southern Missouri. Within the refuge, the active floodplain ranges from 8-14 km wide. The floodplain is often highly connected to the river and becomes inundated seasonally. Flooding is generally extensive from December through June, peaking between March and early June. Conditions generally became dry July-November. The southern portion of the refuge experiences deeper flooding in spring because of backwater flooding from the Mississippi River, but channel incision and entrenchment (i.e., gradual deepening of the main channel with respect to floodplain) of the southern floodplain (Schumm and Spitz 1996) has led to greater depth to water table during the summer (fig. 1). The northern reach of the White River floodplain is less incised and more affected by river flows and headwater events during the summer (fig. 1). This study was conducted in 2014 when conditions were atypically wet in the northern section of the refuge and typical for the southern section of the refuge.

We established eight study plots with contrasting hydrology on these two reaches of the floodplain. Four sites were located in the northern section of the refuge, classified as 'Riverine Backwater' (Klimas and others 2009). Four plots were established in southern zone, also classified as 'Riverine Backwater,' although with a slightly drier vegetation composition (Klimas and others 2009).

Study Species

Overcup oak is a relatively slow growing oak species with shallow roots (often due to association with clayey soils in alluvial wetlands) that generally have later leaf-out than co-occurring species (Solomon 1990). Sugarberry is a shade-tolerant species that is also shallow rooted on wet sites (Kennedy 1990). In a review of species tolerances to water logging, Hook (1984) classified overcup oak as 'highly tolerant' and sugarberry as 'weakly tolerant.'

Measurements

In each plot, four subplots (0.04 ha) were established for assessing stand characteristics and selecting measurement trees. Species and diameter (DBH) were recorded for every tree with DBH > 5.0 cm.

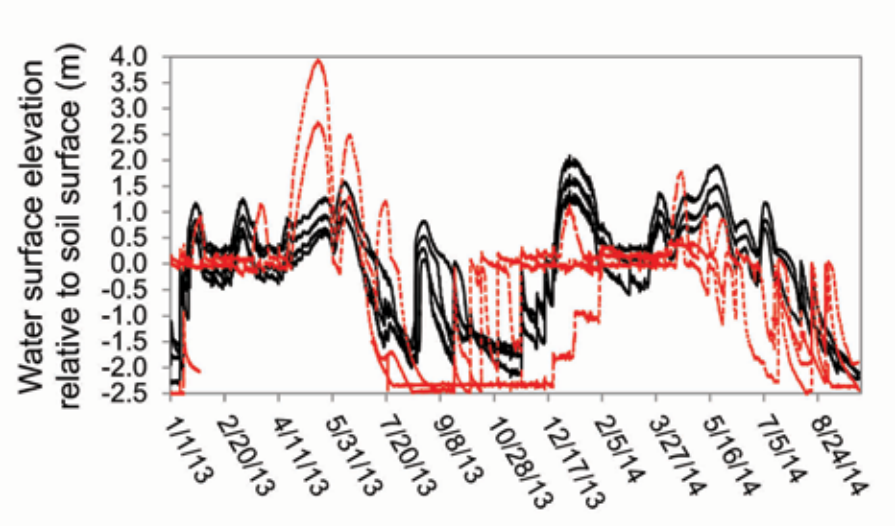


Figure 1—Water levels in plots on the White River floodplain for 2013 and 2014 (study year). The four black lines are for plots on the northern zone of the floodplain and the three red lines are wells in plots in the southern zone.

Stainless steel dendrometer bands (Hall 1944) were installed at 1.4 m height on 216 trees (70 overcup oak and 140 sugarberry). Sizes of banded trees varied considerably (table 1) to represent the range of trees found within the stands. Bands were installed in the summer of 2012 and allowed to settle for a year for accurate measurements (Bower and Blocker 1966, Keeland and Sharitz 1993). Bands were measured with vernier calipers to the nearest 0.1 mm in 2013 and 2014. Measurements were zeroed at the beginning of the 2014 growing season, and we report data from five increments throughout the 2014 growing season (June 25, July 13, August 6, August 23, October 1).

At each plot, a ~2.5 m deep well was installed for monitoring depth to water table with water level recorders (Hobo; Onset Computer Corp., Cape Cod MA, USA), logged at hourly intervals. Relative surface elevation among trees was surveyed with an autolevel (Robert Bosch LLC, Farmington Hills, MI), zeroed with respect to the ground level at the co-located monitoring well.

Analyses

Trees that did not show substantial growth (minimum 0.25 cm in circumference) were excluded because of poor data resolution, which left 42 overcup oak and 86 sugarberry trees remaining. Excluded trees were mostly small and suppressed. Tree diameter increment was compared between species by t-test and compared to diameter for each species by simple correlation. For each tree, data from the dendrometer bands were converted to cumulative fractional growth through the season to examine seasonal diameter growth curves; curve shape indicates timing but not magnitude of differences among trees (i.e., high values do not indicate more growth). Trees were partitioned by size as 'small' (< 20 cm for sugarberry, < 35 cm for overcup oak) versus 'large' (> 20 cm for sugarberry, > 35 cm for overcup oak) to determine size effects on growth curves. Different criteria were used for each species because of different size distributions. Location within plots was separated based on microtopography. The highest 50 percent of trees were classified as 'Higher' and the lower 50 percent as 'Lower' with the exception of one site that was classified entirely as 'Higher' because it had minimal microtopography and was the least flooded site. These stratifications of the data were used as treatments, with differences tested for each measurement date by 2-sample t-tests ($\alpha = 0.05$). We estimated the growing season started with day of year (DOY) 100, corresponding with the approximate leaf out date estimated from satellite imagery (MODIS phenology products, http://phenology.cr.usgs.gov/), although the actual leaf out date was likely later for overcup oak (Solomon 1990). All analyses were conducted in MATLAB (Mathworks Inc., Natick, MA).

RESULTS

Stand Characteristics

Despite different hydrogeomorphic settings, stand structure was similar across all study areas. All plots had multi-cohort, closed canopy forests (basal area of 30.4 ± 10.2 m^2 ha^{-1}; mean ± SD). Dominant species were *Celtis laevigata, Quercus lyrata, Carya aquatica, Liquidambar styraciflua, Fraxinus* spp., *Quercus texana,* and *Ulmus americana.* The study species, sugarberry and overcup oak, accounted for 17 and 27 percent of basal area, and 20 and 13 percent of all stems (605 ± 215 stems ha^{-1} for trees with DBH > 5 cm), respectively, together accounting for 16 to 92 percent of all stems per plot.

Table 1—Sample sizes (N) and tree sizes (diameter at breast height; DBH) and growth (diameter increment; DI) for *C. laevigata* and *Q. lyrata* trees with dendrometer measurements from the North (wetter) and South (drier) zones of the White River floodplain. Small, Med[ium] and Large refers to trees with DBH < 20 cm, 20-35 cm, and > 35 cm, respectively

		Celtis laevigata			*Quercus lyrata*		
Site	*Size*	*N*	*DBH (cm)*	*DI (cm yr^{-1})*	*N*	*DBH (cm)*	*DI (cm yr^{-1})*
North	Small	14	16.0 ± 1.0	0.6 ± 0.4	0	No Data	
North	Med	18	27.0 ± 0.9	0.6 ± 0.4	3	31.6 ± 5.2	0.7 ± 0.4
North	Large	5	43.3 ± 3.4	0.8 ± 0.5	12	54.3 ± 14.0	2.2 ± 1.3
South	Small	29	3.9 ± 4.2	0.6 ± 0.3	9	20.9 ± 2.4	1.1 ± 0.3
South	Med	14	26.1 ± 4.8	0.8 ± 0.5	3	28.2 ± 4.2	1.3 ± 0.7
South	Large	7	44.3 ± 5.7	0.8 ± 0.5	15	59.7 ± 12.9	1.5 ± 0.5

Annual Growth Increments

The magnitude of annual diameter increment and its relationship with size differed between species. The 2014 diameter increment for overcup oak exceeded sugarberry ($p < 0.0001$). For sugarberry, diameter increment was not related to tree diameter ($r = 0.11$, $p = 0.33$), but for overcup oak, diameter increment was positively correlated to tree size ($r = 0.36$, $p = 0.024$). These relationships held across both northern and southern plots (table 1).

Intra-annual Growth Trends

Dendrometer data indicated differences in growth trends between species and hydrogeomoprhic setting. Upon the first dendrometer measurements (DOY 176), 75 ± 17 percent (mean ± SD) of annual growth had occurred in sugarberry trees (fig. 2). In contrast, only 56 ± 18 percent of overcup oak growth had occurred by this time. Accordingly, a lower proportion of late-season growth occurred in sugarberry compared to overcup oak. Growth increments in overcup oak remained steady until DOY 235. Neither species had substantial growth in the last measurement period (DOY 235 to DOY 273). Growth trends were not size dependent for either species (fig. 2A), and showed minor, statistically insignificant effects of microtopographic position (fig. 2B). Stratifying the data by river reaches (north versus south) resulted in separation among means (fig. 2C). For multiple periods, t-tests indicated significant differences in cumulative fractional growth between the northern and southern sites for overcup oak (DOY 176, $p = 0.0082$; DOY 217, $p = 0.0027$) and sugarberry (DOY 176, $p = 0.03$; DOY 194, $p = 0.01$, DOY 217, $p = 0.01$). While our results may suggest that the northern site has a longer growing season, lack of precise north/south leaf-out dates prevents inferring total length.

DISCUSSION

Growth phenology varied by species and by site conditions. For both species, a greater proportion of growth occurred early in the season for the drier, southern sites (fig. 2C). In contrast, the wetter, northern sites (fig. 1) sustained growth later into the season. Phenological differences between sites appear to have been related to differences in hydrology. Greater connectivity between floodplain and river allows for flood waters to replenish soil water and maintain a higher water table. In contrast, the southern reach of the river lost connectivity with the floodplain and the water table lowered with summer recession. A study using sapflow probes to examine functioning of these two species within the same study plots in the refuge found agreement with this study; trees in one of the southern sites had signs of reduced function or early senescence that were not apparent in a northern site (Allen and others 2014).

Species differences in growth timing suggest differences in their response to environmental conditions, which may affect species interactions and community changes. While sugarberry has been classified as 'weakly flood tolerant' and overcup oak as 'highly flood tolerant', based on their typical zonation (Hook 1984), our results suggest that these classifications oversimplify their habits. Further testing is needed, but because the majority of sugarberry growth occurred in the early portion of the season, they showed tolerance to deep flooding in our study year. However, sugarberry grew less in late summer, coincident with drier conditions. Dormancy or a partial state of dormancy is a common means of avoiding water deficits but differs from tolerance because tolerance is generally associated with morphologic adaptations that allow function to be less affected during stressful conditions but otherwise would limit maximum photosynthesis (Lambers and others 2006). In contrast to sugarberry, overcup oak grew during both the early (flooded) and late portion of the growing season. This shift toward later season growth can be considered an adaptation of overcup oak to flooding in spring, and its flood tolerance may be viewed more appropriately as flood avoidance, which has been shown also for overcup oak rooting habits (Burke and Chambers 2003). For overcup oak, later season growth may lead to increased sensitivity to seasonal dynamics of water tables and a need for adaptation to multiple stressors (wet and dry), which is uncommon (Niinemets 2010). Confounding inferences on tolerance and avoidance inferences, there may be inherent differences in carbohydrate usage or storage, and growth does not necessarily reflect tree function or photosynthetic activity.

Similar to reduction in other disturbances (Oliver and others 2005), elimination of, or reduction in, flooding has been shown to favor sugarberry (Gee and others 2014, Hanberry and others 2012). This may be related to shade tolerance of sugarberry (Battaglia and Sharitz 2006), which also allows recruitment in the absence of flood disturbance (i.e., extended flooding kills sugarberry trees; Broadfoot and Williston 1973). Further, presence and success of saplings are not just limited by site conditions, but are also limited by seed production, dispersal, and fecundity (Clark and others 1999, Streng and others 1989). Our results suggest the greater shade tolerance of sugarberry (versus overcup oak) by the lack of size differences in diameter growth increment for sugarberry. For overcup oak trees, larger trees likely receive more sunlight and had greater diameter growth. While recruitment and disturbance regimes are certainly relevant to changing species composition, phenological avoidance of stress can be important for defining species interactions (Lovell and Menges 2013) and could be an additional mechanism that is advantageous to sugarberry in many settings, including novel

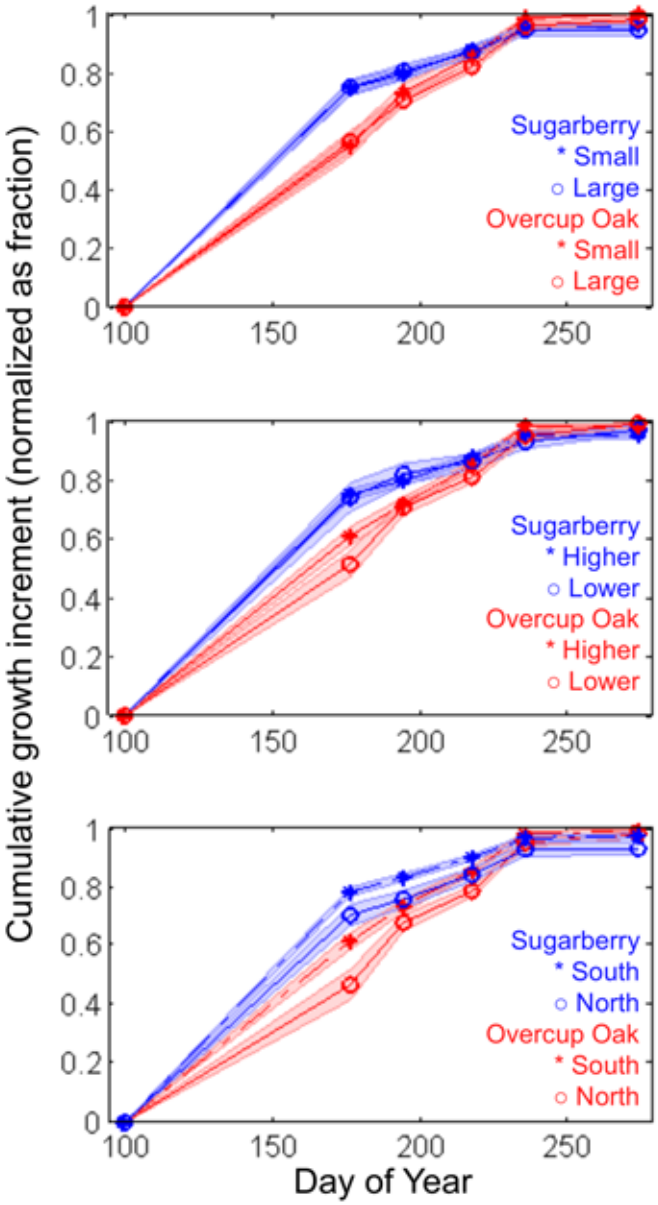


Figure 2—Cumulative fractionational growth curves derived from dendrometer band measurements of sugarberry *(Celtis laevigata)* and overcup oak (*Quercus lyrata*) trees on the White River floodplain from the 2014 growing season. Comparisons are of (a) tree diameters, (b) tree relative elevations within plots, and (c) plot location on the wetter, northern plots or drier, southern plots.

hydrogeomorphic conditions of BLH systems. Because sugarberry grew little during the late season in 2014, thereby demonstrating avoidance of the dry conditions, while overcup oak did not, this may be a relative benefit to sugarberry with respect to hydrologically altered BLH forests with drier conditions in late growing season. Inter-annual variations in sub-annual growth trends must be investigated to develop a fuller understanding of how phenotypic plasticity interacts with hydrologic variability at various time scales in these forests to control species composition.

CONCLUSIONS

Dendrometer data from 128 trees on the White River floodplain suggest that sugarberry grows rapidly in the early growing season while overcup oak grows more steadily and later into the growing season. For both, growth decreases earlier in drier sites. These phenological differences between species indicate potential differences in mechanism for response to stresses associated with transitioning from wet to dry conditions.

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LITERATURE CITED

Allen, S.T.; Cochran, J.W.; Edwards B.L. [and others]. 2014. Hydrologic controls over water use in a forested floodplain wetland [abstract]. In: American geophysical union , Fall Meeting, Dec. 19, 2014. San Francisco, CA: [Not paged].

Anderson, C.J.; Mitsch, W.J. 2008. Tree basal growth response to flooding in a bottomland hardwood forest in Central Ohio. Journal of the Water Resources Association. 44(6): 1512-1520.

Battaglia, L.L.; Sharitz, R.R. 2006. Responses of floodplain forest species to spatially condensed gradients: a test of the flood–shade tolerance tradeoff hypothesis. Oecologia. 147(1): 108-118.

Bower, D.R.; Blocker, W.W. 1966. Notes and observations: Accuracy of bands and tape for measuring diameter increments. Journal of Forestry. 64(1): 21-22.

Broadfoot, W.M.; Williston, H.L. 1973. Flooding effects on southern forests. Journal of Forestry. 71(9): 584-587.

Burke, M.K.; Chambers, J.L. 2003. Root dynamics in bottomland hardwood forests of the Southeastern United States Coastal Plain. Plant and Soil. 250: 141-153.

Conner, W.H.; Mihalia, I.; Wolfe, J. 2002. Tree community structure and changes from 1987 to 1999 in three Louisiana and three South Carolina forested wetlands. Wetlands. 22(1): 58-70.

Clark, J.S.; Beckage, B.; Camill, P. [and others]. 1999. Interpreting recruitment limitation in forests. American Journal of Botany. 86(1): 1-16.

Dewey, J.C.; Schoenholtz, S.H.; Shepard, J.P.; Messina, M.G. 2006. Issues related to wetland delineation of a Texas, USA bottomland hardwood forest. Wetlands. 26(2): 410-429.

Gee, H.K.W.; King, S.L.; Keim, R.F. 2014. Tree growth and recruitment in a leveed floodplain forest in the Mississippi River Alluvial Valley, USA. Forest Ecology and Management. 334: 85-95.

Hall, R.C. 1944. A vernier tree-growth band. Journal of Forestry. 42(10): 742-743.

Hanberry, B.B.; Kabrick, J.M.; He, H.S.; Palik, B.J. 2012. Historical trajectories and restoration strategies for the Mississippi River Alluvial Valley. Forest Ecology and Management. 280: 103-111.

Hook, D.D. 1984. Waterlogging tolerance of lowland tree species of the South. Southern Journal of Applied Forestry. 8(3): 136-149.

Keeland, B.D.; Sharitz, R.R. 1993. Accuracy of tree growth measurements using dendrometer bands. Canadian Journal of Forest Research. 23(11): 2454-2457.

Kennedy, H. 1990. *Celtis laeviga* Willd., Sugarberry. In: Burns, R.M.; Honkala, B.H., tech. cords. Silvics of North America: Vol. 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 258-261.

King, S.L.; Battaglia, L.L.; Hupp, C.R. [and others]. 2012. Floodplain wetlands of the Southeastern coastal plain. In: Batzer, D.; Baldwin, A.H., eds. Wetlands of North America: ecology and conservation concerns. Berkeley, CA: University of California Press: 253-266.

Klimas, C.; Murray, E.; Foti, T. [and others]. 2009. An ecosystem restoration model for the Mississippi Alluvial Valley based on geomorphology, soils, and hydrology. Wetlands. 29(2): 430-450.

Lambers, H., Chapin, F.S., and Pons, T.L. 2006. Plant physiological ecology. New York: Springer Publishing. 540 p.

Lovell, J.T., and Menges, E. 2013. Dominant species responses to drought in seasonal wetlands: evidence from reciprocal transplants across a moisture gradient. Journal of the Torrey Botanical Society. 140(2): 157-169.

Megonigal, J.P.; Conner, W.; Kroeger, S.; Sharitz, R.R. 1997. Aboveground production in southeastern floodplain forests: a test of the subsidy stress hypothesis. Ecology. 78(2): 370-384.

Merritt, D.M.; Scott, M.L.; Poff, L.N. [and others]. 2010. Theory, methods and tools for determining environmental flows for riparian vegetation: riparian vegetation-flow response guilds. Freshwater Biology. 55(1): 206-225.

Mitsch, W.J.; Taylor, J.R.; Benson, K.B. 1991. Estimating primary productivity of forested wetland communities in different hydrologic landscapes. Landscape Ecology. 5(2): 75-92.

Niinemets, Ü. 2010. Responses of forest trees to single and multiple environmental stresses from seedlings to mature Plants: past stress history, stress interactions, tolerance and acclimation. Forest Ecology and Management. 260(10): 1623-1639.

Oliver, C.D.; Burkhardt, E.C.; Skojac, D.A. 2005. Increasing scarcity of red oaks in Mississippi River floodplain forests: influence of the residual overstory. Forest Ecology and Management. 210(1-3): 393-414.

Parolin, P.; Lucas, C.; Piedade, M.T.F.; Wittmann, F. 2010. Drought responses of flood-tolerant trees in Amazonian floodplains. Annals of Botany. 105(1): 129-139.

Patrick, W.H. 1981. Bottomland soils. In: Clark, J.R.; Benforado, J., eds. Wetlands of bottomland hardwood forests. Proceedings of a workshop on bottomland hardwood forest wetlands of the Southeastern United States. Lake Lanier, GA: 177-185.

Reily, P.W.; Johnson, W.C. 1982. The effects of altered hydrologic regime on tree growth along the Missouri River in North Dakota. Canadian Journal of Botany. 60(11): 2410-2423.

Schumm, S.A.; Spitz, W.J. 1996. Geological influences on the lower Mississippi River and its alluvial valley. Engineering Geology. 45(1-4): 245-261.

Solomon, J.D. 1990. *Quercus lyrata* Walt, overcup oak. In: Burns, R.M.; Honkala, B.H., tech. cords. Silvics of North America: Volume 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 681-685.

Sparks, R.E. 1995. Need for ecosystem management of large rivers and their floodplains. BioScience. 45(3): 168-182.

Streng, D; Glitzenstein, J.S.; Harcombe, P.A. 1989. Woody seedling dynamics in an east Texas floodplain forest. Ecological Monographs. 59(2): 177-204.

QUALITY AND YIELD OF SEVEN FORAGES GROWN UNDER PARTIAL SHADING OF A SIMULATED SILVOPASTORAL SYSTEM IN EAST TEXAS

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Abstract—The goal of this project is to evaluate the growth and nutritional characteristics of seven forages, including various warm season native grasses, grown under simulated partial shading (50 percent typical of a loblolly pine silvopastoral system in east Texas. The results are from year two of a three year study. In order to meet the overall objective, individual, slatted shade structures were constructed that simulate the quantity, quality, and overall light regime found beneath loblolly pine stands arranged for silvopasture. The forages selected for the study include 'Tifton 9' bahiagrass (*Paspalum notatum*), 'Tifton 85' bermudagrass (*Cynodon dactylon*), 'Alamo' switchgrass (*Panicum virgatum),* 'Kaw' Big Bluestem (*Andropogon gerardii*), 'Americus' Indiangrass (*Sorghastrum nutans*), 'Harrison' Florida Paspalum (*Paspalum floridanum*), and Nacogdoches Eastern gamagrass (*Tripsacum dactyloides*). The experimental design is a two-way factorial design with forage type randomly assigned to plots, and shade treatment (0 percent, 50 percent) randomly assigned within forage type. Forage produced beneath the slats is managed to simulate intensive grazing, with recognition of minimum and optimal grazing heights based on forage type. Data is presented on dry matter yield, as well as several nutritional parameters including *in vitro* true digestibility (IVTD), crude protein (CP), neutral detergent fiber (NDF), and acid detergent fiber (ADF). Results show that significant differences existed in all parameters ($p < 0.0001$) due to forage type. Significant differences existed due to shade treatment for all parameters except for ADF ($p = 0.1324$). Results showed that shade improved forage quality overall. It reduced NDF ($p = 0.0399$), increased CP ($p = 0.0007$), and increased digestibility IVTD ($p < 0.0001$). The study is currently in year two of three. Results are preliminary.

INTRODUCTION

The number of people dependent on the world's resources has greatly increased (Brown and others 2011). With this increase in human population, there is an increased need for land, food, fiber, and energy that places strain on the world's resources. Production practices must be implemented that will utilize scarce resources in the most efficient way possible while insuring the long term productivity of the land. Agroforestry provides an alternative to traditional agricultural practices and has been shown to reduce nutrient runoff (Verchot and others 2007; Bambo, and others 2009a), increase production (Belsky, 1994; Jose, 2009), and provide more sustainable options for the production of food and fiber (Verchot, and others 2007; Jose, 2009, Aiyeloja and others 2011). Agroforestry combines trees, crops, and possibly grazing animals in a single land base, and provides long-term financial stability for producers while reducing the environmental impact seen in traditional agricultural systems (Jose, 2009). Agroforestry provides an opportunity for a producer to diversify production, allowing the producer to meet their current income needs while investing in a future harvest that will provide long-term economic stability (AFTA, 2000; Bambo and others 2009b; Jose, 2009).

One type of agroforestry, silvopasture, is an intensively managed system that combines trees with pasture and grazing. This system has been shown to increase nutritional quality of warm and cool season grass species (Burner and Brauer, 2003; Buergler and others 2005) as well as increase animal health by providing shelter from heat and cold (Buergler and others 2006; Karki and Goodman, 2010). This system provides an annual income to the producer in the form of cattle, while providing high quality sawtimber for harvest in the future (Grado, 2001). In addition, management of the tree crop to improve stem quality is easily accomplished due to the spacing of trees, while the forage yields of some species under partial shade (up to 50 percent remain similar to that of traditional fields in full sun (AFTA, 2000).

East Texas is unique in that it has high productive potential for a silvopasture production system. Warm season forages provide long grazing periods that last from approximately April through September,

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and sometimes extending into October. Rainfall is relatively high, which would potentially offset some of the moisture competition among the intercropped trees and forages. Also, species common to the area such as loblolly pine (*Pinus taeda*) have shallow lateral roots that allow stratification with the deeper roots of many warm season grasses. Cattle production can be sustained without supplemental feeding almost year round with the use of cool season forages, and there are many legumes available that can extend the grazing period and add nitrogen to the system through symbiosis and N_2 fixation. This also serves to provide additional protein to cattle and can enhance overall gains.

Other environmental factors exist that make this area especially favorable for silvopasture. These include sensitive watersheds that could benefit from the ecosystem services provided by this system of production. Also, marginal overall land productivity inherent to the region may be improved with a multiple production strategy.

There is a lack of site specific research that has been completed on forage quality in a silvopastoral operation in this region. Without data on the potential yields and quality from specific forages, it is unlikely that producers will be willing to move into this system of production. This project seeks to fill information gaps by determining the yield and quality of specific forages grown under the partially shaded environment of a simulated loblolly pine silvopasture system.

METHODS

Sites

Plots were located at the NRCS East Texas Plant Material Center in Nacogdoches Texas. Plots were installed on the Woden soil series (coarse-loamy, siliceous, semiactive, thermic Typic Paleudalfs) characterized by very deep, moderately-well drained, slightly acidic soils formed from alluvial sediments, and recently cleared of loblolly pine forest. Roundup ready soybeans were maintained as a cover crop on the site prior to the establishment of the grasses for this study.

Procedures

To meet the overall goal of this project, slatted lath shade structures were constructed that simulated the quantity, quality, and overall behavior of the light found under a loblolly pine canopy in a silvopasture setting. Shade slats were used for this study instead of cloth in order to most closely mimic light regimes under a loblolly pine silvopastoral canopy, while still allowing for controlled treatments. The size of the slat structures were sufficiently small to cover a single 1.22 m^2 plot, and to allow for overall easy mobility of the structures during forage management.

Individual plants were started in trays in a greenhouse in order to bypass the establishment periods of two to three years common to native grasses. Plugs were hardened off for a week in a shade house before field transplanting. Forage was planted in four rows of twelve plugs, three per linear foot. The total number of plugs in each plot is 48. The middle two square feet are considered the sample zone and forage is clipped at the appropriate height and bagged for analyses. The remaining one foot perimeter zone is a buffer zone that reduces edge effects. This portion of the plot is clipped at the same time as the sample portion but is discarded out of the plot area. Specifically, wooden laths were used to produce shade on a frame of Charlotte PVC pipe 1inch x 20 feet schedule 40 PVC plain end pipe (Lowes: Item #: 23975 | Model #: PVC 04010B 0800). The PVC pipe was cut into 1.22 m (4 foot) sections and attached into a cubical shape using LASCO 1inch Schedule 40 Side Outlet Elbows (Lowes: Item #: 315499 | Model #: 413010RMC). The top portion of the frame was adjustable, achieved by drilling holes in the top of the upper four elbows to create an opening through which the side legs could slide, so that the top could be maintained at an approximate height of 0.3 m (0.98foot) above the forage canopy (Varella and others 2011). The slats not only achieved the same light quantity found in this system (50 percent, but also allowed for recreation of the intermittent light characteristic of sub-canopy environments. The overall light behavior contributes to changes in the morphology of understory vegetation beyond differences in light quantity (Varella and others 2011). Light quality and quantity found beneath the slats were manually measured and compared to light under loblolly pine canopy cover using a FieldSpec® HandHeld Spectroradiometer (model FS HH 325-1075) by ASD, Inc (2555 55th Street, Suite 100 Boulder, CO 80301) with Full Sky Irradiance Reverse Cosine Receptor (RCR) attached. The total shading of 50 percent selected for this study is slightly above the known upper limits of acceptable shading for warm season grasses of approximately 45 percent (Lin and others 2001).

Plots were established using a 7x2, factorial design (seven forages and two shade levels, 0 and 50 percent and data were analyzed with a Model I ANOVA using SAS 9.3 for Windows (SAS Institute, Inc.). The factors were fixed and included seven forages and two light treatments (full sun, 50 percent full sun). All factors were completely randomly assigned to treatment plots. The forages selected for the study include 'Tifton 9' bahiagrass (*Paspalum notatum*), 'Tifton 85' bermudagrass (*Cynodon dactylon*), 'Alamo' switchgrass (*Panicum virgatum),* 'Kaw' Big Bluestem (*Andropogon gerardii*), 'Americus' Indiangrass (*Sorghastrum nutans*), 'Harrison' Florida Paspalum (*Paspalum floridanum*), and Nacogdoches Eastern gamagrass (*Tripsacum dactyloides*).

Shade slats were constructed during the winter 2013 and erected during spring emergence. Fifteen collections were conducted in year one, and 24 collections were completed in year two. Each plot was not cut at each collection, because plots were managed individually. Plots were harvested when heights reached 55-60 cm down to a minimum height of 20 cm for natives, and 10 cm for introduced grasses, to simulate intensive grazing. During the peak growing season plots typically reached height and were harvested every one and a half to two weeks. A total of 539 samples were collected during the 2013 growing season, and at least 800 samples were harvested in season two. Preliminary results for year one were presented at the ASSA, CSSA, and SSSA 2013 annual meetings in Tampa, Florida November 3-6 (Hill, J.E., K.W. Farrish, B. Oswald, J.L. Young and A. Shadow. 2013. Quality and yield of seven forages grown under partial shading of simulated silvopastoral production system in east Texas. Poster presentation.); preliminary results for year two were presented at the ASSA, CSSA, SSSA 2014 annual meetings in Long Beach, California November 2-5 (Hill, J.E., K.W. Farrish, J.L. Young, B. Oswald, and A. Shadow. 2014. Quality and yield of seven forages grown under partial shading of a simulated silvopasture system in east Texas. Oral presentation)

Plant heights were recorded, and then samples were harvested and ground to 1 mm. Samples were analyzed for dry matter yield (DMY), crude protein (CP), neutral detergent fiber (NDF), acid detergent fiber (ADF), and *in vitro* true digestibility (IVTD) (NIRS only). Chemical analyses were conducted at Stephen F. Austin State University Soil, Plant, and Water Analysis Laboratory in Nacogdoches, Texas and included the following: *in vitro* true digestibility (IVTD) via near infrared reflectance spectroscopy (NIRS), CP (NIRS), and neutral detergent fiber (NDF) and acid detergent fiber (ADF) content using Van Soest's detergent method. Soil samples were collected from the research plots and mineral content analyzed in order to determine and define nutritional status of the soil, and for the purpose of fertilizer application. Other measurements included soil temperature and moisture content.

RESULTS AND DISCUSSION

For the establishment year (2013), data suggested little or no shade effect on any of the above constituents or properties ($p = 0.1305$-0.989) except for height ($p < 0.0001$). Shaded plants were taller (mean = 5cm) except for gamagrass (*Tripsacum dactyloides*) and switchgrass (*Panicum virgatum*), where there were no significant differences ($p = 0.5687$ and 0.0593 respectively).

There were no significant differences in yield based on shade treatment ($p = 0.9463$) in year one. However, there was a trend for shaded plants to exhibit lower yields except for Tifton 9 and big bluestem, which were unaffected. There were differences in yield due to forage type ($p<0.0001$); Tifton 9 bahiagrass and Florida Paspalum showed the highest yields regardless of shade treatment. Table 1 shows yields for growing season two.

Overall, year two was much more productive than the establishment year. This is not surprising because native grasses are known to frequently require lengthy establishment periods of up to two or three years. Results show that significant differences existed in all parameters ($p < 0.0001$) due to forage type. Significant differences existed due to shade treatment for all parameters except for ADF ($p = 0.1324$). Results showed that shade improved forage quality overall (table 2). It reduced NDF ($p = 0.0399$), increased CP ($p = 0.0007$), and increased digestibility IVTD ($p < 0.0001$). As a result, TDN which is calculated from NDF, ADF, and CP was also significantly increased under shade ($p = 0.0241$).

Table 1—Comparison of yields in Mg ha^{-1} for sunny versus 50% shaded plots. Each of the forage types in the study are compared side by side and results include only growing season two

	Sunny	Shaded
	(Mg ha$^{-1)}$	
Kaw' big bluestem	7.30	4.70
Nacogdoches' eastern gamagrass	10.10	8.70
Americus' indiangrass	9.80	7.20
Harrison' Florida paspalum	7.60	5.30
Alamo' switchgrass	8.20	5.20
Tifton 85 bermudagrass	5.30	3.30
Tifton 9 bahiagrass	14.50	11.00

Table 2—Results of shade treatment for growing season 2014 showing improved overall quality of forage. All parameters were analyzed at α = 0.005 and all were significant except for ADF

Parameter	Sunny	Shade	P- value
Dry matter yield (Mg ha^{-1})	9.1	6.5	0.0021
Crude protein (g kg^{-1})	14.39	15.65	0.0007
Neutral detergent fiber (g kg^{-1})	65.74	64.76	0.0399
Acid detergent fiber (g kg^{-1})	35.16	35.57	(0.1324)
In vitro true digestibility (g kg^{-1})	65.65	67.31	< 0.0001
Soil moisture (kg HOH kg^{-1} soil)	0.0388	0.0497	0.0222
Soil temperature (C°)	34.6	32.7	0.0084

Tukey and SNK means separations tests were performed on all parameters that showed significant differences. The results of each of these analyses consistently show that switchgrass, Florida paspalum, and Eastern gamagrass are the "best" forages in year two. Furthermore, yields were only slightly reduced in Florida paspalum and Eastern gamagrass by shade, and reduced by approximately half for switchgrass. However, switchgrass has shown high yields in both sun and shade, as well as persistence under intensive defoliation. Eastern gamagrass appears to exhibit some shade tolerance, but persistence is questionable based on visual observation of plots and is likely due to excessive defoliation or to close planting. Florida paspalum also has exhibited some decline, and year three will be a good indicator of the level of decline that can be expected.

Other analyses described in the objectives that have been completed include analyses of soil temperature and moisture differences between shaded and non-shaded plots. No differences in soil moisture were detected in year one, but significant increases in moisture under shade were detected in all plots in year two. Optimal soil temperature (31°C) for root growth was exceeded throughout the study, but shade treatment significantly reduced ($p < 0.0001$) plot temperatures by about 2 C° on plots with forage growth. Temperature reduction was over 4 C° ($p= 0.0058$) for bare mineral plots.

The forages that performed well, but appear to be declining include 'Nacogdoches' Eastern gamagrass (*Tripsacum dactyloides*) and 'Harrison' Florida paspalum (*Paspalum floridanum*). This is possibly due to close planting instead of defoliation or shade because native grasses are typically planted one plant per square foot in pastures whereas there are three plants per linear foot in the current study.

The forages that have performed well *and* have persisted well at end of season 2 include 'Alamo' switchgrass (*Panicum virgatum*) and Tifton 9 bahiagrass (*Paspalum notatum*). It is important to note that big bluestem appears to be improving over time, a trend that was not apparent until the end of season two. Year three results will be included in future papers.

Several warm season grasses, including native grasses, may be productive inputs to a silvopasture system; however, certain recommendations such as grazing intensity (grazing height and frequency) need to be evaluated more closely. The application of shade has positively affected the forage quality in this study through the end of season two. Dry matter yields were lower but the overall possible beef cattle gains per individual animal were higher when estimated using Foragval (TAMU), a software calculator used to estimate beef cattle gains based on forage quality parameters.

In conclusion, intake, digestibility, and metabolism determine quality of forage, and all three of these factors were improved under shade at the end of season two of three. Micro-environments under partial shade were cooler and moister which improves nutrient cycling, plant growth, beef cattle gains, and soil quality. Based on the preliminary results of this study, it is recommended that further research be completed on switchgrass, gamagrass, and big bluestem. Although the bahiagrass is the most productive in this study, it is avoided by many producers because it tends to out-compete adjacent, higher quality fields. Bahiagrass is known to quickly lose quality with maturity, and to produce seed heads early before harvestable yields have been achieved. This makes it difficult for producers who can either harvest early and get high quality and low yields, or harvest later and get higher yields of low quality forage. Further studies on the top native grass performers from this group should be conducted that

include a grazing trial and various defoliation levels. It is recommended that further studies focus on Eastern gamagrass and Florida paspalum, but with lighter grazing intensities. Also, it appears big bluestem is increasing in yields and quality, and seems to be relatively unaffected by shade. No data is available at this time but will be presented after the current and third growing season.

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LITERATURE CITED

Association for Temperate Agroforestry (AFTA). 2000. Agroforestry in the United States. http://www.aftaweb.org/resources1.php?page=32. [Date accessed: September 21, 2011].

Oyebade, B.A.; Aiyeloja, A.A.; Ekeke, B.A. 2010. Sustainable agroforestry potentials and climate change mitigation. Advances in Environmental Biology. 4(1): 58-63.

Bambo, S.K.; Nowak, J.; Blount, A.R [and others]. 2009a. Soil nitrate leaching in silvopastures compared with open pasture and pine plantation. Journal of Environmental Quality. 38(5): 1870-1877.

Bambo, S.K.; Blount, A.R.; Nowak, J. [and others]. 2009b. Effects of annual cool-season forage on Bahiagrass production in silvopastures. Journal of Sustainable Agriculture. 33(4): 418-434.

Belsky, A.J. 1994. Influences of trees on savannah productivity: tests of shade, nutrients, and tree-grass competition. Ecology. 75(4): 922-932.

Brown, J.H.; Burnside, W.R.; Davidson, A.D. [and others]. 2011. Energetic limits to economic growth. BioScience. 61(1): 19-26.

Buergler, A.L.; Fike, J.H.; Burger, J.A. [and others]. 2005. Botanical composition and forage production in an emulated silvopasture. Agronomy Journal. 97: 1141-1147.

Buergler, A.L.; Fike, J.H.; Burger, J.A. [and others]. 2006. Forage nutritive value in an emulated silvopasture. Agronomy Journal. 98: 1265-1273.

Burner, D.M.; Brauer, D.K. 2003. Herbage response to spacing of loblolly pine trees in a minimal management silvopasture in southeastern USA. Agroforestry Systems. 57: 69-77.

Grado, S.C.; Hovermale, C.H.; St. Louis, D.G. 2001. A financial analysis of a silvopasture system in southern Mississippi. Agroforestry Systems. 53: 313-322.

Jose, S. 2009. Agroforestry for ecosystem services and environmental benefits: an overview. Agroforestry Systems. 76(1): 1-10.

Karki, U.; Goodman, M.S. 2010. Cattle distributution and behavior in southern-pine silvopasture versus open-pasture. Agroforestry Systems. 78: 159-168.

Lin, C.H.; McGraw, R.L.; George, M.F.; Garrett, H.E. 2001. Nutritive quality and morphological development under partial shade of some forage species with agroforestry potential. Agroforestry Systems. 53: 269-281.

Varella, A.C.; Moot, D.J.; Pollock, K.M. [and others]. 2011. Do light and alfalfa responses to cloth and slatted shade represent those measured under an agroforestry system? Agroforestry Systems. 81: 157-173.

Verchot, L.V.; Van Noorwijk, M.; Kandji, S. [and others]. 2007. Climate change: linking adaptation and mitigation through agroforestry. Mitigation and Adaptation Strategies for Global Change. 12: 901-918.

THE SILVICULTURE OF SILVOPASTURE

Rebecca J. Barlow, Seth Hunt, and John S. Kush[1]

Abstract—Silvopasture is an agroforestry practice where livestock, forage, and timber are managed on the same parcel of land. The most common form of agroforestry in the Southeastern US is silvopasture. According to the most recent USDA Census of Agriculture, six of the top ten states in the Nation that report that they practice alley cropping or silvopasture are southern. Southern pines are well suited for use in silvopasture systems because of their relative ease of establishment and shorter rotation length. Considerable work has been completed to examine the suitability of southern pines such as loblolly and slash for use in agroforestry systems; however few studies have included longleaf pine. Few studies have addressed the need for modified silvicultural management in these systems. Established in 2008, twelve (four loblolly and eight longleaf), approximately 6 acre double row silvopasture demonstration sites were established on an old field site on the EA Hauss Demonstration forest near Atmore, Alabama. Since establishment, blocks were sampled five times for seedling survival, trees per acre and seedling/tree height, growth, and form. Results from this study have shown that early management practices including the use of prescribed fire can benefit the growth and form of longleaf pine silvopastures in this area. Also, the timing of pruning of both longleaf and loblolly silvopastures is addressed. Through this work we are better able to understand what is needed for the successful establishment and growth of longleaf and loblolly pine silvopastures in Alabama.

INTRODUCTION

Silvopasture is an agroforestry practice where livestock, forage, and timber are managed on the same parcel of land. It is the most common form of agroforestry in the Southeastern US, yet few landowners choose it as a land management option. Why is this? In recent memory, the industrial forest management style has dominated southern forestry. Therefore, professional land managers may hesitate to recommend practices that seem outside the norm. Landowners who are unfamiliar with agroforestry practices hesitate to adopt them for fear of making a costly mistake. However, most private landowners have varied objectives: high value products from their land, both traditional and non-traditional; wildlife; water; aesthetics; and more. Currently, timber markets are not what they were making timber management less attractive to many landowners. History has shown that woodland grazing can work in southern forests. The science of silvopasture can help landowners meet financial goals. But landowners and land managers need to also understand the silviculture of silvopasture to ensure that this land management model is its most productive.

HISTORY

Southeastern United States longleaf-slash pine forests made up much of the historical range for southern cattle grazing. At the turn of the last century, Piney-woods cattle were managed on open rangeland at a rate of about 5-10 acres per head depending on time of year and forage type. Wahlenberg (1946) noted “In accordance with age-old custom, southern landowners usually tolerate grazing on their forest lands by the livestock of numerous small famers. The typical forest range is open, no permits are required, no fees are charged, and usually no attempt is made to control fires set by stock owners”.

A cultural shift occurred that contributed greatly to an ecological change in the Southeast. Fencing laws were enforced in the 1930’s and 40’s. Longleaf pine forests were harvested and converted to pasture or agricultural land. In addition, the country was going through a period of fire suppression. Lack of fire on the landscape caused fallow agricultural land to be reforested with faster growing southern pine species such as loblolly and slash.

The U.S. Forest Service has a history of conducting research on native forage and woodland grazing (Campbell 1946, Lemon 1949), shaping many of the modern agroforestry techniques that take us beyond free-ranging cattle to science-based land management systems (Lewis and others 1985, Grado and Husak 2004, Oswald and others 2008, Cubbage and others 2012). Silvopasture is the agroforestry

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practice of combining livestock and trees on the same management area with the goals of optimizing timber, livestock, and forage components; improve cash flow though annual income from forage and livestock; and long-term income from timber (National Agroforestry Center 2008). So, why not consider silvopasture? This paper will provide an overview of agroforestry in the South and discuss management considerations if silvopasture is going to be considered. Data from an ongoing silvopasture study in south Alabama will be used to elaborate on concerns with silvopasture.

SILVOPASTURE DEMONSTRATION

Few studies have addressed modified silvicultural management in southern pine silvopasture systems but instead address other key issues such as tree and forage establishment, or economic returns (Clason 1999, Grado and others 2001). In an effort to examine the silviculture of silvopasture, a loblolly and longleaf silvopasture demonstration was established on the EA Hauss Demonstration Forest. Owned and managed by the Alabama Forestry Commission, The EA Hauss Demonstration forest is located in Escambia County, AL, north of Atmore, AL. In 2008, a portion of this former tree nursery site was set up as a double row silvopasture demonstration. Six-acre old-field sites (blocks) were planted, four blocks with loblolly and eight blocks with longleaf pine. Measurements were taken at the time of establishment in 2008 and on the odd years thereafter.

The trees were planted at a spacing of 6 feet (within the rows) by 8 feet (between the rows) by 40 feet between the double row sets. This spacing resulted in approximately 300 trees per acre planted in 2008. In early 2009, some replanting occurred to replace dead tree. In the autumn of 2009, “alleys” between the double rows within each block were site prepped for forage establishment. Six blocks (2 loblolly and 4 longleaf) were planted in exotic pasture grasses and the other 6 blocks (2 loblolly and 4 longleaf) were planted in native warm season grasses in spring 2010. Half of the longleaf pine blocks were treated with prescribed fire in the spring of 2013.

2015 Measurements

A subset of the blocks was measured in early 2015. Overall survival was 81percent for loblolly pine and 61percent for longleaf. Loblolly pine density averaged 265 trees/acre with a basal area of 46.5 square feet/acre. The average height was 27 feet. The numbers were lower for longleaf pine. Its density was 182 trees/acre with a basal area of 17.5 square feet/acre. The average tree height was 22 feet. An interesting note about longleaf heights, those blocks that were prescribed burned in 2013 were 1-foot taller than those on the unburned blocks.

SILVOPASTURE SILVICULTURE

The demonstration at Hauss has helped to provide some early rotation silviculture considerations. Scalping helps remove competition and is needed especially in old field plantings of longleaf pine. It is beneficial to herbicide rows to treat pasture grasses prior to planting. Keeping the ecology of longleaf pine in mind, prescribed fire early in longleaf plantings appears to help height growth. The USDA National Agroforestry Center (2008) recommends pruning trees when they are 15 to 20 feet tall and/or the diameter of the tree reaches 5-inches as a height of 6-inches above the ground. Thus, around ages 5-7 years, monitor for early pruning needs and consider stand stocking to schedule pre-commercial thinning, especially for loblolly pine. Unfortunately, trees at Hauss have not yet been pruned or thinned due to unforeseen limited access to contract labor. The resourceful landowner can conduct some silvicultural treatments such as pruning themselves, but future access to labor is something that should be considered during the planning process.

It is also important to remember, that young trees may be browsed or trampled by livestock. If the choice of livestock is cattle, it is beneficial to delay their introduction until the trees are 10 – 15 feet tall. Based on these heights, the loblolly pines at Hauss were considered “cattle resistant” by age 5. It took another year for the longleaf pine to reach that threshold.

CONCLUSION

The time is right for a renaissance in southern pine agroforestry. Grazing in pine forests is an important part of southern land management history. As part of a silvopasture, cattle are a source of additional income and cash flow that can be important to a private landowner. Although it may not have the rapid early growth of loblolly pine, longleaf should not be dismissed as viable options for landowners interested in silvopasture. Pine straw produced from longleaf pine silvopasture can produce additional revenue in addition to timber and cattle.

It is a must to always consider objectives when planning a new forestry activity and planting the tree species that best meets those goals. It is important that landowners and land managers understand the silvics of southern pine species. Differences may impact their land management decisions such as when to introduce cattle, or amount and quality of pine straw produced.

Remember to weigh the options! Your objectives will determine what management approach is most appropriate. A silvopasture system is not a “plant-it and leave-it” system. It is important to consider the economics and the ability to complete future management activities. More demonstrations are

needed to provide landowners with a perspective on this wonderful opportunity.

LITERATURE CITED

Campbell, R.S. 1946. Determination of grazing values of native vegetation on southern pine forest ranges. Ecology. 27(3): 195-204.

Clason, T.R. 1999. Silvopastoral practices sustain timber and forage production in commercial loblolly pine plantations of northwest Louisiana, USA. Agroforestry Systems. 44: 293-303.

Cubbage, F.; Glenn, V.; Mueller, J.P. [and others]. 2012. Early tree growth, crop yields and estimated returns for an agroforestry trial in Goldsboro, North Carolina. Agroforestry Systems. 86: 323-334.

Grado, S.C., Hovermale, C. H., St. Louis, D. G. 2001. A financial analysis of a silvopasture system in southern Mississippi. Agroforestry Systems. 53: 313-322.

Grado, S.C., Husak, A.L. 2004. Economic analysis of a sustainable agroforestry system in the southeastern United States. Valuing Agroforestry Systems. In: Alavalapati, J.R.R.; Mercer, D.E. eds. Valuing agroforestry systems. Netherlands: Kluwer Academic Publishers: 39-57.

Lemon, P.C. 1949. Successional responses of herbs in the longleaf-slash pine forest after fire. Ecology. 30(2): 135-145.

Lewis, C.E.; Tanner, G.W.; Terry, W.S. 1985. Double vs. single-row pine plantations for wood and forage production. Southern Journal of Applied Forestry. 9(1): 55-61.

Oswald, B.P.; Farrish, K.W.; Beierle, M.J. 2008. Survival of longleaf and loblolly pines planted at two spacings in an East Texas bahiagrass silvopasture. Southern Journal of Applied Forestry. 32(1): 44-45.

USDA National Agroforestry Center. 2008. Silvopasture: Establishment & management principles for pine forests in the southeastern United States. Lincoln, NE: U.S. Department of Agriculture National Agroforestry Center. 71 p.

Wahlenberg, W.G. 1946. Longleaf pine: Its use, ecology, regeneration, protection, growth and management. Washington, DC: Charles Lathrop Pack Forestry Foundation, in cooperation with the U.S. Department of Agriculture Forest Service. 429 p.

EFFECTS OF CULTURAL INTESITY AND DENSITY REGIME TREATMENT ON POST-THINNING LOBLOLLY PINE INDIVIDUAL TREE DBH INCREMENT IN THE LOWER COASTAL PLAIN OF THE SOUTHEASTERN UNITED STATES

John T. Perren, Michael Kane, Dehai Zhao, and Richard Daniels[1]

Abstract—Thinning is a well understood concept used to manage density dependent factors at the stand level. This study evaluates the effect of planting density, cultural intensity, and thinning treatment on loblolly pine post-thinning individual tree development. The Lower Coastal Plain Culture/Density Study, has four initial densities, in combination with two cultural treatments, that were thinned at age 12 to the current density on the lowest initial density. Analysis of data six years post-thin for the Lower Coastal Plain focuses on the effects of cultural intensity and density regime effects alone or in combination on individual tree development in DBH. We evaluated thinning intensity effects on individual tree development. Results indicate that initial density has the largest effect on the development of diameter growth following the thinning with trees from higher densities having a higher greater response in diameter increment.

INTRODUCTION

The southeastern United States produces a large volume of commercially grown timber that comes from plantations. The most dominant species in the region is loblolly pine (*Pinus taeda*). In an attempt to maximize returns on investment, many intensive management practices have been implemented to increase the production on any given site. One of the first considerations is planting density. Initial planting density has a strong effect on diameter growth. As planting density increases, mean diameter of individual stems declines (Zhao and others 2011). Another consideration for initial planting density is the desired products from the stand, with high densities being more appropriate for lower value product classes (Ameteis and Burkhart 2012). Fertilization and weed control have both been shown to have large impacts on the productive capacity of a stand. Nutrients are limiting in most southern timberland, with nitrogen and phosphorous being of primary concern. Fertilization and the removal of competing vegetation to increase the availability of nutrients to crop trees has been shown to more than justify the price of application in most markets (Fortson and others 1996; Fox and others 2007). Thinning is often used to maintain vigorously growing stands. When stands reach high levels of stocking, there is more competition for a fixed amount of resources. Competition leads to a higher mortality rate in stands with a greater stocking, as compared to those with lower levels (Hennessey and others 2004). Thinning is often used in conjunction with other silvicultural treatments. Many studies have looked at thinning as a component of intensive management. A study in the Western Gulf region showed that with thinning alone, 50 percent of the trees were classified as chip-and -saw by age 17. When fertilization was added to the thinning regime, greater than 70 percent of the trees were classified as chip-and-saw (Sword Sayer and others 2004).

The Plantation Management Resource Cooperative (PMRC) established the Culture Density Study to evaluate the effect of initial density and cultural intensity on stand development. At age 12, a subset of four installations from the original study were thinned to further evaluate the relationship between density management and stand development. A stand level view of these installations was completed by the PMRC, but analysis at the individual tree scale is needed to better explain the relationship between thinning and stem. Understanding individual tree response to treatment will lead to better prescriptions at the stand level.

METHODS

The data used in this analysis is a subset from the PMRC's Lower Coastal Plain Culture Density Study that was established in 1996. There were four installations

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with thinning across the Lower Coastal Plain. Two installations in South Carolina, one in Georgia, and one in Florida gave a good geographic distribution of site locations. These sites have six year post thinning data, which should be a sufficient amount of time to express the effects of thinning. Each installation has four initial densities of interest: 300, 600, 900, and 1200 trees per acre. Each of these densities has two cultural treatments (operational and intensive). The operational plots received bedding, banded chemical site preparation, and first year banded herbaceous weed control (table 1). These plots also received fertilization at planting, year eight and year twelve. The intensive culture plots included the same treatments as the operational plots with the addition of tip moth control, complete vegetation control, and fertilization in year three and even aged years starting in the fourth year.

Thinning was imposed on the 600, 900, 1200 TPA plots under operational and intensive culture at age 12. Plots were thinned to match age 12 TPA on the corresponding 300 TPA-planting density plots, which provided the non-thinned counterpart. Thinning consisted of third row removal with selection. Thinning to a standard number of trees per acre resulted in different levels of residual basal area and percent of the initial stand removed (table 2). Due to different levels of residual basal area left in each of the initial planting density and management combinations, the initial planting densities were referred to as density regimes after thinning.

Data collection

Data was collected by the PMRC field crew. The PMRC crew re-measured each installation periodically with no longer than two years between measurements through

Table 1—Silvicultural treatments for each cultural regime

Operational Regime	Intensive Regime
Bedding	Bedding
Fall banded chemical site preparation	Fall broadcast chemical site preparation
	Tip moth control
Herbaceous weed control: 1st yr. banded	Repeated herbicide application to achieve complete vegetation control
Fertilization: at planting, 500 lbs./acre of 10-10-10; spring of 8th, 12th, and 16th growing season, 200 lbs./acre of N and 25 lbs./acre of P	Fertilization: at planting, 500 lbs./acre of 10-10-10; spring 3rd growing season, 600 lbs./acre 10-10-10 + micronutrients + 117 lbs./acre NH4NO3; spring 4th growing season 117 lbs./acre NH4NO3; spring 6th growing season 300 lbs./acreNH4NO3; spring 8th, 10th,12th,14th , 16th and 18th growing season 200 lbs./acre of N and 25 lbs./acre of P

Table 2—Mean basal area removal by management and density regime across four location

Management	Density Regime	BA Age 12 Pre-Thin (ft^2/acre)	BA Age 12 Post-Thin (ft^2/acre)	BA % Removal
Intensive	300 (No-thin)	133.3	133.3	No Thin
	600 (Thin)	173.5	86.7	50%
	900 (Thin)	167.6	67.6	60%
	1200 (Thin)	196.8	63.8	68%
Operational	300 (No-thin)	115.8	115.8	No Thin
	600 (Thin)	144.0	79.3	45%
	900 (Thin)	159.8	68.4	57%
	1200 (Thin)	170.1	64.7	62%

age 18. Standard measurements, such as diameter at breast height (DBH), were taken. Measurement plots varied in number of trees per plot by initial density and were surrounded by a treated buffer.

Analysis

This analysis focused on the DBH and DBH growth in relation to the cultural intensity and density regimes. For the analysis, trees that had died prior to age eighteen were omitted from the data set. Analysis of variance was used to determine if there were statistically significance differences across density regimes, cultural intensity, and their combinations. To address and illustrate the effect of thinning on the individual density regimes and management combinations, linear regression was used to compare the growth increment in DBH over the post-thinning period to the initial DBH at the time of thinning. The data was divided by density regime and cultural regime (management). Each combination was an aggregate of the data from all four installations. DBH increment for each tree was calculated using the age 12 measurements and post thin measurements at age 18.

RESULTS

Management intensity, density regime and their interaction showed a significant effect on both age twelve and eighteen DBH (table 3). More intensive management resulted in increased mean DBH. For density regimes, mean DBH decreased with increasing density. Both of these results follow the trends reported by Zhao and others (2011) when evaluating all of the sites from the culture density study in the Lower Coastal Plain. Though the treatment effects on DBH further validated the earlier studies, the main focus of this analysis was the effects of the different treatments on DBH increment. DBH increment showed a different trend than the age twelve and eighteen year DBH results. DBH increment increased with increasing initial density. The effect of management was not significant on DBH increment (table 3). Density regime effects and the interaction of management and density regime on DBH increment were significant. The different density regimes showed the same pattern of growth response in DBH within each management intensity (fig. 1). For both the operational and intensive treatments, larger more dominant trees outgrew smaller trees and the higher initial planting density-thinned regimes had a higher level of growth response in DBH. When comparing the differences in DBH growth for a given density regime by management intensity the results vary. For both the 300 no-thin and 600 thin density regime the operational management treatment produced greater DBH increment than the intensive across the entire pre-thin age twelve DBH range (fig. 2). A different relationship was observed with the 900 and 1200-thin density

Table 3—Effects of management, density, and their interaction on DBH age 12, age 18 and DBH increment across four locations

Source	DBH Age 12	DBH Age 18	DBH Increment
Management	**	**	NS
Density Regime	**	**	**
Man x Den Reg	*	**	*

(NS= not significant,*P< 0.05, **P<0.01)

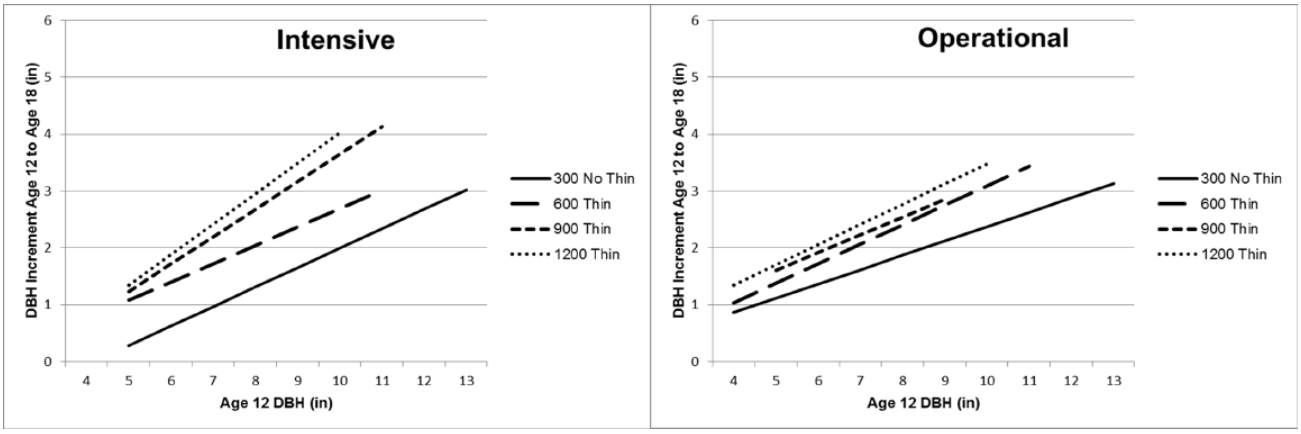


Figure 1—Relationship between post-thin DBH increment (age 12 to 18) and age 12 diameter distribution by density regime for operational and intensive treatments.

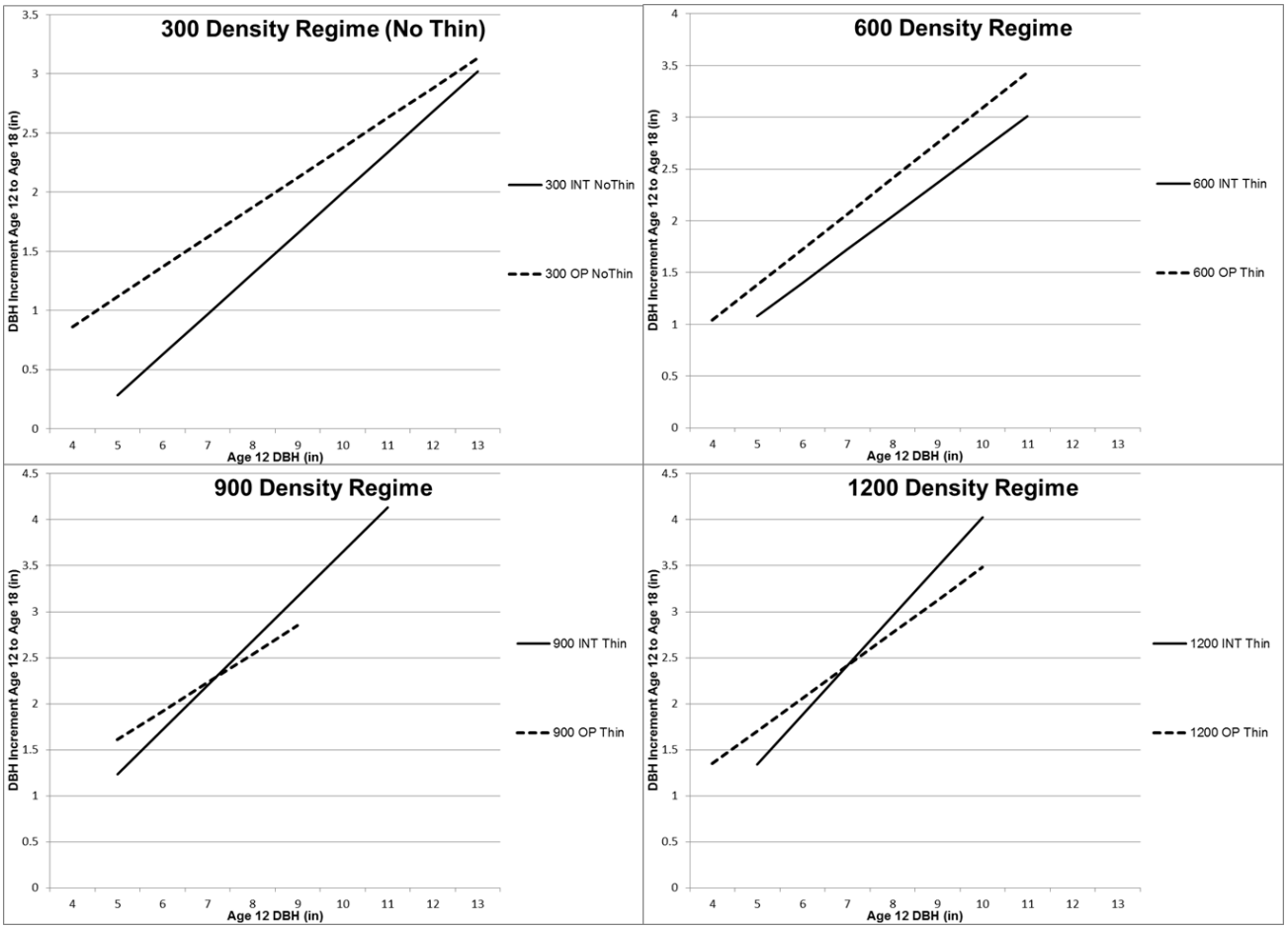


Figure 2—Relationship between post-thin DBH increment (age 12 to 18) and age 12 diameter distribution by individual density regime and management combination.

regimes. With these two density regimes, operational management outperforms the intensive at the lower DBH range, but the intensive management produced larger increments at initial age twelve DBHs that were greater than approximately 7.5 inches.

DISCUSSION

Trees in each density regime and management combination responded differently to thinning due to the different stages of stand development at the time of thinning and the relative thinning intensity imposed (table 2). Though cultural intensity does not seem to effect individual tree DBH response to thinning, it has a large effect on the pre-thin basal area of the 300 and 600 density regimes. When this specific thinning method was imposed to the 600 density regime, there was a large variability in residual basal area between cultural intensities with 87 square feet per acre in the intensive and 79 square feet per acre in the operational due to smaller individual stem DBH. This led to different growing conditions post thinning with the intensive plot having a higher average residual basal area, leading to less available resources per tree. Once trees have similar room to grow, nutrient availability seems to become a more important influence impacting growth. This is seen on the 900 and 1200 density regimes, where cultural intensity has a more noticeable impact. The range in residual basal area was 64 to 68 square feet per acre for the 900 and 1200 density regimes for both management treatments. Due to the small range in average residual basal, trees in the intensive plots had more available resources and more competitive trees responded with greater DBH increment growth. Across all management and density regime combinations, there was a common trend of trees with larger initial DBH having larger DBH increments in response to thinning than trees with smaller initial DBH.

CONCLUSION

Managing intraspecific competition may be the most important factor in this study for maintaining vigorously growing individual trees and consequently stands. Improving the nutrient availability of a stand will also increase the growth of individual trees that have room to grow. Trees with larger initial diameter at the time of thinning have a greater ability to obtain more

resources needed to increase growth, and are thus more competitive. The nutrient availability at stand establishment, up to the time of thinning, has a large effect on the initial condition of stand. It also determines when thinning should occur and the level of response that should be expected.

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LITERATURE CITED

Amateis, R.L.; Burkhart, H.E. 2012. Rotation-age results from a loblolly pine spacing trial. Southern Journal of Applied Forestry. 36(1): 11-18.

Fortson, C.F.; Shiver, B.D.; Shackelford, L. 1996. Removal of competing vegetation from established loblolly pine plantations increases growth on Piedmont and Upper Coastal Plain Sites. Southern Journal of Applied Forestry. 20(4): 188-192.

Fox, T.R.; Allen, H.L.; Albaugh, T.J. [and others]. 2007. Tree nutrition and forest fertilization of pine plantations in the southeastern United States. Southern Journal of Applied Forestry. 31(1): 5-11.

Hennessey, T.C.; Dougherty, P.M.; Lynch, T.B. [and others]. 2004. Long-term growth and ecophysiological responses of a southeastern Oklahoma loblolly pine plantation to early rotation thinning. Forest Ecology and Management. 192(1): 97-116.

Sword Sayer, M.A.; Goelz, J.C.G.; Chambers, J.L. [and others]. 2004. Long-term trends in loblolly pine productivity and stand characteristics in response to thinning and fertilization in the West Gulf region. Forest Ecology and Management. 192(1): 71-96.

Zhao, D.; Kane, M.; Borders, B.E. 2011. Growth responses to planting density and management intensity in loblolly pine plantations in the southeastern USA Lower Coastal Plain. Annals of Forest Science. 68: 625-635.

SOUTHERN PINE BEETLE IN LOBLOLLY PINE: SIMULATING WITHIN STAND INTERACTIONS USING THE PROCESS MODEL SPBLOBTHIN

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Abstract—Pine stand density is a key determinant of damage resulting from attacks by the southern pine beetle (SPB, *Dendroctonus frontalis* Zimm.). High-density stands of maturing loblolly pine (*Pinus taeda* L.) are at high risk for losses to SPB, and reducing stand density is the primary tool available to forest managers for preventing and mitigating damage. Field studies are expensive and inflexible for evaluating various stand treatments over large areas and long time periods, especially with unpredictable SPB population levels. To address these shortcomings, and to provide guidance on silvicultural treatment selection for future field studies, we have developed a process model (SPBLOBTHIN) to simulate the joint population dynamics of loblolly pine and SPB in single stands. Our model grows and tracks individual trees and beetles, both temporally and spatially, and includes stochasticity where desired. The model allows great flexibility in assigning values to input parameters, including those that designate temperature, site index, tree resistance, stand density, and thinning regime. We have also linked model output values to the U.S. Department of Agriculture, Forest Service's Stand Visualization System (SVS), an existing tool for producing realistic and useful views of stand and tree characteristics over time. The objectives of this report are to highlight the flexibility and utility of SPBLOBTHIN, present output created by its linkage with SVS, and compare model simulation results with those obtained in an earlier field study from which pest management thinning recommendations were developed for loblolly pine.

INTRODUCTION

Loblolly pine, *Pinus taeda* L., is the most important commercial forest tree species in the southern U.S. and is a preferred host of the southern pine beetle (SPB, *Dendroctonus frontalis* Zimm.), the most damaging insect pest within its range. Current estimates are that about 32 million ha of SPB-susceptible forests exist nation-wide, of which about 56 percent is dominated by loblolly (Krist and others 2014). Host stand density has long been recognized as a key determinant of damage by SPB in relation to both infestation occurrence and expansion. In turn, thinning is recognized as an important tool for reducing susceptibility of stands to SPB invasion. Specific target basal areas (BAs) are available for residual densities of loblolly pine stands in order to mitigate SPB impacts (Nebeker and others 1985). Current recommendations arose primarily from a 3-year controlled field study conducted by Nebeker, Hodges, Brown and colleagues at Mississippi State University from 1978–80 (herein the NHB study, Brown and others 1987, Nebeker 1981, Nebeker and Hodges 1985). Since the NHB study, thinning recommendations have remained relatively unchanged for southern pines, but information has increased about underlying mechanisms that may be responsible for determining stand resistance levels (e.g., stand meteorology; Thistle and others 2011). Recent results are complementary to older ones, but they have added complexity by increasing the number of relevant variables.

While the risk of SPB damage has continued to grow with increasing stand densities and tree ages in the South, our resources to establish long-term, large-scale field studies to address these concerns and to identify potential solutions have declined. As a result, predicting the outcomes from interactions between existing and future forests and their pests has become more dependent than ever on predictive simulation models. In this paper, we describe SPBLOBTHIN, one of the suite of "SPBLOB" process models (i.e., based on simulating biological processes and interactions), which tracks day-to-day changes in SPB populations and annual changes in loblolly pine growth and mortality in a plantation setting (Bishir and others 2009). The growth-and-yield portion of SPBLOBTHIN is derived from PTAEDA2 (Burkhart and others 1987, Daniels and Burkhart 1975), thus providing spatially explicit individual tree simulations for growth characteristics and mortality. The full model accounts for tree and beetle locations within the stand, with the latter being spatially referenced by tree locations. Beetle

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population updates are provided up to four times per day depending on temperature, while tree growth and mortality from intraspecific competition are updated annually.

Field studies have been essential for documenting the relationship of SPB attacks to stand characteristics (see Coster and Searcy 1981) and for developing thinning guidelines. Those of the NHB study are the basis for current pest management thinning recommendations in the southern pines. However, field studies are expensive and burdensome to maintain, and complex and unpredictable processes such as climate, host resistance, temperature, canopy structure, stand structure, host availability, and feral insect populations interact to produce variable outcomes. Each site and SPB spot is unique, so prediction of tree mortality and stand damage must address shifting probabilities and chance occurrences. Along with accumulation of this knowledge has come recognition that a modeling approach can be an effective or even necessary method for mechanistically addressing important management factors. The results can be used to revise and improve current thinning recommendations under variable meteorological and climatic conditions as well as to account for the dynamic constraints imposed by forest harvesting and business operations. That is, market and business forces—along with technological advances in equipment design, etc.—can foster changes in forest management practices over time that may not be predictable. Models offer maximum flexibility to address such issues in a timely manner. Accordingly, our objectives in this paper are:

1. To demonstrate the function, flexibility, and utility of SPBLOBTHIN as a means to evaluate thinning regimes for SPB prevention;
2. To compare results obtained from model simulations to those obtained from the NHB field study, a cornerstone study from which current southern pine pest management thinning guidelines are based.

MATERIALS AND METHODS

General Introduction to SPBLOBTHIN

SPBLOBTHIN uses a modified version of PTAEDA2 (Daniels and Burkhart 1975, Burkhart and others 1987) to produce spatially explicit individual tree simulations for growth and the mortality resulting from intraspecific competition within a stand. Each stand simulation consists of five stages: 1) choosing tree variables, site index, temperature regime, and planting density/spacing; 2) tree growth and survival through an 8-year juvenile period during which no SPBs are present; 3) daily updates of the number of SPBs at each tree and the tree mortality they cause (some beetle activities are updated four times daily); 4) yearly updates of tree growth and mortality related to tree competition; and 5) timber harvest at designated periods determined by user-selected triggers and targets. The combination of thinning triggers along with the pattern and severity of tree removal can be set by the user, but 100 of the most useful combinations are "preprogrammed" into SPBLOBTHIN and may be chosen by their regime number (Bishir 2014). Regimes may be updated in future versions of SPBLOBTHIN based upon user input. In general, tree activities are simulated using PTAEDA2 subroutines (Burkhart and others 1987, Daniels and Burkhart 1975) generously provided by H. E. Burkhart for our use, while SPB activities are unique to SPBLOBTHIN. Output data vary by user preference and may include such things as the number of trees killed by competition or SPB, trees surviving, residual volume, and removed volume by product class (i.e., biomass, chip-and-saw, pulpwood, sawtimber). Additional information about these and many other model details—including parameter estimates, output options and references—are available in Bishir and others (2009) and Bishir (2014).

SPBLOBTHIN tracks day-to-day changes in tree survival and beetle populations, including the location within the stand of individual female beetles (males are assumed to join females; Bishir and others 2009). Beetle locations are spatially referenced by tree, and following Bunt and others (1980) are categorized as landed, parent, dispersing, or juvenile. Beetle population updates are provided up to four times per day, whenever thresholds of developmental and flight temperatures are achieved during the period (Bishir and others 2009).

Tree data output from SPBLOBTHIN may be visualized by inputting it to the U.S. Department of Agriculture, Forest Service's Stand Visualization System (SVS, v. 3.30; McGaughey 1997; fig. 1). This can be done on a per treatment simulation basis by copying output and pasting into SVS one output file (simulation result) at a time or as a batch process via the SVS Image Generator (SIG), which is software code written for this project by Tony Austin (Austin and Bishir 2014) and updated by Dan Leduc (Information Technology Specialist, USDA Forest Service, Pineville, LA). The SIG links output from SPBLOBTHIN to SVS, producing pictures of events (before and after stand entries for thinning, initiation of SPB spots and their development over time, etc.) in the simulated stand through the time period specified by the user. The SIG output resembles a movie that can be played at speeds chosen by the user.

Objective 1—To demonstrate the function, flexibility, and utility of SPBLOBTHIN, we selected examples to show how simulated stands change following thinning or SPB infestation (spot occurrence and growth). Both examples are the result of SPBLOBTHIN simulations run with a 17 °C average annual temperature (as in Pineville,

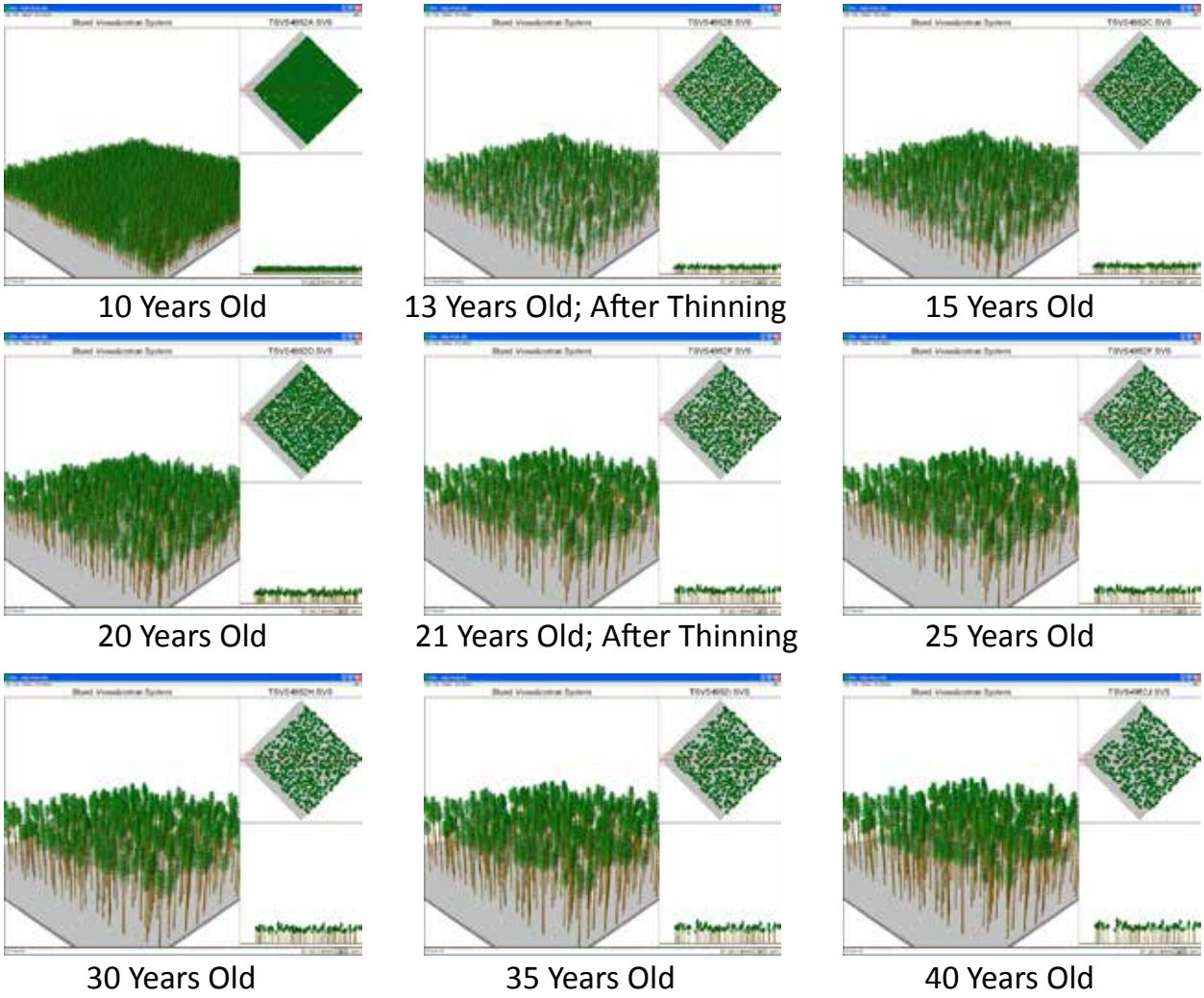


Figure 1—Example of SVS output at various stand ages between 10 and 40 years old.

LA) and a site index of 60 (base age 25). We also note that in our experience, PTAEDA2 (the base growth-and-yield model) does not provide reliable results for stands older than 40 years, so we end all of our stand simulations by that age.

The first example begins with an illustration of an unthinned stand (the no-management alternative, regime 1 of the preprogrammed regimes; Bishir 2014) and a pattern of tree mortality from SPB that is commonly observed in SPBLOBTHIN under this regime (fig. 2). In this simulation, the stand was planted at 2,000 trees per acre (TPA), a surrogate for high planting density with natural infill, and received no precommercial or commercial thinning.

In the second example (fig. 3), the stand was again established at 2,000 TPA, but in this simulation, we selected one of the preprogrammed management regimes that included thinning (number 39; Bishir 2014). In this regime, there is a precommercial swath thinning triggered when there are >750 live TPA with average annual diameter growth of <5 percent for two consecutive years and average diameter at breast height (dbh) <5 inches. Target residual TPA was 450. This was followed by a commercial fifth-row thinning, perpendicular to the swath thinning, triggered in this regime when stand basal area (BA) exceeds 120 square feet per acre and average dbh is at least 5.5 inches. Target residual BA following the commercial thinning is 70 square feet per acre in this regime. We are not aware of field data documenting the pattern of tree mortality caused by SPB in row-thinned stands, but we believe the pattern resulting from this simulation is reasonable (fig. 3).

Objective 2—To compare field results to those produced by SPBLOBTHIN, we chose data and

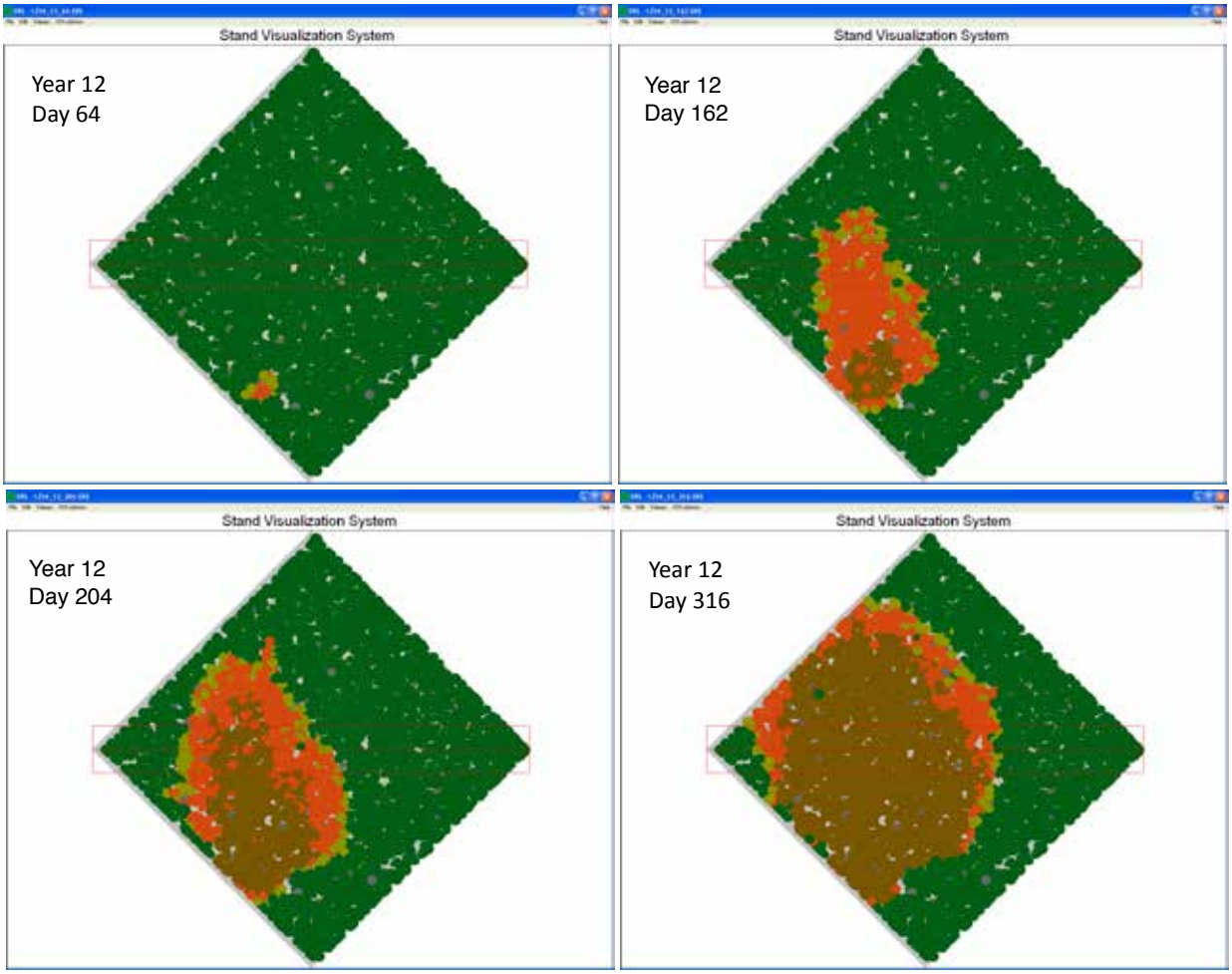


Figure 2—SPB spot in unmanaged stand (Example 1). Unmanaged stand regime 1 of pre-programmed regimes in SPBLOBTHIN. Stand was planted at 2,000 trees per acre to simulate planted seedlings and natural infill. The simulation realized its first southern pine beetle infestation at year 12 and the infestation is followed until day 316. As is typical in SPBLOBTHIN simulations (and in the real world), unmanaged stands realize extensive damage from SPB spots.

outcomes from the NHB study reported in Brown and others (1987), Nebeker (1981), and Nebeker and Hodges (1985). As mentioned, the NHB study was conducted from 1978–80 and evaluated the impact of thinning intensity on the resulting stand damage (primarily tree mortality) caused by SPB within each year. The loblolly pine plantation used for the study was established at the Starr Memorial Forest, Mississippi State University, Mississippi in 1956 using a spacing of 10- by 10-feet. Stand basal area averaged 197 square feet per acre prior to initiation of thinning treatments in winter 1977-1978 at tree age 22. Thirty-six 1-acre plots were established, providing nine replicates for each of four thinning intensities [untreated (BA = 197, called "200" in the literature describing study results), 130, 100, and 70 square feet per acre]. Thinning began with the removal of every fifth row in each plot, except where high levels of fusiform rust (*Cronartium fusiforme* Hedge & Hunt) precluded this method from leaving enough residual trees to meet the BA target. In this case, infected trees were preferentially removed in lieu of row trees. The fifth-row thinning was followed by secondary removal stages that consisted of "thinning from below" and removal of trees infected with fusiform rust between rows to reach the final BAs desired for individual plots (Brown and others 1987, Nebeker 1981).

Beginning in 1978, the NHB study challenged 12 plots per year with SPB. To promote attacks and evaluate spot initiation and growth, a single tree at each plot center was baited with SPB aggregation pheromone to provide an initial focal point. Beetle-infested logs and/or bark were placed at the plot center. In 1978, an estimated 5,000 SPBs were introduced on a single date to each plot (as brood). Because this provided a challenge that was less severe than desired, the number of SPBs was increased with the introduction of additional SPB brood material to 10,000-15,000 accumulating over multiple dates in 1979 and 1980.

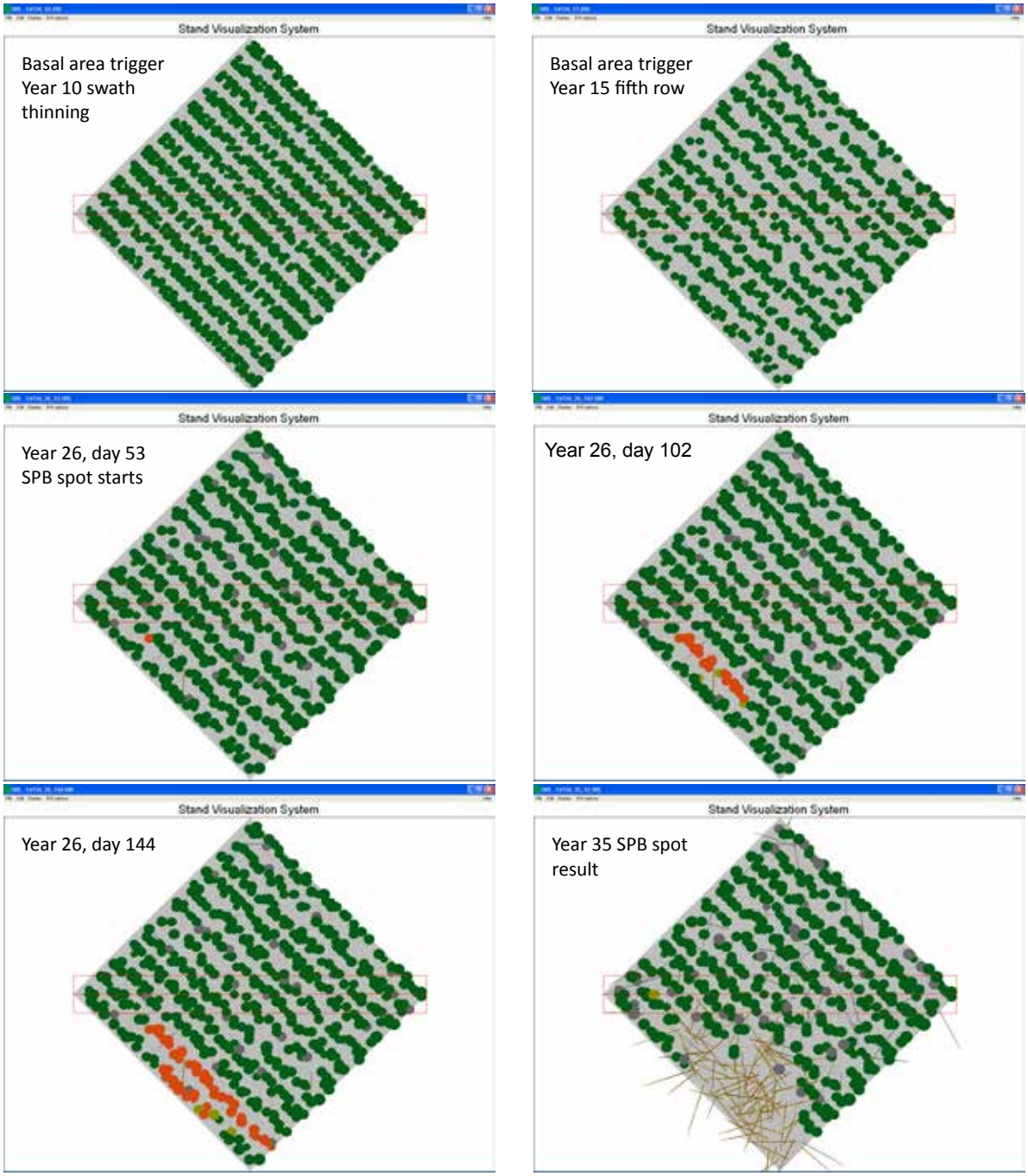


Figure 3—Example 2 showing densely planted loblolly pine as in Figure 2, but with precommercial thinning at year 10 and commercial thinning at year 15. During year 26 the first SPB spot was observed at day 53 and it is followed until day 144. Tree mortality progressed along a single swath of residual trees before extending across to a second swath of residual trees. Year 35 is also provided to show the depiction of fallen (attacked) trees in SVS.

Feral SPBs also moved via immigration throughout the NHB plots, but no data are provided to estimate their numbers (e.g., from trapping). As we discuss later, immigration of SPB into the plots was critical for the 1979 results to be realized by simulation. In each plot, the total number of trees killed by SPB was tallied (Brown and others 1987, Nebeker 1981), and these data were used to make the primary comparisons between results from NHB and corresponding SPBLOBTHIN simulations. Secondary data also were recorded, including estimates of SPB attacks, brood production, and other selected variables (Nebeker 1981).

Stand inputs to SPBLOBTHIN simulations followed as closely as possible the data and procedures reported by the NHB study. Each simulated 1-acre stand was "planted" using 10- by 10-foot spacing. Stand site index was defined to be 70 feet at base age 25 [about equivalent to the 110 feet at age 50 (Hamilton 2000) used by NHB]. To check these approximations, we simulated 100 independent, unthinned stands, each with 10- by 10-foot spacing and SI = 70. These showed an average BA value of 194.4 ± 4.08 square feet per acre (mean +/- standard deviation) at age 22 compared to an overall average of 197.0 square feet per acre computed from the NHB data (Nebeker 1981). Simulated plots were then challenged with SPB by infesting a central tree with a sufficient number of SPBs to kill it. Each treatment combination (four levels of stand density) was simulated over 500 independent trials with the mean number of trees killed per plot being tallied.

In addition to tree mortality, Nebeker (1981) reports the total brood emergence for each of 17 trees killed by SPB in plots during the critical year of 1979. These field results were compared to brood emergence results from SPBLOBTHIN (1,500 simulations, 336 killed trees) by employing a chi-square test with four degrees of freedom; brood counts were first binned into five ordinal categories following the procedures of Hogg & Tanis (2001).

RESULTS

Objective 1

In the first example, SPB-caused tree mortality in an unmanaged stand was first observed in year 12 at day 64 (fig. 2). As the model updates, spot growth is obvious, depicting a spot head advancing in a northwesterly direction with expanding flanks and culminating on day 316 with much of the stand being lost to SPB.

In the second example (fig. 3), the high-density stand initially underwent a precommercial thinning followed later by a commercial thinning, both dictated by the user-selected regime 39. Figure 3 begins with the first thinning, conducted as a precommercial swath thinning, at year 10. This was triggered "automatically" in SPBLOBTHIN by the established precommercial thinning parameters. At age 15, the first commercial thinning treatment was employed (fifth row perpendicular to previous swath thinning), this time by the model's automatic commercial thinning trigger. The stand grew until year 26, when a SPB spot first appeared with a single dead tree at day 53. The spot expanded in a pattern that we often see in simulations produced by the model (and occasionally in the field)—tree mortality progressing along rows of more closely spaced trees until attacking beetles spread across the swath of removed rows. Ultimately, the simulation culminated with SPB-killed trees being present in a multiple row spot, and by year 35 scattered elsewhere, with SVS pictorially showing the attacked trees as fallen (fig. 3).

Objective 2

The primary objective of the NHB study was to evaluate the effect of thinning treatments on tree mortality and consequent losses of timber volume. For convenience, table 1 provides the yearly tree mortality data by BA class observed in the NHB study (Brown and others 1987). In our simulations, we resorted each year to a "trial and error" method of choosing the model immigration parameter (*parimmi*) values that produced the best approximations to the NHB field observations (table 2). It is evident from table 1 that tree mortality was greater in 1979 than in the other years, especially at the higher BAs. We found that the extent of these two results from 1979, greater tree mortality and a nonlinear, positive relationship between the number of trees killed by SPB and BA, were unique to that year and were the key factors necessary for successfully simulating the NHB results.

Because there are no field data from which to realistically base the immigration parameter in SPBLOBTHIN (*parimmi*), we employed a range of positive immigration values to the 1979 BA 130 and BA 200 plots. Results are displayed in table 2 using 450 landed female SPBs per tree (the number that SPBLOBTHIN simulations show were most probably present in NHB) and range from 0.004 (BA 130) to 0.008 (BA 200). We note that if field-based information becomes available to indicate values the immigration parameter should take, they can easily be specified in SPBLOBTHIN because *parimmi* is user-defined.

A trend readily apparent in results observed in both the NHB field experiment and our model simulations is that denser stands—those with higher BAs—tend to experience larger spots than do stands with lower BA values. This trend was observed in earlier studies of SPB spot growth in east Texas (Cameron and Billings 1988, Hedden and Billings 1979) and was the primary finding from the NHB field study. Likewise, similar

Table 1—Observed yearly mean numbers of trees killed per plot in each combination of year and BA as derived from Table 4 of Brown and others (1987)

	BA Values			
Year	70	100	130	200
	Mean numbers of trees killed per plot			
1978	0	0.33	1.0	2.67
1979	0.67	1.0	2.67	11.33
1980	0.33	1.0	0.67	2.67

Table 2—Mean numbers of trees killed per plot in simulations of 1979 (age 23) plots having 450 'Landed' SPB assigned to a single, initially infested tree (as per Nebeker, 1980). For BA groups 130 & 200, the non-zero immigration parameter values in these simulations were chosen through trial and error in order to approximate the mean numbers of trees reported by Nebeker (1980) and Brown and others (1987) as having been killed by SPB in 1979

Post-Thin BA Values	70	100	130	200
Immigration parameter parimmi	0.0	0.0	0.004	0.008
Simulated mean #s of trees killed	0.59	1.28	2.70	10.93
NHB data from Table 1	0.67	1.00	2.67	11.33

behavior observed in the simulations, whether or not beetle immigration occurred, further validates the SPBLOBTHIN model code.

In addition to tree mortality, we compared NHB counts of SPB brood from trees killed in the NHB study in 1979 to average brood counts obtained from SPBLOBTHIN simulations. Nebeker (1981) reports total brood emergence for each of 17 trees killed by SPB in the 12 experimental plots infested during 1979, and we evaluated brood emerged from 336 trees killed by SPB in 1,500 simulation runs of SPBLOBTHIN in this experiment. As indicated earlier, our test of the hypothesis H_0, that the field and model distribution functions for brood counts were the same, was conducted by employing a chi-square test with four degrees of freedom. Because the sum (4.46) was less than the critical value for 5 percent significance (9.49), H_0 was not rejected, and the brood numbers from SBPLOBTHIN were deemed an acceptable representation of the field results.

DISCUSSION

The process model SPBLOBTHIN was employed to generate results and images to demonstrate the model's flexibility, utility, and performance. Example output illustrates that thinning treatments followed the expected pattern of row and between-row tree removals (fig. 3). In addition, development of SPB infestations (spots) seemed also to follow expected patterns (figs. 2 and 3). Simulation of SPB spots showed attacked and dead trees to be aggregated, damage to increase over time, and damage to be more severe in unmanaged stands. In thinned stands, when SPB spots did occur, they frequently extended along rows where distances to adjacent trees were shortest. This expected behavior is sometimes observed in the field, but spatially explicit field data to test this behavior are lacking.

Current pest management recommendations for thinning loblolly pine stands are primarily based upon the NHB study (Brown and others 1987, Nebeker 1981, Nebeker and others 1985). To compare field and model results, we employed SPBLOBTHIN to simulate the interaction between SPB and loblolly pine under the conditions reported for the NHB field study. Mean levels of field tree mortality by year of the NHB study are shown in table 1. Similar results were produced by SPBLOBTHIN for the years 1978 and 1980 (table 1). The exceptions in year 1979 are the untreated (200 square feet BA) and 130-square-foot BA stands, which were recognized as anomalies within the NHB study as well (table 1). For SPBLOBTHIN to effectively simulate the NHB field data on tree mortality in 1979, the realization

of two phenomena was necessary. First, immigration of SPB adults into the stand was required for 1979 (immigration rate greater than zero in that year), and second, immigration rates needed to be higher in denser (higher BA) stands (table 2). The likelihood of these phenomena being reasonable is discussed below.

The probability that greater SPB immigration could have taken place in the NHB field plots in 1979 is supported by two lines of evidence. First, Nebeker (1981) provides estimates of the number of attacking beetles and the number of beetles emerging from the bolts and bark brought into selected plots to challenge trees. By comparing the number of SPBs brought into the plots to the number of attacking beetles, an indication is given as to whether immigration of beetles into plots was necessary to produce the observed level of beetle activity. To do this, we limited our comparison to the first 30 days after attacks were first observed so that brood adults could be excluded from consideration. This approach did not account for parent adult reemergence, but the discrepancy between the number of SPB attacks and the estimated brood emergence in BA = 200 (unthinned) plots is too great for reemergence to account for the difference observed. In 1979, as in each year of the study, three unthinned plots were challenged. In plot 18, Nebeker (1981) estimates that 11,121 SPBs were released into the plot and that 24,808 attacked 9 trees in the plot by day 28. Similarly, in plot 20, counts indicated that 16,356 beetles were released and 38,466 attacked 12 trees by day 21. Finally, in plot 29, there were an estimated 14,439 SPBs released and 24,700 beetles attacked 10 trees. These estimates support that immigration into the unthinned plots in 1979 was necessary to account for the level of SPB activity observed.

In addition, records of SPB spot abundance at the county level (Pye and others 2008) support a higher probability of immigration into plots in 1979. Historical data on spot numbers from Oktibbeha County, MS (where the NHB study was located), indicate that in 1978, SPB spots were present below outbreak levels, between 0.1 to <1.0 spot per 1,000 acres of host type. In 1979, the county transitioned to severe outbreak status (≥3.0 spots per 1,000 acres), suggesting that area-wide beetle populations were abundant and that active spots would initially serve as sinks for dispersing SPB because there were fewer competing spots in the surrounding landscape to draw beetles from the plots. By comparison, in 1980, during the second year of severe outbreak levels of SPB activity, attractive (competing) spots would more likely already be present and relatively abundant in the landscape. In 1981, the county was again below outbreak levels. These records are consistent with a pattern of the greatest numbers of dispersing beetles being present in the exponential population growth phase during the transition into a severe outbreak (Clarke 2012).

Available evidence from field studies also supports the indication from SPBLOBTHIN simulations that the immigration rate is higher for denser stands as determined by the number of trees killed. A recent study by Nowak and others (2015) on the Homochitto and Bienville National Forests (MS) investigated SPB spots that occurred during 2012. Of the 910 spots evaluated, only 2 (0.22 percent) were found in recently (past 6 years) thinned stands. In addition, research on the movement of air in thinned versus unthinned loblolly pine stands has demonstrated that the latter provides a meteorological environment that promotes structural integrity of gas plumes (Thistle and others 2011). Such an environment should allow easier semiochemical tracking by beetles during the host-finding process.

The biological process model SPBLOBTHIN performed well by the measures investigated in this study. Linkage with SVS provided pictorial evidence that simulated thinning operations resulted in appropriate patterns of tree removal and retention, and target residual basal areas being achieved. The model has great flexibility to simulate a wide variety of conditions and actions as specified by the user. Our simulations that mimicked conditions in the NHB study, performed from 1977–80 on trees planted in 1956, provided valuable insight into the number of landed SPBs necessary for tree mortality to be consistent with that observed in the field (450 female beetles per tree) and led to the conclusion that immigration behavior must have occurred in 1979. This was heretofore unrecognized from the field study. The model appears to be a useful tool for investigating joint population dynamics of SPB and loblolly pine. SPBLOBTHIN may be particularly valued for its flexibility and as a surrogate for large-scale field studies, which are difficult to implement and inflexible over the space and time needed to answer SPB/loblolly pine interaction questions at the stand level. It is probable that multiple factors favor host trees over beetles when stands are thinned, but our lack of knowledge about stand-beetle interactions contributes to uncertainty in predicting the impact of thinning regimes, especially as they evolve with changes in forests and forestry practices. Simulation models allow the flexibility necessary for navigating through changes that occur at a pace faster than long-term field studies can accommodate. They also help to identify knowledge gaps, provide a tool for preliminary evaluation of proposed operational treatments for field studies, and provide estimates of outcomes in interim periods before field studies are completed. Although irreplaceable, long-term field studies are expensive, burdensome to maintain, and difficult to replicate. As an alternative, we propose the use, application, and continued refinement of

process models such as SPBLOBTHIN to guide and assist decisionmaking for field studies and timber management operations.

LITERATURE CITED

Austin, T.; Bishir, J. 2014. SVS image generator (SIG) user guide. In: Bishir, J. User guide for the SPBLOBTHIN Code. Appendix Two: 24-32. Available at http://www.srs.fs.usda.gov/idip/tools. [Date accessed: Dec. 29, 2015].

Bishir, J. 2014. User guide for the SPBLOBTHIN code. 32 p. Available at http://www.srs.fs.usda.gov/idip/tools. [Date accessed: Dec. 29, 2015].

Bishir, J.; Roberds, J.; Strom, B.; Wan, X. 2009. Documentation and user guides for SPBLOB: a computer simulation model of the joint population dynamics for loblolly pine and the southern pine beetle. Gen. Tech. Rep. SRS-114. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 83 p.

Burkhart, H. E.; Farrar, K. D.; Amateis, R. L.; Daniels, R. F. 1987. Simulation of individual tree growth and development in loblolly pine plantations on cutover, site-prepared areas. Publ. No. FWS-1-87. Blacksburg, VA: Virginia Polytechnic Institute and State University, School of Forestry and Wildlife Resources. 47 p.

Brown, M. W.; Nebeker, T. E.; Honea, C. R. 1987. Thinning increases loblolly pine vigor and resistance to bark beetles. Southern Journal of Applied Forestry. 11: 28-31.

Bunt, W.D.; Coster, J. E.; Johnson, P. C. 1980. Behavior of the southern pine beetle on the bark of host trees during mass attack. Annals of the Entomological Society of America. 73: 647-652.

Cameron, R. S.; Billings, R. F. 1988. Southern pine beetle: factors associated with spot occurrence and spread in young plantations. Southern Journal of Applied Forestry. 12: 208-214.

Clarke, S. 2012. Implications of population phases on the integrated pest management of the southern pine beetle, Dendroctonus frontalis. Journal of Integrated Pest Management. 3(1): 1-7. Available at DOI: http://dx.doi.org/10.1603/IPM11021. [Date accessed: March 1, 2012].

Coster, J.E.; Searcy, J.L., eds. 1981. Site, stand, and host characteristics of southern pine beetle infestations. Tech. Bull. No. 1612. Washington, DC: US Department of Agriculture Combined Forest Pest Research and Development Program. 115 p.

Daniels, R.F.; Burkhart, H.E. 1975. Simulation of individual tree growth and stand development in managed loblolly pine plantations. Publ. No. FWS-5-75. Blacksburg, VA: Virginia Polytechnic Institute and State University, Division of Forestry and Wildlife Resources. 69 p.

Hamilton, R.A. 2000. Forest soils and site index. Woodland Owner Notes. Raleigh, NC: North Carolina State University, Cooperative Extension. 2 p.

Hedden, R.L.; Billings, R.F. 1979. Southern pine beetle: factors influencing the growth and decline of summer infestations in East Texas. Forest Science. 25: 547-556.

Hogg, R.V.; Tanis, E.A. 2001. Probability and statistical inference. 6th Ed. Upper Saddle River, NJ: Prentice-Hall Inc. 704 p.

Krist., F.J., Jr.; Ellenwood, J.R.; Woods, M.E. [and others]. 2014. 2013-2027 National insect and disease forest risk assessment. FHTET-14-01. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Forest Health Technology Enterprise Team. http://www.fs.fed.us/foresthealth/technology/nidrm.shtml. [Date accessed: April 18, 2014].

McGaughey, R.J. 1997. Visualizing forest stand dynamics using the stand visualization system. In: Proceedings of the 1997 ACSM/ASPRS Annual Convention and Exposition, April, 7–10. Seattle, WA. American Society for Photogrammetry and Remote Sensing. 4: 248–257.

Nebeker, T.E. 1981. Manipulation of stand density for prevention of southern pine beetle infestation in loblolly pine plantations. U.S. Department of Agriculture Combined Forest Pest Research and Development Program Final Report. 87 p.

Nebeker, T.E.; Hodges, J. D. 1985. Thinning and harvesting practices to minimize site and stand disturbance and susceptibility to bark beetle and disease attacks. In: Branham, S. J. Thatcher, R. C., eds. Integrated pest management research symposium. Gen. Tech. Rep. SO-56. New Orleans, LA. U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 263-271.

Nebeker, T.E.; Hodges, J. D.; Karr, B.L.; Moehring, D.M. 1985. Thinning practices in southern pines—with pest management recommendations. Tech. Bull.1703. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 1-36.

Nowak, J. T.; Meeker, J.R.; Coyle, D.R. [and others]. 2015. Southern pine beetle infestations in relation to forest stand conditions, previous thinning, and prescribed burning: evaluation of the southern pine beetle prevention program. Journal of Forestry. 113(5): 454-462. DOI: 10.5849/jof.15-002. [Date accessed: Dec. 29, 2015].

Pye, J. M.; Price, T. S.; Clarke, S. R.; Huggett, R. J., Jr. 2008. A history of southern pine beetle outbreaks in the southeastern United States through 2004. Available at http://www.srs.fs.usda.gov/econ/data/spb/index.htm. [Date accessed: March 5, 2013.]

Thistle, H.W.; Strom, B.; Strand, T. [and others]. 2011. Atmospheric dispersion from a point source in four southern pine thinning scenarios: basic relationships and case studies. Transactions of the ASABE. 54(4): 1219-1236.

WILL CROWN IDEOTYPE HELP DETERMINE OPTIMUM VARIETAL SILVICULTURE?

Timothy J. Albaugh, Thomas R. Fox, Marco A. Yanez, Rafael A. Rubilar, and Barry Goldfarb[1]

Abstract—Recent advances in somatic embryogenesis permit large numbers of clonal loblolly pine (*Pinus taeda* L.) to be produced and deployed. Clones may have greater growth (mean annual increment exceeding 30 cubic meters per hectare per year), greater stand uniformity and may be more susceptible to genotype by environment interactions when they are deployed in intensively managed plantations. Consequently, large numbers of clones will need testing under a range of silvicultural treatments to effectively and efficiently identify those that are best for deployment. Crown ideotypes are described by their crown dimensions, where the crop ideotype has tall (long live crown length) and narrow (short branch length) crowns and the competition ideotype has short (short live crown) and wide (long branch length) crowns. We tested the hypothesis that the response to silvicultural input was not related to crown dimension in a study in the Virginia Piedmont where six genetic entries (four clones, one mass control pollinated family and one open pollinated family) were planted at three initial stocking levels (617, 1235, and 1852 stems per hectare) and two levels of silvicultural input (low, i.e., similar to typical operations, and high, where nutrient limitations were eliminated). After three years of growth, significant silviculture and genetic entry effects were observed for diameter, height, live crown length, crown width, crown volume and stem volume increment, where silviculture increased these variables for all genetic entries although the increase due to silviculture was not consistent across genetic entry (genetic entry by silviculture interaction). The stem volume growth response to silvicultural treatment decreased with increasing crown volume. Crown ideotype may be useful in determining the response of clonal material to silvicultural input. However, it is important to know the conditions under which the ideotype is determined to be able to successfully use this method to predict the response to silvicultural input.

INTRODUCTION

Clonal forestry holds great promise to increase forest plantation productivity in the near term (Wright and Dougherty 2007). Clonal forestry relies on vegetative propagation to mass-produce identical copies of selected individual trees that possess improved genetic potential (Gleed and others 1995). Although the technology to mass-produce loblolly pine (*Pinus taeda* L.) clones is still developing, recent advances in somatic embryogenesis now permit large numbers of clonal loblolly pine to be produced and deployed. Based on results from clonal plantations in other parts of the world, it may be possible to increase loblolly pine productivity up to 50 percent (mean annual increment exceeding 30 cubic meters per hectare per year) by deploying appropriate clones to specific soil types, and then implementing integrated, intensive silvicultural regimes. Clonal plantations typically have greater stand uniformity but only when resource limitations are eliminated. In addition, genotype by environment interactions may be more common when clones are deployed in intensively managed plantations (Sierra-Lucero and others 2003). Consequently, the deployment of elite genotypes as clonal material requires a better understanding of how these genotypes respond to silvicultural manipulations (Li and others 1991). At the same time, a clonal forestry program should include an ongoing process of testing large numbers of candidate clones to determine if they are suitable for forest plantation deployment (Gleed and others 1995). These conditions create a situation where large numbers of clones need to be tested under a range of silvicultural treatments to be effectively and efficiently evaluated to identify those that are best for deployment.

One possibility for screening clones and how they respond to silvicultural treatment is to utilize the ideotype concept (Dickmann 1985). An ideotype has a consistent set of properties or characteristics that tend to respond to management practices in a uniform and consistent manner. In forestry, the ideotype is usually defined by crown characteristics such as

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branch size, branch angle, number of branches and tendency to self-prune (Martin and others 2001). 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. In contrast, the competition ideotype has a large crown and aggressively expands its crown to exploit site resources to the detriment of its neighbors. The appropriate ideotype to plant in a specific stand depends on management objectives. For example, a stand with narrow crown trees may be desired in highly stocked plantations where biomass production is the management objective. In contrast, a stand with wide, large crown trees may be desired in a plantation managed at low stocking for sawtimber production. If crown ideotype can be used to classify clone response to silvicultural input, it may be possible to predict the growth responses of newly developed clones to silviculture treatments with minimal additional empirical testing.

With this background, our interest was to examine how the concept of crown ideotype could be used to predict response to silvicultural treatment. Crown ideotypes are described by their crown dimension, where the crop ideotype has tall (long live crown length) and narrow (short branch length) crowns and the competition ideotype has short (short live crown) and wide (long branch length) crowns. Consequently, we tested the hypothesis that response to silvicultural input was not related to crown dimension.

METHODS

We installed a block plot study using a split-split plot design with four replications (Gomez and Gomez 1984) at a site located at the Reynolds Homestead Research Center in the Virginia Piedmont in 2009 (Vickers and others 2012). The main plot treatment was silvicultural level (low and high). The split plot treatment was genetic entry (1 open pollinated family, 1 control mass pollinated family and 4 clones). The split-split plot treatment was initial stocking (617, 1235 and 1852 trees per hectare). This design resulted in 36 plots in each of the four blocks, yielding a total of 144 plots.

The entire site (all treatments) was chemically prepared using an aerial application of 0.29 liters per hectare of Accord XRT II, 0.29 liters per hectare of Milestone VM Plus, 1.46 liters per hectare of Chopper and 1.46 liters per hectare of DLZ oil in the summer of 2008, followed by broadcast burning in November 2008. The silvicultural treatments were designed to follow typical operational treatments (low) and a treatment where no nutrient limitations would be experienced by the trees (high). Low silviculture plots received banded herbaceous weed control during the first growing season. High silviculture plots received broadcast herbaceous weed control during the first two years. Directed sprays of herbicide were used to eliminate all hardwood competition in these plots. High silviculture plots also received tip moth control (soil-injected fipronil applied at 1.5 ml per tree) at planting and were fertilized with 112 and 12 kg per hectare of elemental nitrogen and phosphorus, respectively, at the beginning of the second year after planting. All fertilizer applications were banded on the row.

The genetic entries were one open-pollinated family, one control mass pollinated family and four clones. The clones represent a range in crown ideotypes; two with moderately wide crowns and two with broad crowns.

Initial stocking was designed for three product types, i.e., sawtimber (617 trees per hectare), pulp and sawtimber (1235 trees per hectare), and biomass or pulpwood (1852 trees per hectare). The between-row spacing for all stocking levels was 3.65 meters for all three planting densities. The distance between the trees within the row was 4.42, 2.21, and 1.48 meters, for the 617, 1235 and 1852 trees per hectare treatments, respectively. Individual plots in three blocks had 81 trees in a 9 row by 9 planting spot configuration and one block had 63 trees per plot in a 7 row by 9 planting spot configuration. The internal 25 trees (5 rows x 5 spots) were used as measurement trees for this analysis.

Trees were planted in February 2009 as containerized seedlings for the clonal material and as bare root plants for the open pollinated and mass control pollinated families.

Height, diameter at breast height (1.3 m), height to live crown (height of the lowest branch with live foliage) and crown width (average of the on row and the across row distance from drip line to drip line) were measured annually in the dormant season (December- January). Individual tree volume was calculated using an equation derived from an over-bark volume equation for unthinned trees (Tasissa and others 1997) as:

$$V = (0.21949+(0.00238 * ((D*0.3937008) * (D*0.3937008) * (H*3.28084))))*0.02831685 \quad [1]$$

where V is individual tree volume in cubic meters per tree, D is diameter at breast height in centimeters and H is height in meters. The volume increment in year three was the volume at the end of year three minus the volume at the end of year two. The response to silviculture was calculated as the difference between the three-year volume increment in the high and low silvicultural plots for corresponding combinations of block, genetic entry and stocking. Live crown length

was calculated as height minus height to live crown. Crown volume was calculated as the volume of a cone:

$$CRV = 3.14 * (CW/2) * (CW/2) * LCL \quad [2]$$

where CRV is crown volume in cubic meters per tree, CW is crown width in meters, and LCL is live crown length in meters.

PROC MIXED (SAS-Institute 2002) was used to examine treatment effects after three years for diameter, height, live crown length, crown width, crown volume and stem volume increment. Block was considered a random effect. PROC MIXED was also used to examine the relationship between crown dimensions (live crown length, crown width and crown volume) of the low silvicultural treatments and the volume response to silvicultural treatment. Tukey-Kramer means tests were used to determine treatment differences for volume response to silviculture after three years. All statistical tests were evaluated with alpha equal to 0.05.

RESULTS

Significant clone and silviculture differences were observed for diameter, height, live crown length, crown width, crown volume and volume increment (table 1). Stocking did not have a significant effect on any of the measured variables. Averaged across all genetic entries, high silviculture increased diameter (1 cm, 31 percent), height (0.3 m, 11 percent), live crown length (0.4 m, 14 percent), crown width (0.5 m, 43 percent), crown volume (1.6 cubic meters per tree, 146 percent) and stem volume increment (1145 cubic centimeters per tree per year, 95 percent) compared with low silviculture (table 2). Significant clone by silviculture interactions were observed for diameter, height, live crown length, and crown volume. All genetic entries responded positively to silviculture, whereas Clone 4 had a significantly greater stem volume response to silviculture compared with the other genetic entries (table 2). There were no differences in stem volume response to silviculture among the other genetic entries.

Stem volume incremental response to silviculture decreased with increasing crown volume across all genetic entries (fig. 1). Genetic entry did not affect the slope of the stem volume increment response to silviculture and crown volume relationship but it did affect the intercept. The intercept for Clone 4 was greater than the intercept observed for all other genetic entries.

DISCUSSION

Stem volume growth response to silviculture was related to crown volume. Crown volume is calculated from crown dimensions (live crown length and crown radius); consequently, we rejected our hypothesis that response to silviculture was not related to crown dimension. Interestingly, growth response to silviculture decreased with increasing crown volume. At this stage of stand development, crown volume is likely a surrogate for leaf area index and there is considerable evidence in the literature that leaf area and light interception drive growth (Cannell 1989; Landsberg and Sands 2011). If that is the case, this observation is in keeping with work in the literature where response to silvicultural treatment was reduced with increasing initial leaf area (Fox and others 2007). If trees in the low silviculture plots already had a large crown, it was likely that they had access to a relatively greater amount of resources that enabled them to produce a bigger crown and more leaf area. These trees would then have less response to additional resources from additional silvicultural inputs. It is important to note that large crowns do not always indicate more leaf area. Tyree and others (2009) unexpectedly found that a crop ideotype clone responded to silvicultural input (fertilization) by increasing the amount of foliage per unit crown, which led to a greater growth response to fertilizer than a competition ideotype clone.

In our analysis, crown ideotype was treated as a continuous variable. The continuous relationship we derived may be more useful than thinking of crown ideotype as a categorical variable because selected clones did not necessarily exhibit their expected ideotype. Data from the clone selection process indicated that Clone 1 and 2 were moderate crown ideotypes and Clones 3 and 4 were broad (competition) ideotypes. Our observations in the low silvicultural treatments suggest that Clone 1 and 3 are more moderate ideotypes and Clones 2 and 4 are broad ideotypes. However, the response to silviculture treatment for all genetic entries was to increase crown dimension in height (greater live crown length) and crown width. Consequently, if ideotype is used as a tool for determining the best silvicultural treatments for a specific clone, it will be important to know the conditions under which the ideotype is determined. However, if crown dimensions are used to determine 'ideotype,' the measured dimensions may act as a surrogate for resource availability up to the current stage of stand development. The potential response of the clone to additional silvicultural input would then be similar to the data in Figure 1 with less response expected for larger initial crowns.

For the genetic entries in this study, there was no crown volume by genetic entry interaction (no genetic entry effect on slope) in Figure 1. There were differences in the intercept that were related to genetic entry where Clone 4 grew better than all the other entries across the range in crown volume. In this analysis, there was no genotype by environment interaction in that all the clones responded positively to an improved

Table 1—Main and interaction effects significance levels (p values) for crown dimensional measurements at the end of the third growing season and stem volume increment for the third year for six genetic entries of loblolly pine planted in the piedmont of Virginia at three stocking levels (618, 1235, 1853 stems per hectare) and two silvicultural levels (low and high)

Effect	Diameter	Height	Live crown length	Crown width	Crown volume	Stem volume increment
Clone (C)	0.000	0.000	0.000	0.000	0.000	0.000
Stocking (D)	0.345	0.987	0.926	0.124	0.090	0.205
Silviculture (S)	0.000	0.000	0.000	0.000	0.000	0.000
C*D	0.983	0.981	0.953	0.999	0.999	0.975
C*S	0.036	0.022	0.047	0.345	0.030	0.248
D*S	0.111	0.137	0.175	0.118	0.086	0.236
C*D*S	0.982	0.915	0.934	0.990	0.997	0.996

Table 2—Genetic entry and silviculture treatment means for for crown dimensional measurements at the end of the third growing season and stem volume increment for the third year for six genetic entries of loblolly pine planted in the piedmont of Virginia at three stocking levels (618, 1235, 1853 stems per hectare) and two silvicultural levels (low and high). Means are across stocking level because stocking was not a significant effect

Genetic entry	Diameter	Height	Live crown length	Crown width	Crown volume	Stem Volume increment
	(cm)	*(m)*	*(m)*	*(m)*	*(m^3 $tree^{-1}$)*	*(cm^3 $tree^{-1}$ yr^{1})*
Low Silviculture						
Clone 1	3.5	3.0	2.8	1.1	1.1	1560
Clone 2	3.7	3.1	2.7	1.2	1.2	1554
Clone 3	3.1	2.9	2.7	1.1	1.0	1118
Clone 4	3.4	3.0	2.7	1.4	1.5	1291
Mass control pollinated	2.8	2.7	2.5	1.1	0.9	939
Open pollinated	2.5	2.5	2.3	1.2	1.0	769
High Silviculture						
Clone 1	4.3	3.4	3.1	1.5	2.3	2521
Clone 2	4.4	3.2	3.0	1.7	2.6	2551
Clone 3	4.5	3.4	3.2	1.7	2.8	2423
Clone 4	4.8	3.6	3.3	2.0	3.9	2869
Mass control pollinated	3.7	3.0	2.8	1.6	2.2	2044
Open pollinated	3.1	2.7	2.5	1.6	2.3	1692

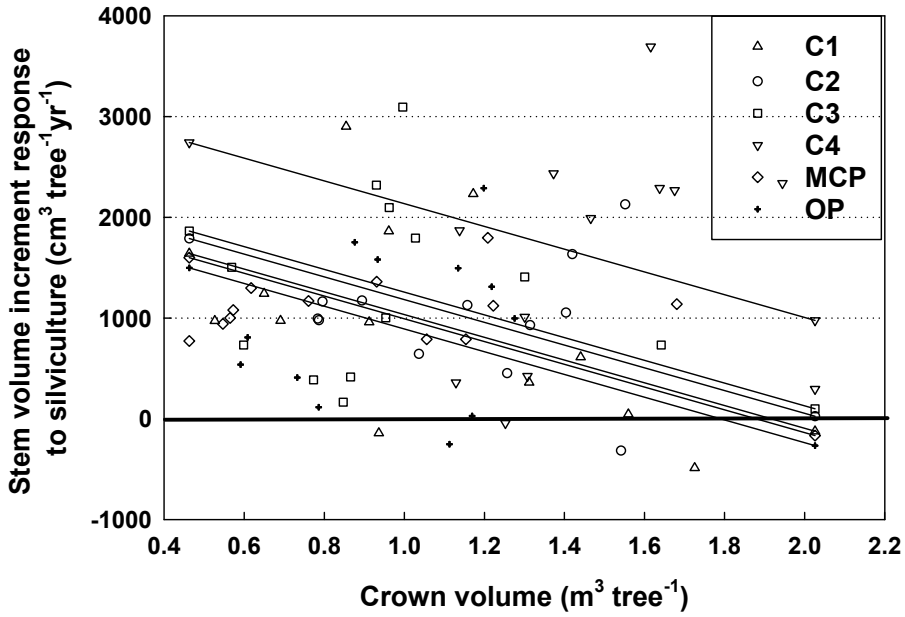


Figure 1—Stem volume incremental response to silviculture (high silviculture minus low silviculture treatments at the end of the third year of growth) versus crown volume of low silviculture trees at end of the third growing season for six genetic entries (Clones 1-4 (C1-C4), and mass control pollinated (MCP) and open pollinated (OP) families) planted in the Virginia Piedmont at three stocking levels. There was no crown volume by genetic entry interaction (no genetic effect on the slope of the relationship), but there was a significant genetic effect on the intercept, where Clone 4 had a greater intercept than all of the other genetic entries.

environment (high silviculture) although the degree of response differed by clone. However, when examining a larger population of genetic entries, this may not be the case because genotype by environment interactions have been observed with clonal material (Sierra-Lucero and others 2003).

At this point in stand development, there appeared to be relatively little above ground competition for resources (light) because even in the most densely stocked treatments the branches of adjacent trees did not overlap. This largely explains the lack of a stocking effect for any of the data reported here. However, this is not likely to be the case in the future in the 1235 and 1852 trees per hectare treatments. Anecdotal observations already indicate that branches are overlapping in these treatments. Crown closure may have a large effect on further development. Previous work indicated that the total amount of foliage (leaf area) reached an asymptote for a given level of silvicultural input when the canopy closed (Albaugh and others 2006). Crowns no longer expand outward and can only expand up. Consequently, any use of ideotype for understanding silvicultural treatment responses need to be completed before branches of adjacent trees touch.

Crown ideotype may be useful in determining the response of clonal material to silvicultural input. However, there may be some limitations noted here that may influence the utility of this method. These include: tree leaf area may not be related to crown size in all clones; information about the conditions under which ideotype is determined is required; few clones were tested here and other clones may exhibit genotype by environment interactions that were not observed here; and determination as to how the clones will respond to silvicultural input should be completed prior to crown closure. Two studies with similar designs have been installed in North Carolina and Brazil with the same genetic material and study design, which will permit additional testing to determine the robustness of the relationships developed here under a wider range in environmental conditions.

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LITERATURE CITED

Albaugh, T.J.; Allen, H.L.; Fox, T.R. 2006. Individual tree crown and stand development in *Pinus taeda* under different fertilization and irrigation regimes. Forest Ecology and Management. 234: 10-23.

Cannell, M.G.R. 1989. Physiological basis of wood production: a review. Scandinavian Journal of Forest Research. 4: 459-490.

Dickmann, D.I. 1985. The ideotype concept applied to forest trees. In: Cannell, M.G.R; Jackson, J.E., eds. Attributes of trees as crop plants. Huntingdon, England: Institute of Terrestrial Ecology: 89-101.

Fox, T.R.; Allen, H.L.; Albaugh, T.J.[and others]. 2007. Tree nutrition and forest fertilization of pine plantations in the southern United States. Southern Journal of Applied Forestry. 31(1): 5-11.

Gleed, J.A., Darling, D.; Muschamp, B.A.; Nairn, B.J. 1995. Commercial production of tissue cultured *Pinus radiata*. Tappi. 78(9): 147-150.

Gomez, K.A.; Gomez, A.A. 1984. Statistical procedures for agricultural research. 2nd ed. New York: John Wiley & Sons. 680 p.

Landsberg, J.J.; Sands, P.J. 2011. Physiological ecology of forest production: principles, processes and models. London: Academic Press. 352 p.

Li, B.; McKeand, S.E.; Allen, H.L. 1991. Genetic variation in nitrogen use efficiency of loblolly pine seedlings. Forest Science. 37(2): 613-626.

Martin, T.A.; Johnsen, K.H.; White, T.L. 2001. Ideotype development in southern pines: Rationale and strategies for overcoming scale-related obstacles. Forest Science. 47(1): 21-28.

SAS-Institute. 2002. SAS Version 9.1 TS. SAS Institute, Inc: Cary, NC.

Sierra-Lucero, V.; Huber, D.A.; McKeand, S.E. [and others]. 2003. Genotype-by-environment interactions and deployment considerations for families from Florida provenances of loblolly pine. Forest Genetics. 10(2): 85-92.

Tasissa, G.; Burkhart, H.D.; Amateis, R.L. 1997. Volume and taper equations for thinned and unthinned loblolly pine trees in cutover, site-prepared plantations. Southern Journal of Applied Forestry. 21(3): 146-152.

Tyree, M.C.; Seiler, J.R.; Maier, C.A.; Johnsen, K.H. 2009. *Pinus taeda* clones and soil nutrient availability: effects of soil organic matter incorporation and fertilization on biomass partitioning and leaf physiology. Tree Physiology. 29: 1117-1131.

Vickers, L.A.; Fox, T.R.; Stape, J.L.; Albaugh, T.J. 2012. Silviculture of varietal loblolly pine plantations: second year impacts of spacing and silvicultural treatments on varieties with differing crown ideotypes. In: Butnor, J.R., ed. 2012. Proceedings of the 16th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 363-367.

Wright, J.; Dougherty, D. 2007. Silviculture for your varietal loblolly pine plantations. Forest Landowner, May/June: 26-29.

EFFECTS OF FIRST THINNING ON GROWTH OF LOBLOLLY PINE PLANTATIONS IN THE WEST GULF COASTAL PLAIN

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The purpose of this research is to analyze thinning response in basal area and height growth of residual loblolly pine trees growing in plantations located in the West Gulf Coastal Plain. Thinning is a well-known silvicultural practice that increases the growing space available to desirable trees by removing competing trees. The response to thinning in residual loblolly pine trees is well-documented, except in the Western Gulf Coastal Plain and especially east Texas. Burrow (2001) investigated thinning response in east Texas loblolly pine plantations on a per acre basis, but not for individual trees. In order to quantify the effects of first thinning on individual planted loblolly pine trees, basal area and height pre- and post-thinning growth was measured on 11 permanent growth and yield plots within 3 years of thinning. These plots are part of the East Texas Pine Plantation Research Project and are located in east Texas and western Louisiana. Treatment plots are 0.23-acre squares (100 by 100 feet) and are located across a range of soil types, soil drainage classes, and site preparation practices that characterize intensively managed plantations in the West Gulf region. Trees were measured for diameter, total height, live crown height, crown class, fusiform rust incidence, and damage. There are a total of 126 of these plots, but only these 11 have been thinned, though the others will eventually receive an operational thinning. The landowners typically schedule a plantation to be thinned when it reaches a stand density index target of one-half of the maximum stocking, which corresponds to a 225 stand density index (SDI). The thinning prescription is an operational geometric (row) and from below (low) thinning where the removed trees are selected by the operator and not marked (operator select). The intent of this research is to document the thinning response in operationally thinned plantations. The growth responses, individual tree basal area (square feet) and total height (feet), were measured on 507 survivor trees in the 11 plots (survivor trees are residual trees in the post-thinned plantation). Based on these measurements, the average pre-thin basal area per acre is 117 sq. ft. and the average post-thin basal area per acre is 68 sq. ft. The average trees per acre is 580 and 226 for pre- and post-thin, respectively. The average dbh in inches is 6.0 and 7.4 for pre- and post-thin, respectively. The average tree height in feet is 41 and 48 for pre- and post-thin, respectively. The average total age in years is 11 and 14 for pre- and post-thin, respectively. These results show that using an operational thinning prescription, the average dbh increased 1.4 inches and the average height increased 7 feet. Thus, operator select did achieve a diameter lift and a removal of the shorter, overtopped trees in the thin. Operator select also achieved on average the target 226 tpa or almost exactly one-half the maximum SDI as well as a basal area per acre of 68 sq. ft. for these 11 plots. We conclude that operator select in these plantations works well to achieve target thinning prescriptions. Analysis shows that thinning response in both basal area and height growth of individual trees is detectable as early as two years after thinning, and separation in response increases as years since thinning (YST) increases. Thus, thinning response is detectable in this small sample in a relatively short time period. Further growth analysis shows that tree basal area increases in response to thinning, but this effect decreases as YST increases, presumably because the thinned plantation will approach its unthinned counterpart over time (Pienaar and Rheney 1995). The data collected for this study are from operationally thinned plantations, and more data will become available as more plots are thinned. Results from this study will be compared to data from a new experimentally designed thinning study. This new study is designed to measure thinning response of first-thinned east Texas loblolly pine plantations at three controlled levels of stocking based on SDI targets, develop a thinning response model for first-thinned loblolly pine plantations in the West Gulf region, and evaluate the new thinning response model with other regional thinning models. We would like to thank Campbell Global, Rayonier, Resource Management Services (RMS), Hancock Forest Management, the McIntire-Stennis program, and Stephen F. Austin State University for their support of this research as well as all the ETPPRP student workers who helped collect the data over the years.

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LITERATURE CITED

Burrow, A.M. 2001. Predicting basal area growth and yield of thinned loblolly pine (*Pinus taeda*, L.) plantations in the West Gulf Region. Nacogdoches, Texas: Stephen F. Austin State University. 81 p. M.S. thesis.

Pienaar, L.V.; Rheney, J.W. 1995. Modeling stand level growth and yield response to silvicultural treatments. Forest Science. 41(3): 629-638.

Longleaf 1

Moderator:

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EARLY DENSITY MANAGEMENT OF LONGLEAF PINE REDUCES SUSCEPTIBILITY TO ICE STORM DAMAGE

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Abstract—The Pax winter storm of February 2014 caused widespread damage to forest stands throughout the southeastern U.S. In a long-term study of savanna plant community restoration at the Savannah River Site, Aiken, SC, precommercial thinning (PCT) of 8- to 11-year-old plantations of longleaf pine (*Pinus palustris*) in 1994 reduced their susceptibility to stem bending and breakage from the storm 20 years later, despite the occurrence of an intensive commercial thinning just two years before the storm. Tree mortality in areas that had received PCT (25 percent) was less than where PCT had not occurred (40 percent). Incidence of heavy injury (e.g., crown loss of 75-99 percent, stem bent greater than 45 degrees) also was less with PCT (11 percent) than without PCT (16 percent).

INTRODUCTION

Prescribed fire and overstory density management are being used to restore and maintain the highly diverse savanna communities typical of longleaf pine (*Pinus palustris*) forests of the southeastern U.S. (Kilgo and Blake 2005). Without density management, fully-stocked stands of longleaf pine will competitively exclude understory herbaceous species (Mulligan and others 2002, Harrington 2011). However, timing of density management is critical because overstocked conditions can decrease tree stability (Oliver and Larson 1996). Differences in stand density do not directly influence susceptibility of southern pines to storm damage (Amateis and Burkhart 1996), but rather they result in changes in tree allometry (i.e., crown height, crown length, and stem slenderness) that impart greater risk when the stand is opened up by a subsequent disturbance, such as thinning. Recently thinned stands are particularly susceptible to toppling and breakage from ice storms (Bragg and others 2003).

This paper describes how density management treatments influenced the severity of ice-storm damage in a long-term study of longleaf pine savannas on the Savannah River Site. The Pax winter storm occurred during February 11-13, 2014 and affected most of the southeastern U.S. Glaze ice thickness was up to 1.25 inches in an area roughly centered over the Savannah River Site (National Weather Service 2014).

METHODS

The study was conducted at the Savannah River Site, a National Environmental Research Park near Aiken, SC. Detailed methodology and previous results are described in Harrington (2011). Study sites were selected to have fully stocked stands of longleaf pine with mid-story hardwoods. Six 8- to 11-year-old plantations of longleaf pine were located during winter 1993-94 having average stem densities of 600 pines and 240 hardwoods per acre. Soils include loamy sands of the Blanton, Lakeland, or Troup series with low to very low available water-holding capacities (Rogers 1990). Each site was divided into four treatment plots 7 to 17 acres in area. One of the following treatments was randomly assigned to each plot:

1. Nontreated.
2. Pine thinning. In May 1994, pines were precommercially thinned to leave a uniform spacing of trees at approximately half of the original stem density.
3. Woody control. In 1995 and 1996, several herbicide treatments were applied to virtually eliminate all non-pine woody vegetation.
4. Combined treatments of pine thinning and woody control.

The experimental design is a randomized complete block with six replications (sites) of the four treatments arranged as a 2 x 2 factorial (i.e., presence versus absence of pine thinning and woody control). Within each of the 24 treatment plots, 10 sample points were systematically located for measuring tree growth and injury. Convergence of pine crown cover in thinned and nonthinned treatments (Harrington 2011) prompted the need for a commercial thinning in early 2012 to

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rejuvenate the savanna communities. Treatments 2 and 4 were thinned to leave a pine basal area of about 40 square feet per acre (BA40), and treatments 1 and 3 were thinned to leave about 70 square feet per acre (BA70). To reduce fuel accumulations and abundance of non-pine woody vegetation, dormant-season prescribed fires were applied to each site every 3 to 5 years.

In September 2010, each pine located within 19.7 feet of a sample point was measured for diameter at breast height (dbh; nearest 0.04 inch at 4.5 feet aboveground). Following the commercial thinning in 2012, the radius of sampled area around each sample point was increased to maintain adequate tree sample sizes. In September 2012, each longleaf pine located within 29.3 feet of a sample point was tagged and measured for dbh. Stem diameter measurements were repeated in September 2013. In May 2014, each tagged pine was assessed for glaze-ice damage from the Pax winter storm. Trees were classified according to their degree of stem bending (less than 15 degrees, 15 to 45 degrees, or greater than 45 degrees), crown loss (0, 1-24, 25-49, 50-74, 75-99, or 100 percent), stem breakage (none, broken top, or broken stem), and uprooting (present or absent). Using these data, each tree was assigned to one of the following five injury classes:

1. None. No visual injury from the ice storm or stem bent less than 15 degrees.
2. Light. Crown loss 1-24 percent or stem bent 15 to 45 degrees.
3. Moderate. Crown loss 25-74 percent or presence of a broken top.
4. Heavy. Crown loss 75-99 percent, stem bent greater than 45 degrees, presence of a broken stem, or presence of uprooting.
5. Dead. Crown loss 100 percent.

All statistical analyses were limited to the longleaf pine component of the study. Loblolly pine (*Pinus taeda*) comprised about 4 percent of pine stem density and these data were excluded from the analyses. Stem density and stand basal area of longleaf pines were calculated for each sample point and then averaged by treatment plot. Each density variable was subjected to repeated measures analysis of variance (ANOVA) in SAS (SAS Institute, Inc., 2013) to test for fixed effects of thinning level, woody control level, year (2010, 2012, and 2014), and their interactions after adjusting for random effects of blocking. A natural logarithmic transformation was applied to each variable prior to ANOVA to improve the homogeneity of the residual variances. Multiple comparisons of treatment means were conducted with Tukey's HSD test using Bonferroni-adjusted probabilities to reduce the likelihood of Type I errors. All statistical tests were conducted with an alpha level of 0.05.

Injury class data for each treatment plot were expressed as a proportion of stand density (i.e., stem density or stand basal area) to remove inherent differences between thinning levels (after Amateis and Burkhart 1996). For each injury class, a mixed-model ANOVA was conducted on each density variable to test for fixed effects of thinning level, woody control level, and their interaction after adjusting for random effects of blocking. An angular transformation (arc-sine, square-root) was applied to each variable prior to ANOVA to improve the homogeneity of the residual variances. To illustrate treatment effects on longleaf pine damage, back-transformed, least-squares means for proportionate stem density and proportionate stand basal area by injury class were re-expressed in the original units of trees per acre and square feet per acre, respectively.

RESULTS

A total of 1,942 longleaf pines were included in the ice damage assessments. In 2010, 16 years after the precommercial thinning treatment, stem density of longleaf pines was greater in nonthinned stands than in thinned stands (353 versus 212 trees per acre, respectively), but stand basal area did not differ because of convergence of the two treatments (table 1). The commercial thinning of 2012 re-established the desired differences in stand density between the two thinning levels. However, mortality of longleaf pines from the Pax winter storm in 2014 was greater in BA70 (40 percent of stems) than in BA40 (25 percent of stems) (fig. 1), and as a result, stand basal area no longer differed between the two treatments.

For both stem density and stand basal area, proportionate injury differed between BA40 and BA70 for the following injury classes: none, light, heavy, and dead (fig. 1). There was a higher proportion of trees in BA70 than in BA40 for the non-injured, heavily injured, and dead classes; however, there was a lower proportion of trees for the light injury class. In addition, mortality of longleaf pines was lower in plots that received woody control (29 percent) than in those that did not receive the treatment (36 percent).

DISCUSSION AND CONCLUSIONS

In this long-term study of longleaf pine savanna restoration, precommercial thinning at ages 8 to 11 years resulted 20 years later in contrasting differences in plantation susceptibility to glaze ice damage. In the fourteenth year of the study (2008), crown height was 1.5 m greater and crown length was 1.1 m less in nonthinned stands than in thinned stands (Harrington 2011), resulting in greater top-heaviness and less overall stability for nonthinned stands.

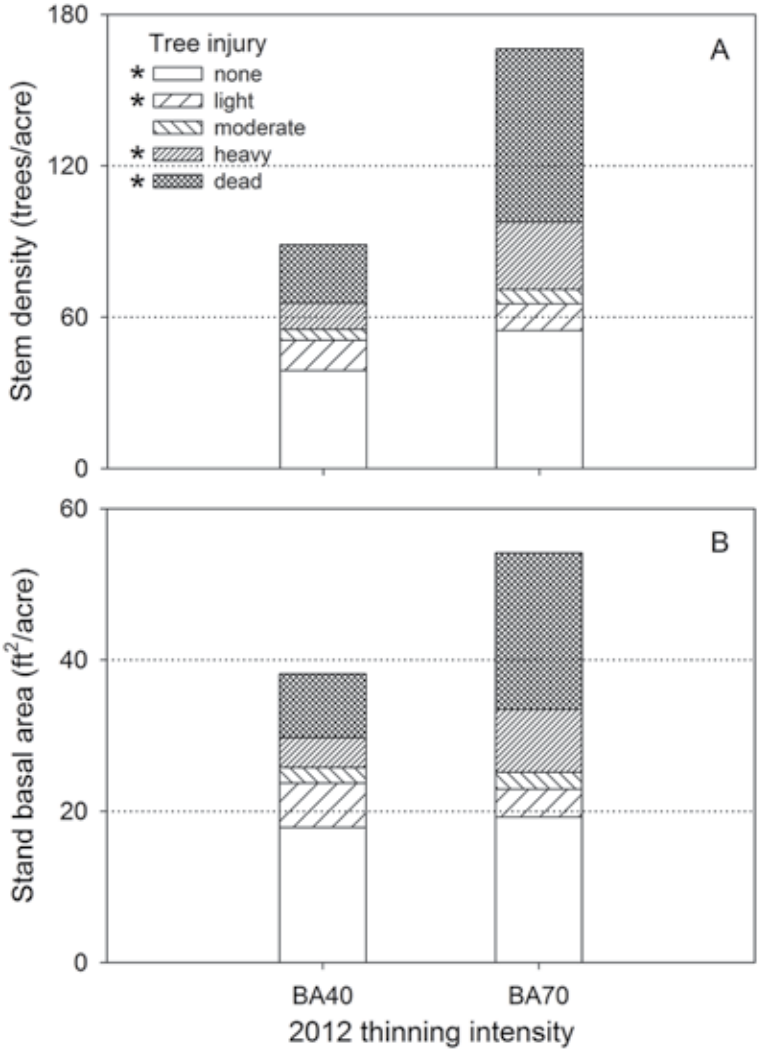


Figure 1—Effects of 2012 thinning intensity on: (a) stem density and (b) stand basal area of 28- to 31-year-old longleaf pine plantations by tree injury class three months after the February 2014 Pax ice storm at the Savannah River Site, Aiken, SC. For both stem density and stand basal area, asterisks (*) in the legend indicate injury classes that differed significantly (P<0.05) between thinning intensities.

The commercial thinning of 2012 opened up both thinned (BA70) and nonthinned stands (BA40) sufficiently to render them susceptible to stem bending and breakage from an ice storm. The Pax winter storm occurred just two years after the commercial thinning when the stands were still in a vulnerable stage. Had the storm occurred four or more years after thinning, injury and mortality likely would have been much less because the trees would have had sufficient time to become more stable with increased growth in stem diameter and slowed upward crown recession.

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LITERATURE CITED

Amateis, R.L.; Burkhart, H.E. 1996. Impact of heavy glaze in a loblolly pine spacing trial. Southern Journal of Applied Forestry. 20: 151-155.

Bragg, D.C.; Shelton, M.G.; Zeide, B. 2003. Impacts and management implications of ice storms on forests in the southern United States. Forest Ecology and Management. 186: 99-123.

Table 1—Changes in stand density of longleaf pine plantations from 2010 to 2014 at the Savannah River Site, Aiken, SC. For a given variable and date, numbers followed by the same letter do not differ significantly (P<0.05)

		Date[a]		
Variable	Treatment[b]	September 2010	September 2012	May 2014
Stem density (trees per acre)	Nonthinned/BA70	353 a	168 a	92 a
	Thinned/BA40	212 b	92 b	66 b
Stand basal area (square feet per acre)	Nonthinned/BA70	88 a	54 a	32 a
	Thinned/BA40	72 a	38 b	30 a

[a] The second thinning occurred in early 2012 and the Pax winter storm occurred in February 2014.

[b] "Nonthinned" and "thinned" indicate the level of precommercial thinning that occurred in 1994, and "BA70" and "BA40" indicate the intensity of commercial thinning that occurred in 2012.

Harrington, T.B. 2011. Overstory and understory relationships in longleaf pine plantations 14 years after thinning and woody control. Canadian Journal of Forest Research. 41: 2301-2314.

Kilgo, J.C.; Blake, J.I., eds. 2005. Ecology and management of a forested landscape: fifty years on the Savannah River Site. Washington, DC: Island Press. 479 p.

Mulligan, M.K.; Kirkman, L.K.; Mitchell, R.J. 2002. *Aristida beyrichiana* (wiregrass) establishment and recruitment: implications for restoration. Restoration Ecology. 10: 68-76.

National Weather Service. 2014. February 2014 significant snow and ice storm. http://www.weather.gov/cae/Feb2014SnowIceEvent.html [Date accessed: April 3, 2015].

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. Update ed. New York: John Wiley and Sons, Inc. 520 p.

Rogers, V.A. 1990. Soil survey of Savannah River Plant area, parts of Aiken, Barnwell, and Allendale Counties, South Carolina. Washington, DC: U.S. Department of Agriculture Soil Conservation Service. 127 p.

SAS Institute, Inc. 2013. The SAS System for Windows, version 9.4. Cary, North Carolina.

AIR LATERAL ROOT PRUNING AFFECTS LONGLEAF PINE SEEDLING ROOT SYSTEM MORPHOLOGY

Shi-Jean S. Sung and James D. Haywood[1]

Abstract—Longleaf pine (*Pinus palustris*) seedlings were cultured with air lateral root pruning (side-vented containers, VT) or without (solid-walled containers, SW). Seedling root system morphology and growth were assessed before planting and 8 and 14 months after planting. Although VT seedlings had greater root collar diameter than the SW before planting, seedling height and ground line diameter were not affected by root pruning 14 months after planting. Root pruning did not affect seedling biomass. However, root pruning changed the patterns of biomass allocation. Before planting, VT seedlings allocated a greater percentage of biomass to roots and less to shoots than SW seedlings. Furthermore, VT seedlings allocated more root biomass to taproots at the expense of the first-order lateral roots (FOLR) and fine roots. This trend of favoring taproots by VT seedlings continued through 14 months after planting. For both types of seedlings, more than 50 percent of the FOLR originated in the top 2.5 cm of the taproot whereas greater than 70 percent of FOLR egressed below 7.5 cm of the root plug after planting. Fourteen months after planting, SW seedlings accumulated greater root biomass within the dimensions of the original root plug than VT seedlings. The VT seedlings had less extent of FOLR spiraling within the root plug than the SW seedlings before and after planting. Less FOLR spiraling and less root biomass increase within the root plug after planting in the VT seedlings may improve the physical stability of the VT longleaf pine saplings.

INTRODUCTION

Since the late 1980s, public, industrial, and private forest managers and land owners have been actively restoring the longleaf pine (*Pinus palustris* Mill.) ecosystem in the southeastern United States (Barnett 2002, Boyer 1989, Landers and others 1995). Container-grown longleaf pine seedlings are the preferred planting stock type because they have a higher first-year survival rate and a wider planting window than bareroot seedlings (South and others 2005 and references cited therein). For the 2005-2006 planting season in the southern United States, 70 percent of the longleaf pine seedlings produced was of the container type (McNabb and Enebak 2008). The demand for container stock longleaf pine continues to increase. However, one noted drawback of using container-grown seedlings for reforestation and afforestation is that the established saplings may succumb to wind-throw during high sustained winds (South and others 2001).

Improvements in the morphological quality of container stock root systems have been attempted by adding ridges to the container cavity or coating the cavity wall with a layer of low concentration copper compounds (Cu). Cavity ridges help reduce root spiraling by training primary lateral roots to grow vertically (Barnett and Brissette 1986). Slow release of low-concentration Cu from the cavity coating stops lateral roots from elongating once they reach the cavity wall (Ruehle 1985). Copper root pruning increased taproot and secondary lateral root dry weights at the expense of primary lateral root dry weight in container-grown longleaf pine seedlings before planting (Sayer and others 2009, 2011). These Cu-grown longleaf pine seedlings were greater in size 5 years after planting than the non-Cu seedlings (Haywood and others 2012). Dumroese and others (2013) also reported that longleaf pine cultured in Cu cavities had greater root collar diameter and tap root biomass but less total root volume compared to the non-Cu seedlings. In a root growth potential test, longleaf pine seedlings cultured in Cu cavities produced more new roots than those grown in non-Cu cavities or bareroot seedlings (South and others 2005). Lodgepole pine (*P. contorta* Dougl.) grown in Cu cavities had fewer leaning seedlings 3 years after planting than those from the non-Cu cavities (Krasowski 2003).

In spite of these reported improvements in seedling root system architecture and field performance by Cu root pruning, reluctance to growing or planting Cu seedlings in forest industry still persists. It appears that this reluctance stems from the operational aspects of reforesting longleaf pine. For example, Cu seedling root plugs are less firm than the non-Cu root plugs; thus, harder to handle from lifting to planting (Sayer and

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others 2009). Furthermore, the Styrofoam Cu containers cost 2 cents more per cavity than the non-Cu Styrofoam ones. Compared to the plastic containers, the Styrofoam containers are two- to three-fold less expensive; but they have a shorter usable life (3 versus 10 years) and are harder to clean for reuse. Since it is impossible to coat the cavities of plastic containers with Cu, an alternative to the chemical pruning of lateral roots to improve root system architecture in the root plug is air pruning of the lateral roots. Here, we tested the effects of air lateral root pruning using side vents on plastic cavity walls on longleaf pine seedling stock quality and field performance, with emphasis on root system morphology. Seedlings cultured in plastic, solid-walled containers were used as the non-lateral root pruning control.

MATERIALS AND METHODS

Container-Grown Seedlings

Longleaf pine seedlings were obtained from two nurseries in early November 2013. Seed sources for these two groups of seedlings were not the same, but both were from Louisiana. The State Nursery in Monroe, LA, grew longleaf pine seedlings for 30 weeks without air lateral root pruning in plastic, solid-walled IPL Rigi-pots® IP110 cone-shaped containers (SW) with dimensions of 3.95 cm-12.7 cm-110 ml (cavity top diameter-depth-volume) and a density of 569 cavities m^{-2}. The International Forest Genetics & Seed Company (Moultrie, GA) grew longleaf pine seedlings for 31 weeks with air lateral root pruning in the Size 128 vented containers (VT) with dimensions of 3.8 x 3.8 cm-12.0 cm-110 ml (square top opening-depth-volume) and a density of 309 cavities m^{-2}. The interior 84 cavities (14 x 6) in the Size 128 container trays have vents on all sides with the top tier starting 6.0 cm from the cavity top and the bottom tier starting at 9.5 cm. Cavities on the outside of the container trays have vents on the sides facing inside. Both SW and VT containers have cavity ridges.

Field Experiment

The study site was on the Palustris Experimental Forest within the Kisatchie National Forest in Rapides Parish of central Louisiana (31°01´N, 92°37´W). The soils were a Malbis fine sandy loam (fine-loamy, siliceous, subactive, thermic Plinthic Paleudults), a Ruston fine sandy loam (fine-loamy, siliceous, semiactive, thermic Typic Paleudults), and a Smithdale sandy loam (fine-loamy, siliceous, subactive, thermic Typic Hapludults). In November 2013, all longleaf pine seedlings were planted on the same day and within 3 days of receipt from the nurseries. In each of 4 plots, 6 rows were planted with 13 seedlings per row. Within each plot, three rows were randomly assigned to one of the two container types. Seedlings were planted with a Terra Tech Styro 5® dibble stick at 2 x 2 m spacing.

Laboratory and Field Assessments

Before planting, 20 seedlings of each root pruning treatment were randomly selected for destructive assessments of chlorophyll content, biomass, and root system morphology. Seedling root collar diameter and height were measured on all seedlings immediately prior to planting. Eight and 14 months after planting, one seedling from each row was randomly selected for the same destructive assessments as the pre-planting seedlings. The selected seedlings were excavated within a 30-cm radius of the stem and to a 30-cm depth in soil. Survival, height and ground line diameter were measured for the remaining seedlings in the field in January 2015.

Chlorophyll determination—Two needle fascicles per seedling were randomly selected from seedlings selected for pre-planting destructive sampling and for excavated seedlings. Needle surface area was measured with a displacement method by Johnson (1984). Chlorophylls a and b were extracted with N,N-dimethylformamide and the absorbance of the extract was read at 664 nm and 647 nm by a DU-70 spectrophotometer (Beckman Coulter Inc., Fullerton, CA) as described in detail by Sung and others (2010).

Biomass and root system morphology—Seedlings selected for pre-planting destructive sampling and the excavated seedlings were severed at the root collar. Growth variables, such as height, root collar diameter, and dry weights of needles, stems, and root systems were recorded. Root systems were carefully washed to remove growth media and soils. Root system morphology was assessed by placing a root system over a root plug template of corresponding container type. For seedlings before planting, only those primary lateral roots originating from the taproot, sturdy in structure, and with a diameter of at least 0.5 mm were counted as the first-order lateral roots (FOLR). For seedlings excavated 8 and 14 months after planting, primary lateral roots originating from the taproot within the original root plug and with a diameter of at least 0.9 mm were counted as the FOLR. For each FOLR, the depth of origination from the taproot, the depth of egress from the root plug, and the length within the root plug were measured.

To estimate the spiraling extent within the root plug for each FOLR, the portion of an FOLR between its originating point and first point of contact with the cavity wall was measured. This point was visually determined by finding the first deflection point on an FOLR where the orientation of the FOLR changed due to contact with the wall. The extent of FOLR spiraling within a root plug was estimated as follows:

For the FOLR that extended to the bottom end of the root plug:

Estimated FOLR spiraling extent (cm) = L - DL - (RP-DP) [equation 1],

where

L (cm) = total length of a stretched FOLR, measured from the origination point on the taproot to the end of the FOLR within a root plug;

DL (cm) = deflection length, measured between the origination point of a stretched FOLR and its first contact point with the cavity wall;

RP (cm) = root plug length;

DP (cm) = depth of the first deflection point of a FOLR on the root plug.

For a FOLR that probably would egress (for seedlings before planting) or had egressed from the side of a root plug (for the excavated seedlings), [equation 2] was used.

Estimated FOLR spiraling extent (cm) = L - DL - (EG-DP) [equation 2],

where

EG (cm) = depth of FOLR egress from the side of a root plug

Statistical Analysis

The study was a generalized randomized block design with four blocks each containing three replicates of two lateral root pruning treatments (VT and SW). Each of the replicates was a randomly positioned row of 13 seedlings with the same root pruning treatment. For analysis of non-destructive variables such as ground line diameter and height, the means of all surviving trees in a row were used in an analysis of variance (SAS Institute Inc. 2004). All of the destructive sampling variables such as dry weight and root system morphology variables were analyzed using a single subsample from each row. At planting time, 20 samples of all available seedlings for each root pruning treatment were selected and compared using a *t*-test. Significant treatment effects existed when Pr > *F* (for the planted seedlings) or Pr > *t* (for the not-planted seedlings) values were < 0.05.

RESULTS AND DISCUSSION

Seedling Field Performance

Fourteen-month field survival of the SW and VT seedlings was 93 and 96 percent, respectively. A high survival rate is expected in container longleaf pine stock cultured in container cavities greater than 90 ml in volume (Dumroese and others 2009, Sung and others 2010). Lateral root pruning treatment did not significantly affect seedling height after 14 months in field with 5.3 cm (standard deviation (sd)= 0.8) for VT and 5.7 cm (sd= 0.6) for SW seedlings. No planted seedlings had emerged from the grass stage. The VT seedlings had significantly greater root collar diameter at planting than the SW seedlings [8.6 mm (sd= 0.5) versus 7.2 mm (sd= 0.4)]. However, seedling ground line diameter did not differ between treatments 14 months after planting with 19.2 mm (sd= 1.9) for SW and 18.9 mm (sd= 1.9) for VT. These values were about 5 mm greater than the longleaf pine seedlings cultured in Cu and non-Cu cavities of comparable volume and 1 year after planted in central Louisiana (Sayer and others 2009).

Seedling Morphology and Biomass Allocation

Compared to SW seedlings, VT seedlings had a higher chlorophyll a and b content at planting, but not after 8 or 14 months in field (table 1). Fourteen months after planting, the chlorophyll content had doubled in SW seedlings and was within the range reported for second year planted longleaf pine seedlings (Sung and others 2010). The chlorophyll content may further increase in second year because needles were sampled in January. A decrease in chlorophyll content from 25.6 nmol cm^{-2} in November to 14.5 nmol cm^{-2} in December was found in second year field longleaf pine seedlings in central Louisiana (Sung and others 2010). Increases in needle chlorophyll content and total needle biomass in seedlings after 14 months in field (table 1) suggested that the root systems of these planted longleaf pine seedlings had acquired mineral nutrients from the soils to sustain the growth of new needles.

Seedling biomass increased more than 3-fold after 8 months in the field and almost doubled again during the subsequent 6 months for both SW and VT seedlings (table 1). Similar to the ground line diameter, seedling biomass was 5 to 11 g greater in this study than seedlings in the study of Sayer and others (2009). Seedlings in this study were cultured for 30 or 31 weeks whereas a culture period of 27 weeks was used by Sayer and others (2009) which resulted in smaller size seedling stock. This may partially explain the greater biomass accumulation in planted seedlings in this study. Although air lateral root pruning treatment did not affect seedling ground line diameter or total seedling biomass, it affected the biomass allocation patterns. Compared to VT seedlings, SW seedlings allocated more biomass to needles and less to root systems before planting and less to root systems 8 months after planting (table 1). This was in contrast to the results report by Sayers and others (2009). They did not observe differences in biomass allocation patterns between the Cu root pruning and non-Cu longleaf pine seedlings.

Root System Morphology and Root Biomass Allocation

The VT seedlings had greater root weight than the SW seedlings before planting (table 2). However, after 8 and 14 months in the field, total root weights were similar between these seedlings. The VT nursery stock allocated more biomass to the taproot and less to the

Table 1—Effects of side-vent air lateral root pruning on growth variables (mean±standard deviation) of container-grown longleaf pine seedlings sampled before planting (*n*=20) and 8 and 14 months after planting (*n*=12)

Type	Chlorophyll a and b	Root collar diameter[a]	Height	Seedling dry weight	Biomass allocated to Needles	Stems	Roots
	nmol cm^{-2}	*mm*	*cm*	*g*	----------%----------		
Before planting (November 2013)							
SW	8.4±2.2*[b]	6.9±1.3*	3.2±0.5*	4.3±1.1	62±6*	10±2*	28±5*
VT	11.4±1.4	7.9±1.0	2.9±0.5	3.9±0.8	41±6	16±3	43±6
Eight months after planting (July 2014)							
SW	9.9±4.9	14.4±1.9	3.8±0.8*	16.6±5.2	60±5	13±3	27±4*
VT	9.1±4.9	14.2±2.3	3.1±0.7	14.1±5.5	56±9	13±3	31±7
Fourteen months after planting (January 2015)							
SW	18.5±4.0	16.2±1.8	6.1±1.5	30.8±10.0	52±4	9±3	39±6
VT	15.7±2.9	16.0±2.3	6.0±1.7	27.9±11.8	54±4	10±2	36±3

Note: seedlings were cultured in side-vented containers (VT) for 31 weeks in Georgia or in solid-walled containers (SW) for 30 weeks in Louisiana. Seedlings were planted in Louisiana in November 2013.

[a] Ground line diameter for planted seedlings.

[b] For each variable within a sampling period, values associated with * have Pr > F or PR > *t* values of < 0.05.

FOLR compared to the SW stock (table 2). This trend of favoring taproot growth in VT seedlings continued through 14 months after planting (table 2). Apparently, lateral roots of the VT seedlings were air pruned during nursery culture. It was reported that longleaf pine seedlings cultured in Cu-coated cavities also showed greater root biomass allocation to taproots and secondary lateral roots at the expense of primary roots (Dumroese and others 2013, Sayer and others 2009). Although air pruning decreased the number of FOLR in VT stock at the time of planting, VT seedlings had a similar number of FOLR as the SW seedlings after 14 months in the field (table 2). There were block and container type interactions for FOLR number, diameter, and percent of biomass allocation to FOLR in seedlings 8 months after planting. Compared to the VT seedlings, SW seedlings had greater FOLR number, diameter, and biomass allocation except for seedlings in one block.

Air lateral root pruning treatment did not affect the depths of the FOLR origination from the upper 2.5 cm (table 2) or 5.0 cm (data not shown) of the taproots. More than 50 and 80 percent of the FOLR originated, respectively, from the upper 2.5 cm and 5.0 cm of the taproots. This pattern of FOLR origination was similar to the one found in container-grown longleaf pine seedlings 3 years after planting (Sung and others 2009). It was reported that 59 and 35 percent of FOLR originated in the upper 5.0 and 10.0 cm of taproots, respectively. The high density of FOLR near the top of the taproot cautions against a shallow planting strategy for container-grown longleaf pine stock. Seedlings planted too shallow would inadvertently leave the upper portion of the FOLR exposed after the growth media disintegrated and render these roots at risk when a prescribed burn is introduced to the planting site later.

The spiraling estimating system used here showed that VT seedlings had less FOLR spiraling within the root plug even after 14 months in field (table 2). Although the cavity ridges trained most FOLR to elongate vertically within the root plug for both seedling types, the ridges did not eliminate FOLR spiraling completely. For example, a SW and a VT seedling with similar total root biomass had 3.7 cm and 1.2 cm FOLR spiraling, respectively (fig. 1).

Treatments did not affect the percent of FOLR egress below 7.5 cm of the root plug (table 2). More than 70 percent of FOLR egress below this zone (table 2). This result is in contrast to other studies that found root pruning using Cu coating increased the proportion of FOLR egress from the top portion of the cavity plug, For example, Sayer and others (2009) reported that Cu and non-Cu longleaf pine seedlings had, respectively, 46 and 19 percent of lateral roots egressed from the top 5 cm of the root plug. Sung and others (2009) showed that Cu-root pruning increased the percent of FOLR egress from the top 5.0 cm of the root plug from 26 to 48 percent in longleaf pine seedlings. In other words, the low percent of FOLR egress from the upper half of the root plug observed in the current study without a significant treatment effect indicates that air lateral root pruning only provided a limited improvement in the root system architecture. Sayer and others (2009) found a greater effect on first-year Cu-seedling growth by roots egressing into the top 5 cm of soils than those egressing from the bottom depth of root plug

and suggested these roots near the soil surface may increase soil resource acquisition and sapling vertical stability.

Both taproot biomass and number of FOLR number are important for seedling quality and field performance for many tree species (Kormanik and others 1998). Fourteen months after planting, both types of seedlings hade similar number of FOLR (table 2) and seedling dry weight (table 1). After 14 months in the field, biomass of SW seedling FOLRs within the confines of the original root plug increased 11.89 g (from 0.43 g to 12.32 g);

Table 2—Root system characteristics (mean±standard deviation) of longleaf pine seedlings in response to side-vent air lateral root pruning during nursery culture

Type	Total root system	All egressed roots	Root biomass allocated to taproot	FOLRs[a]	FOLR diameter	Root biomass allocated to FOLRs	FOLRs Originating Upper 2.5 cm	FOLR Egress Below 7.5 cm	FOLR Spiral Length
	----------g----------		*%*	*no.*	*mm*	*----------%----------*			*cm*
Before planting (November 2013)									
SW	1.19±0.37*[b]	-	26±8*	6.4±2.5*	0.79±0.17	10±5*	64±23	-	1.6±1.2*
VT	1.67±0.33	-	51±9	4.5±2.7	0.82±0.19	6±4	75±23	-	0.9±0.8
Eight months after planting (July 2014)									
SW	4.41±1.48	0.56±0.46	51±7*	10.5±2.6*	2.11±0.30*	31±8*	56±22	90±10	3.8±2.3*
VT	4.19±1.06	0.62±0.32	74±10	8.3±2.1	1.78±0.20	15±7	65±12	84±18	0.9±0.6
Fourteen months after planting (January 2015)									
SW	12.13±4.32	3.12±1.75	47±9*	9.9±2.7	3.39±0.51	40±8*	52±13	79±12	4.5±2.4*
VT	9.73±3.53	3.08±1.99	61±14	8.7±3.5	3.04±0.67	28±15	56±15	71±23	2.1±1.1

Note: seedlings were sampled before planting (n=20) and 8 and 14 months after planting (n=12). Seedlings were cultured in side-vented containers (VT) or in solid-walled containers (SW).

[a] FOLRs = first-order lateral roots.

[b] For each variable within each sampling period, values associated with * have Pr > F or Pr > t values of <0.05.

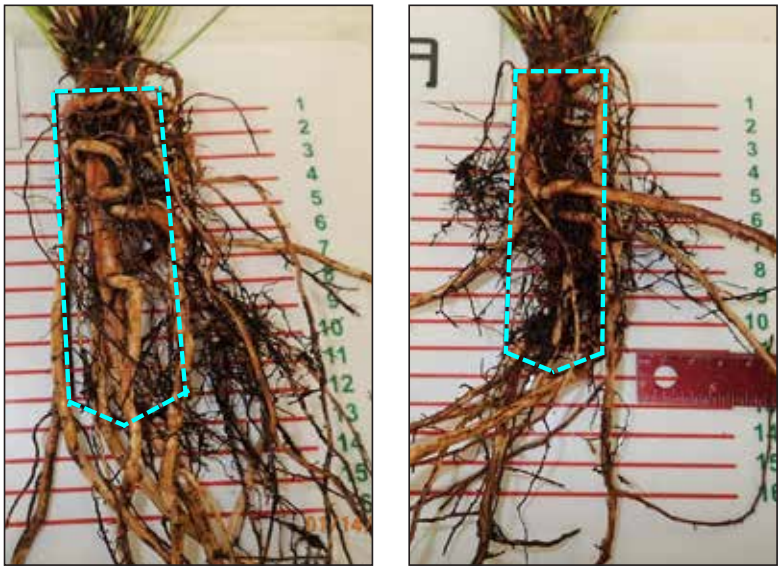


Figure 1—(L) Root system of a longleaf pine seedling cultured in a solid-walled container cavity and excavated after 14 months in the field. Most of the first-order lateral roots egressed from the bottom of the confines of the original root plug (dashed blue line) and were spiraling. (R) Root system of a longleaf pine seedling cultured in a side-vented container cavity and excavated after 14 months in the field. Most of the first-order lateral roots egressed from the bottom of the confines of the original root plug (dashed blue line) and were not spiraling.

whereas biomass of VT seedling FOLRs within the root plug increased 7.58 g (from 0.23 g to 7.81 g. The faster growing and sometimes spiraling FOLR within the SW seedling root plug can exacerbate the FOLR spiraling problem and these FOLR may strangulate each other or the taproot later. The issue of longleaf pine sapling's vertical instability has been shown to be associated with root system architecture (such as lateral root spiraling and absence of taproot) and not necessarily with shoot growth (Sung and others 2009, 2012).

CONCLUSIONS

Longleaf pine seedlings cultured with air lateral root pruning in side-vented containers increased biomass allocation to the root system, especially to the taproot compared to those cultured without lateral root pruning in the solid-walled containers even after 14 months in the field. Furthermore, seedlings with root pruning had less extent of FOLR spiraling than the non-root pruning seedlings through 14 months in the field. Seedling root plugs from the vented containers did not create any difficulty for planting in this study. Since these seedlings were grown commercially, the extracting and handling at the nursery should be problem free. The air pruning treatment can be a solution to ease the reluctance to growing or planting longleaf pine seedlings cultured with chemical root pruning treatment such as Cu. Unlike the Cu root pruning, the treatment of air pruning of lateral roots did not change the vertical pattern of FOLR egress from the root plug after planting. More than 70 percent of the FOLR egressed into soil at a depth of greater than 7.5 cm. Further study is needed to test sapling vertical stability in longleaf pine with shallow or deep lateral rooting.

LITERATURE CITED

Barnett, J.P. 2002. Longleaf pine: Why plant it? Why use containers? In: Barnett, J.P.; Dumroese, R.K.; Moorhead, D.J. eds. Proceedings of workshops on growing longleaf pine in containers – 1999 and 2001. Gen. Tech. Rep. SRS-56. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 5-7.

Barnett, J.P.; Brissette, J.C. 1986. Producing southern pine seedlings in containers. Gen. Tech. Rep. SO-59. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 71 p.

Boyer, W.D. 1989. Response of planted longleaf pine bare-root and container stock to site preparation and release: fifth-year results. In: Miller, J.H., comp. Proceedings of the 5th biennial southern silvicultural conference. Gen. Tech. Rep. SO-74. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 165-168.

Dumroese, R.K.; Barnett, J.P.; Jackson, D.P. [and others]. 2009. 2008 Interim guidelines for growing longleaf pine seedlings in container nurseries. In: Dumroese, R.K.; Riley, L.E., tech coords. National proceedings: forest and conservation nursery associations –2008. RMRS-P-58. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 101-107.

Dumroese, R.K.; Sung, S.S.; Pinto, J.R. [and others]. 2013. Morphology, gas exchange, and chlorophyll content of longleaf pine seedlings in response to rooting volume, copper root pruning, and nitrogen supply in a container nursery. New Forests. 44: 881-897.

Haywood, J.D.; Sung, S.S.; Sayer, M.A.S. 2012. Copper root pruning and container cavity size influence longleaf pine growth through five growing seasons. Southern Journal of Applied Forestry. 36: 146-151.

Johnson, J.D. 1984. A rapid technique for estimating total surface of pine needles. Forest Science. 30: 913-921.

Kormanik, P.P.; Sung, S.S.; Zarnoch, S.J. 1998. Immature loblolly pine growth and biomass accumulation: Correlations with seedlings initial first-order lateral roots. Southern Journal of Applied Forestry. 22: 117-123.

Krasowski, M.J. 2003. Root system modifications by nursery culture reflect on post-planting growth and development of coniferous seedlings. The Forestry Chronicle. 79: 882-891.

Landers, J.L.; Van Lear, D.H.; Boyer, W.D. 1995. The longleaf pine forests of the southeast: Requiem or renaissance? Journal of Forestry. 93: 39-44.

McNabb, K.; Enebak, S. 2008. Forest tree seedling production in the southern United States: the 2005 – 2006 planting season. Tree Planters' Notes. 53: 47-56.

Ruehle, J.L. 1985. The effect of cupric carbonate on root morphology of containerized mycorrhizal pine seedlings. Canadian Journal of Forest Research. 15: 586-692.

SAS Institute Inc. 2004. SAS/STAT® 9.1 User's Guide. Cary, NC: SAS Institute Inc.

Sayer, M.A.S.; Haywood, J.D.; Sung, S.S. 2009. Cavity size and copper root pruning affect production and establishment of container-grown longleaf pine seedlings. Forest Science. 55: 377-389.

Sayer, M.A.S.; Sung, S.S.; Haywood, J.D. 2011. Longleaf pine root system development and seedling quality in response to copper root pruning and cavity size. Southern Journal of Applied Forestry. 35: 5-11.

South, D.B.; Harris, S.W.; Barnett, J.P. [and others]. 2005. Effect of container type and seedling size on survival and early height growth of *Pinus palustris* seedlings in Alabama, U.S.A. Forest Ecology and Management. 204: 385-398.

South, D.B.; Shelton, J.; Enebak, S.A. 2001. Geotropic lateral roots of container-grown longleaf pine seedlings. Native Plants Journal. 2: 126-130.

Sung, S.S.; Haywood, J.D.; Sword-Sayer, M.A. [and others]. 2010. Effects of container cavity size and copper coating on field performance of container-grown longleaf pine seedlings. In: Stanturf, J.A., ed. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-121. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 241-245.

Sung, S.S.; Haywood, J.D.; Zarnoch, S.J. [and others]. 2009. Long-term container effects on root system architecture of longleaf pine. In: Proceedings of 2009 society of American foresters convention. Bethesda, MD: Society of American Foresters. 9 p.

Sung, S.S.; Leduc, D.L.; Haywood, J.D. [and others]. 2012. Methodology and preliminary results of evaluating stem displacement and assessing root system architecture of longleaf pine saplings. In: Butnor, J.R., ed. Proceedings of the16th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 336-341.

RESTORATION OF LONGLEAF PINE—THE STATUS OF OUR KNOWLEDGE

James M. Guldin, James F. Rosson, Jr., and C. Dana Nelson[1]

Abstract—By the fifth anniversary of the America's Longleaf Restoration Initiative in 2014, the decline in longleaf pine (*Pinus palustris* L.) appears to have been reversed. The area in longleaf pine-dominated stands currently exceeds 4 million acres, up from a low of about 3 million acres two decades ago. A major contribution to this reversal has been the recent establishment of more than 500,000 acres of longleaf pine plantations. However, the plantation approach has limitations in an environment where stand establishment costs exceed $300 per acre, when large-diameter sawlogs currently bring stumpage prices of $30 per ton or less, and when the most urgently-needed habitat is found in mature stands with well-developed understory flora. In addition, expanding the restoration of planted longleaf pine stands will require excluding hybrid Sonderegger pines (*P.* ×*sondereggeri* H.H. Chapm.) from longleaf pine seed orchards and nurseries, and developing silvicultural tactics for sustainable and lucrative pine straw harvests or other local niche markets. A broader and more inclusive approach to restoration might include identifying mixed-pine and pine-hardwood stands with a minor manageable longleaf pine component. Evidence from State surveys and FIA data suggests that such stands could add several million acres of restorable longleaf pine stands across the South. The silvicultural tactics in these stands are simple—harvest the non-longleaf pine component and restore prescribed burning. Not only would this bring an early economic return to the landowner, it would more quickly restore habitat conditions for the flora and fauna of interest in longleaf pine ecosystems. If the approach is feasible, expanding the use of prescribed fire on private lands will become more important, and perhaps more problematic, in restoration of longleaf pine ecosystems.

INTRODUCTION

In July 2014, a celebration was held in Washington DC to honor the fifth anniversary of America's Longleaf Restoration Initiative (ALRI), a program developed to bring Federal resources to bear to recover longleaf pine (*Pinus palustris* L.) across its former range. The ALRI began in 2009 with the publication of the Rangewide Conservation Plan for Longleaf Pine (ALRI, 2009) by an ad hoc Regional Working Group for America's Longleaf. This group is under the direction of three Cabinet-level Federal agencies—the Department of Defense, the Department of Agriculture (both the U.S. Forest Service and the Natural Resources Conservation Service), and the Department of the Interior (specifically, the U.S. Fish and Wildlife Service). The conservation plan laid out a 15-year mission to restore longleaf pine to at least 8 million acres on public and private lands across the South.

The driving force behind the restoration initiative was the catastrophic loss of area in longleaf pine forests over the past four centuries. The magnitude of the loss was calculated by Frost (1993) who concluded that before European settlement there were 74.0 million acres of forests where longleaf pine was dominant, plus another 17.0 million acres where longleaf pine was a component of mixed stands with hardwoods and/or other pine species.

A recent study based on FIA inventory data for the year 2010 puts the area of the longleaf pine-dominated stands at approximately 3.3 million acres. An additional 985,000 acres of the mixed longleaf pine-scrub oak type brings the Southwide estimate to 4.3 million acres (Oswalt and others 2012). This appears to show a reversal of the long-term decline in longleaf pine acreage (fig. 1), not only through better afforestation and reforestation, but also through more widespread use of prescribed burning and invasive species control (ALRI, 2014).

Of the 4.3 million acres in longleaf pine-dominated forest types, two common trends can be detected. First, a quarter of it is young; 758,000 acres is in the 0- to 10-year age class, and another 415,000 acres is in the 11- to 20-year age class (Oswalt and others 2012). Second, the large majority of young stands are planted

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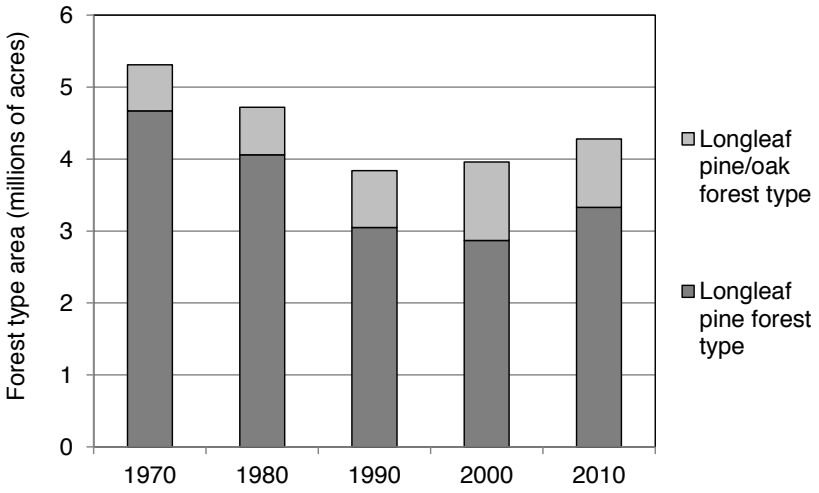


Figure 1—The trends in longleaf pine forest type and longleaf pine/oak forest type from 1970 to 2010. Adapted from USDA Forest Service, Forest Inventory and Analysis, Forest Inventory Online State Reports, August 2013.

relatively recently; of the 1.1 million acres of planted longleaf pine-dominated forests, 84 percent are ≤25 years old (Oswalt and others 2012). This suggests two conclusions—one, that new stand establishment is actively occurring, and two, that planting is generally the tool being used to do it. That has several implications for the short-term and long-term success of restoration.

THE PLANTATION APPROACH TO LONGLEAF PINE RESTORATION

Clearly, if a species is absent from a site, the first step in restoration is to reintroduce the species to the site, and planting is, by far, the most effective tool to meet that goal. However, there are some challenges to overcome with respect to establishment and management of planted longleaf pine stands in the context of restoration, especially on private lands.

Genetics— Seed Sources and Hybrids

Genetic improvement of longleaf pine has lagged behind that achieved for both loblolly pine (*P. taeda*) and slash pine (*P. elliottii* var. *elliottii*). Early genetic studies in longleaf pine showed significant heritable variation in growth and quality traits (Snyder and Namkoong 1978) and limited problems with diseases (Boyer 1990). Seed orchards and associated progeny tests were set-up by the USDA Forest Service's Southern Region and the Western Gulf Forest Tree Improvement Program[2] (Crane and Barbour 2009). The western gulf program subsequently rogued their orchards using progeny test data, however these orchards are owned by states without tree improvement programs. Only one advanced generation breeding program exists, consisting of an experimental population maintained at the Harrison Experimental Forest in south Mississippi (Nelson and others 2005). The established guidelines for seed source movement (Schmidtling 2001) still serve as the best information for seed source selection in consideration of planting programs. Utilizing proper seed sources consisting of multiple seed trees to ensure genetic diversity is a recommended practice (Echt and others 2011) and will need to suffice until more advanced tree improvement programs can deliver improved materials at appropriate scales. With the increased planting called for and being documented through the ALRI, care must be taken in evaluating early plantation success for disease incidence. In particular, brown spot needle blight (caused by Mycosphaerelia dearnessii) and fusiform rust (caused by Cronartium quercuum fsp. fusiforme) have the potential to cause significant losses in large-scale planting programs, and the potential for genetic selection for resistance exists and should be leveraged with ongoing programs. New genomic technologies also allow for reconstructing pedigree information in plantings of known seed sources, and these plantings could be used retrospectively to learn about the genetics of important traits such as disease resistance and emergence from the grass stage (El-Kassaby and Lstiburek 2009).

Beyond the limited availability of optimal seed sources for longleaf pine, there is the potential for increased hybridization with loblolly pine. Anecdotal and scientific evidence suggests that hybrids in pine planting stock may be more common than previously thought. Inter-species hybridization can erode the unique characteristics of the parental species and change their functional attributes in the ecosystem. If the goal is to restore the longleaf pine ecosystem, monitoring and maintenance of species integrity should be considered.

[2]Personal communication, 2015. Dr. Tom Byram, Assistant Professor, Texas A&M Forest Service, College Station, TX.

In studies with shortleaf pine (*P. echinata* Mill.) in the Ouachita Mountains, the prevalence of hybrids between loblolly pine and shortleaf pine especially in native stands appears to be increasingly common. This may be due to increasing overlap between the species in pollen flight and conelet receptivity resulting from changing climatic conditions, coupled with increasing proximity to loblolly pine plantations (Stewart and others 2012, Tauer and others 2012). Shortleaf pine is unique among southern pines in that it sprouts prominently if top-killed (as by fire) when young, an attribute correlated to the unique presence of a basal crook on shortleaf seedlings and saplings (Lilly and others 2012). Among the traits of the hybrids between loblolly pine and shortleaf pine is a much less well developed basal crook, which results in less effective sprouting, implicating fire as a tool to maintain the genetic integrity of shortleaf pine by eliminating the hybrids (Lilly and others 2012, Stewart and others 2014). As a result, The Forest Service's Southern Region (Region 8) is undertaking a major effort to genetically test second-generation shortleaf pines in their seed orchards (fig. 2), and removing those trees that are hybrids between loblolly and shortleaf pine.



Figure 2—Pure shortleaf seedlings (with crook) and putative hybrids (without crook) naturally regenerated from beneath a second-generation shortleaf pine seed orchard on the Kisatchie National Forest. (photo by James M. Guldin)

Anecdotal observations suggest that similar considerations might apply with respect to Sonderegger pine, the hybrid between loblolly and longleaf pines. First described by Chapman (1922), Sonderegger pines emerge from the grass stage sooner and grow more rapidly than longleaf pine (Wells and Wakeley 1970), and they tend to have heavier branches and forks with multiple stems in the upper canopy. Several longleaf pines in the first-generation seed orchard managed by the USDA Forest Service at the Stuart Nursery in Louisiana show what appears to be this attribute typical of Sonderegger pine (fig. 3). As a result, Region 8 and the Southern Research Station are expanding their genetic testing into longleaf pine seed orchards, again with the goal of removing the influence of Sonderegger hybrids on seedlings provided for outplanting in longleaf pine restoration efforts.

Economics and Markets

Longleaf pine plantations are costly to establish. Assuming one is reforesting a cutover site, recommendations to establish a planted stand of longleaf pine might include chemical site preparation and possibly mechanical site preparation. Planting 500-600 trees per acre incurs costs for buying containerized longleaf pine seedlings, and contracting with a planting crew to plant them. All of these costs together can easily exceed $300 per acre for stand establishment (Dooley and Barlow 2013). At a $300 investment per acre and assuming a 6 percent compound interest rate, the compounded costs of stand establishment will grow to more than $1700 per acre after 30 years. The standing volume of an average longleaf pine plantation after 30 years can be expected to be in the ballpark of 60 green tons per acre (Gonzalez-Benecke and others 2012). Current stumpage prices for pine pulpwood are roughly $10 per green ton (Timber-Mart South 2015). This suggests if a landowner was growing longleaf pine for commercial fiber production on a 30-year rotation (itself an unlikely supposition), stumpage prices for pine pulpwood would have to rise considerably to break even on an investment with a 6 percent internal rate of return.

Longer rotations bring additional financial concerns for owners of planted longleaf pine stands. Through the latter part of the 20th century, experts repeatedly advised landowners that the real value for longleaf pine timber products is in large-diameter longleaf pine sawlogs, which commanded premium stumpage prices because of the high quality of lumber they produce. Unfortunately, the start of the 21st century has not been kind to landowners who have pursued this strategy. As an example, pine sawtimber stumpage prices in south Alabama and elsewhere in the South have fallen from nearly $60 per ton in the late 1990s to below $30 per ton in 2012-2015 (Timber-Mart South 2015).



Figure 3—Possible presence of Sonderegger pine (hybrid between loblolly and longleaf pine) in a first-generation seed orchard on the Kisatchie National Forest. (photo by James M. Guldin)

In addition, logging technology is changing, with cutting machines and forwarders replacing loggers with chain saws and skidders for felling and bucking pine sawlogs. Changes in mill technology are also leading to the processing of smaller sawlogs. As a result, it is increasingly difficult to sell sawlogs larger than 24 inches in diameter at the large end to timber buyers.

Finally, long rotations are very difficult to justify economically when stand establishment costs are high and early returns from thinning are limited. Yields that might be expected in a typical 80-year even-aged rotation in a typical planted longleaf pine stand could include thinning at age 30 (12 tons per acre), age 45 (19 tons per acre), and age 60 (24 tons per acre), and a final standing volume of 90 tons per acre (two-thirds of which is sawtimber) at age 80 (Gonzalez-Benecke and others 2012). Carrying a high stand establishment cost over an 80-year rotation is not an economically robust investment.

A key alternative for landowners managing longleaf pine stands over longer rotations is to take advantage of unique markets. One example is to sell high-quality sawlogs in the specialized market for utility poles, which brings a premium stumpage price; another is to consider harvest of pine straw, which can be quite profitable (Roise and others, 1991). Local market development for high-quality dimension lumber is urgently needed. However, without access to niche markets, the only way to generate a positive return over a sawtimber rotation is for a landowner to be comfortable with a low internal rate of return of about 4 percent or less.

Habitat in Planted Stands

The main reason for concern about the loss of longleaf pine ecosystems across the South is the loss of habitat for fauna and flora associated with mature open forests and woodlands. Species such as gopher tortoise (*Gopherus polyphemus*), red-cockaded woodpecker (*Picoides borealis*), Bachman's sparrow (*Peucaea aestivalis*), northern bobwhite (*Colinus virginianus*), bluestem grasses (*Andropogon* spp.) and wiregrass (*Aristida stricta* Michx.) thrive in open woodlands maintained by frequent fire.

This is not the kind of habitat that is associated with young pine plantations, including longleaf pine plantations. Planted stands quickly develop into an early stem exclusion stage (cf. Oliver and Larsen 1996), when crown closure in plantations reduces understory development to the point where native forbs and grasses are suppressed. In addition to the constraint on development of legumes and grasses for which longleaf pine ecosystems are known, the lack of ground-layer biomass reduces foraging habitat for gopher tortoise as well as nesting and foraging habitat for northern bobwhites. Depending on subsequent silvicultural activities, the period during which understory vegetation is suppressed may last for two to three decades.

There is also a considerable difference in the ease with which new understory plant communities can become established in planted stands of longleaf pine, depending on whether the site has a history of agricultural use. Converting land from forest to agricultural use not only removes the woody vegetation, but plowing and planting crops also degrades the understory woodland flora that had existed prior to conversion (Walker and Silletti 2006). Restoration of forest land that has never been in agricultural production is relatively easy; in addition to replanting trees, reintroduction of prescribed burning associated with some precommercial or commercial thinning promotes natural development of the existing seed bank and rootstocks of understory forbs, legumes, and grasses, even in poletimber stands. However, restoration of longleaf pine ecosystems planted on abandoned agricultural land requires not only planting longleaf pine, but also direct seeding or planting of the complex of native warm season grasses and legumes (Kaeser and Kirkman 2014, Norden and Kirkman 2014). The cost of restoration of understory plants adds to the financial commitment a landowner may make.

In summary, the approach to restoration of longleaf pine through establishment of new plantations will continue to be important, but remains challenging both economically and ecologically. There are issues to ponder genetically as well, though those seem manageable. This begs the question about whether a second approach, in addition to the plantation model might be considered.

MANAGEMENT OF EXISTING MIXED PINE STANDS

Several of the issues with the plantation approach to restoring longleaf pine stands might be addressed by a different approach—actively managing the composition of immature and mature mixed-species pine and pine-hardwood stands that are not dominated by longleaf pine, but that contain a minor and manageable component of longleaf pine. This approach involves first identifying these stands, and then using commercial thinning to remove most, if not all, of the trees other than longleaf pine, to convert the stand to a longleaf pine-dominated stand. That would be followed by initiation of cyclic prescribed burning as the subsequent step to restore the understory plant community.

Quantifying a Minor Manageable Component of Longleaf Pine

Managing stands that are essentially understocked with longleaf pine has some support in silviculture research in southern pines from the latter part of the

20th century. The classic work of Croker and Boyer (1975) on the Escambia Experimental Forest in south Alabama clearly established the feasibility of using the shelterwood method to obtain natural regeneration of longleaf pine. The recommendation of leaving 30 square feet per acre after the seed cut was based on work by Boyer (1979) who noted that stand-level seed production declined at residual basal areas greater than 40 square feet per acre and less than 30 square feet per acre. Based on this, one suggestion for a lower limit of longleaf pine basal area in a mixed stand would be 30 square feet per acre. The prescription would retain all the longleaf pines, harvest most or all of the other trees, conduct any supplemental midstory removal that might be necessary through harvest or as a mechanical or chemical intermediate treatment, and initiate cyclic prescribed burning. The goal would be to either maintain an understocked pine woodland condition dominated by longleaf pine, or develop a second age cohort of naturally-regenerated longleaf pine seedlings beneath the residual longleaf pine overstory in a manner approximating the shelterwood-with-reserves method.

A second approach to quantifying the lower threshold for a minor manageable longleaf pine component might be through research on rehabilitation of understocked pine stands. Resource analysts with the Forest Inventory and Analysis Program of the U.S. Forest Service consider a stand with 60 percent stocking as the lower limit of full stocking (USDA Forest Service 1972). However, stand development in the Farm Forestry Forty demonstrations at the Crossett Experimental Forest in south Arkansas (in mixed loblolly-shortleaf pine stands) showed a relatively rapid recovery from cutover understocked conditions in the first two decades of management (Reynolds 1969, Reynolds and others 1984). Baker and Shelton (1998) expanded upon those early observations with a study of stand growth and yield in response to five levels of understocking. They concluded that loblolly-shortleaf pine stands on medium sites that were cut back to as little as 30 percent stocking were able to recover to fully stocked conditions within 15 years. The fact that stands could recover from slightly understocked conditions was not surprising, but stands with 30 percent stocking do not give a visual impression that they would be able to recover that rapidly. In the absence of better data specifically for longleaf pine stands, the 30-percent stocking figure might be a second point of departure to quantify the lower limit of a minor but manageable longleaf pine component in mixed stands.

Estimates of Area in Mixed Longleaf Pine Stands

Among the reasons this approach to restoration of manageable mixed pine stands might be fruitful is the potential increase in area of longleaf pine that could be in restorable condition. The report on longleaf pine area published by Oswalt and others (2012) is based on an analysis of FIA data where longleaf pine accounts for a plurality of stocking on the plot. This resulted in two predominant FIA forest types: longleaf pine (where longleaf pine clearly made up a plurality of stand stocking), and longleaf pine-oak (where scrub oaks made up a plurality of stocking and longleaf pine accounted for 25- to 50 percent of stand stocking.) This is the area where longleaf pine was never lost, or where it is being recovered using planted stands.

A case study within the natural range of longleaf pine might illustrate the potential gains in recovery of longleaf pine using this approach. As previously stated, the area of longleaf pine-dominated forests, according to FIA data, is roughly 4.3 million acres (Oswalt and others 2012). But FIA data can also provide a rough estimate of the area that might be available for restoration in stands with a minor component of longleaf pine that could possibly attain a plurality of longleaf pine stocking with the removal of more prominent species in the stand.

One analytical approach includes all FIA plots with at least one longleaf pine ≥5.0 inches diameter at breast height (d.b.h.) and in which longleaf pine may or may not be the dominant species. This results in an estimate of 7.9 million acres of stands within the natural range of longleaf pine in which some portion of stand basal area includes longleaf pine (fig. 4). This analysis probably excludes most of the 758,000 acres of planted longleaf pine stands ≤ 10 years old, because they would not contain trees ≥ 5.0 inches d.b.h. Assuming that stands with longleaf pine basal area from 20- to 50 percent of stand basal area might qualify as having a minor manageable longleaf pine component (fig. 4), the South may have 1.825 million acres of stands where longleaf pine is a minor manageable component, and in which restoration might be used to create longleaf pine-dominated stands.

A second analysis includes all FIA plots that have two or more species that are ≥5.0 inches d.b.h. and where longleaf is not the dominant species (fig. 5). Stands in this analysis total 4.11 million acres across the natural range of longleaf pine. Again, assuming that stands with longleaf pine basal area from 20- to 50 percent of stand basal area might qualify as having a minor manageable longleaf pine component (fig. 5), the South may have an additional 1.24 million acres of stands in which longleaf pine is not the dominant species but where restoration to create longleaf pine-dominated stands might be considered.

DISCUSSION

Reliance on restoration of longleaf pine ecosystems using planted stands on abandoned agricultural lands or in reforestation of cutover stands that did not support longleaf pine will continue to be a fundamental objective under the Rangewide Conservation Plan for Longleaf

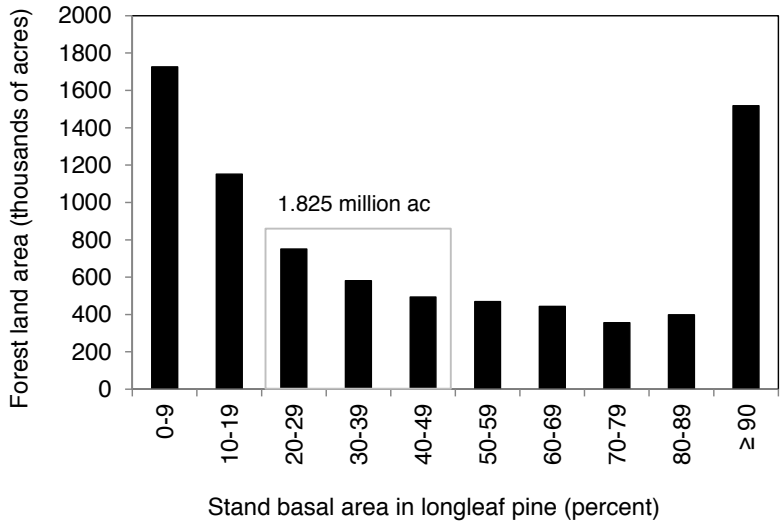


Figure 4—Forest land area by stand basal area classes where longleaf pine may or may not be the dominant species. Data include all FIA plots that have at least one longleaf pine ≥ 5.0 inches in diameter at breast height.

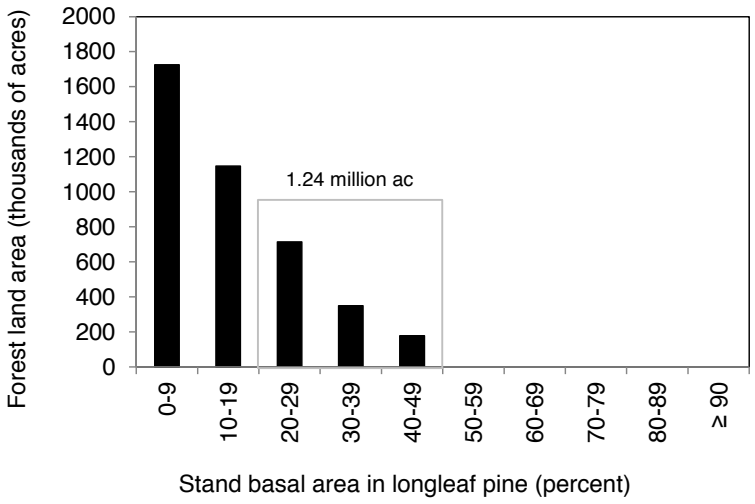


Figure 5—Forest land area by stand basal area classes where longleaf pine is not the dominant species. Data include all FIA plots that have at least one longleaf pine ≥5.0 inches in diameter at breast height.

Pine. The best way to re-establish longleaf pine on sites adapted to longleaf pine without a seed source is to use planting. The nursery practices to raise containerized longleaf pine seedlings and the details of the silviculture required to establish new planted stands are well documented. However, the practice is expensive, and development of appropriate habitat for species at risk from loss of native pine ecosystems may take two decades to establish, if not longer.

Silvicultural practices developed in southern pines in the latter part of the 20th century offer an alternative approach to augment existing plantation restoration efforts. Looking at those studies from a 21st-century perspective suggests that understocked pine stands can often be effectively managed and used to recover full stocking over time, not just for a timber resource but also for habitat associated with mature pine forest ecosystems. We suspect that if these stands have a manageable longleaf pine component in them and have never been in agricultural use, restoration of functional habitat appropriate for the species of conservation concern in longleaf pine stands can occur more quickly, and potentially at a much lower cost to the landowner, than it can through the creation of new plantations.

If removing all other pine species in mixture with longleaf pine creates stand conditions that a landowner thinks are too understocked, it is probably feasible to retain a mixed stand as long as the percentage of longleaf is increased to some degree. This could be important if a broader suite of habitat values can be

maintained through retention of a mixed-dominance pine component (Lavoie and others 2011) and again, if the stands do not have an agricultural history and the native understory flora can be brought back through prescribed burning.

The surprising element of this approach is the area that could potentially be restored. If converting stands with a minor manageable longleaf pine component into longleaf pine-dominated stands proves fruitful, the area that could potentially be restored in a relatively simple manner would increase by a range of from 1.24 million to more than 3 million acres. That would make dramatic progress toward achieving the 8 million acres targeted for achievement by 2025 in the Rangewide Longleaf Conservation Plan.

It would be relatively easy to make a start at this on Federal and State lands simply by adopting some modifications in existing silvicultural prescriptions for stands with a minor manageable longleaf pine component to increase the percentage of longleaf pine through thinning and re-establishment of cyclic prescribed burning. The challenge on public and private lands would be to expand the use of prescribed burning, which is increasingly constrained by air quality regulations especially in heavily populated areas. A doubling of area in longleaf pine stands being restored would double the need for annual prescribed burning to create and maintain open forest and woodland habitat. This may be difficult to accomplish in the current social and political climate.

LITERATURE CITED

America's Longleaf Restoration Initiative (ALRI). 2009. Rangewide conservation plan for longleaf pine. Atlanta, GA: America's Longleaf Restoration Initiative. 42 p. http://www.americaslongleaf.org/media/86/conservation_plan.pdf. [Date accessed: January 29, 2015]

America's Longleaf Restoration Initiative (ALRI). 2014. 2013 Range-wide accomplishment report. Atlanta, GA: America's Longleaf Restoration Initiative, Longleaf Partnership Council. 19 p. [Date accessed: January 29, 2015]

Baker, J.B.; Shelton, M.G. 1998. Rehabilitation of understocked loblolly-shortleaf pine stands—I. recently cutover natural stands. Southern Journal of Applied Forestry. 22(1): 35-40.

Boyer, W.D. 1979. Natural regeneration of longleaf pine. Journal of Forestry. 77(9): 572-575.

Boyer, W.D. 1990. Pinus palustris Mill., Longleaf pine. In: Burns, R.M., Honkala, B.H., tech. cords. 1990. Silvics of North America. Vol. 1. Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 405-412.

Chapman, H.H. 1922. A new hybrid pine (*Pinus palustris* x *Pinus taeda*). Journal of Forestry. 20: 729-735.

Crane, B. and Barbour, J. 2009. Status of longleaf pine seed orchard resources in support of restoration. In: Proceedings of the 30th southern forest tree improvement conference. Blacksburg, VA: 74-76. http://www.rngr.net/publications/tree-improvement-proceedings/sftic/2009/status-of-longleaf-pine-seed-orchard-resources-to-support-restoration.pdf [Date accessed: July 7, 2015]

Croker, T.C., Jr.; Boyer, W.D. 1975. Regenerating longleaf pine naturally. Res. Pap. SO-105. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 21 p.

Dooley, E.; Barlow, R. 2013. Special report: 2012 costs and cost trends for forestry practices in the South. Forest Landowner. July/August, 22-28. http://digital.graphcompubs.com/publication/?i=171803&p=24. [Date accessed: February 3, 2015].

El-Kassaby, Y.A.; Lstiburek, M. 2009. Breeding without breeding. Genetics Research. 91(2): 111-120.

Frost, C.C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. In: Hermann, S.M., ed. The longleaf pine ecosystem: ecology, restoration and management. In: Proceedings of the Tall Timber fire ecology conference No. 18. Tallahassee, FL: Tall Timbers Research Station: 17-43.

Gonzalez-Benecke, C.A.; Gezan, S.A.; Leduc, D.J. [and others]. 2012. Modeling survival, yield, volume partitioning and their response to thinning for longleaf pine (*Pinus palustris* Mill.) plantations. Forests. 3: 1104-1132.

Kaeser, M.J.; Kirkman, L.K. 2014. Field and restoration guide to common native warm-season grasses of the longleaf pine ecosystem. Unnumbered publication. Newton, GA: Joseph W. Jones Ecological Research Center at Ichauway, in partnership with the Georgia Native Plant Society. 71 p.

Lavoie, M.; Kobziar, L.; Long, A.; Hainds, M. 2011. Problems and needs for restorationists of longleaf pine ecosystems: a survey. Natural Areas Journal. 31(3): 294-299.

Lilly, C.J.; Will, R.E.; Tauer, C.G. [and others]. 2012. Factors affecting the sprouting of shortleaf pine rootstock following prescribed fire. Forest Ecology and Management. 265: 13-19.

Nelson, C.D.; Lott, L.H.; Gwaze, D.P. 2005. Expected genetic gains and development plans for two longleaf pine third-generation seedling seed orchards. In: Proceedings of the 28th southern forest tree improvement conference, June 20-23, 2005, Raleigh, NC: North Carolina State University, Department of Forestry and Environmental Resources: 108-114.

Norden, H.; Kirkman, K. 2014. Field guide to common legume species of the longleaf pine ecosystem. Unnumbered publication. Newton, GA: Joseph W. Jones Ecological Research Center at Ichauway, in partnership with the Georgia Native Plant Society. 72 p.

Oliver, C.D.; Larsen, B.C. 1996. Forest stand dynamics. New York: John Wiley and Sons. 544 p.

Oswalt, C.M.; Cooper, J.A.; Brockway, D.G. [and others]. 2012. History and current condition of longleaf pine in the Southern United States. Gen. Tech. Rep. SRS-166. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 51 p.

Reynolds, R.R.; Baker, J.B.; Ku, T.T. 1984. Four decades of selection management on the Crossett Farm Forestry Forties. Bulletin 872. Fayetteville, AR: University of Arkansas, Arkansas Agricultural Experiment Station. 43 p.

Reynolds, R.R. 1969. Twenty-nine years of selection timber management on the Crossett Experimental Forest. Res. Pap. SO-40. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 19 p.

Roise, J.P.; Chung, J.; Lancia, R. 1991. Red-cockaded woodpecker habitat management and longleaf pine straw production: an economic analysis. Southern Journal of Applied Forestry. 15(2): 88-92.

Schmidtling, R.C. 2001. Southern pine seed sources. Gen. Tech. Rep. SRS-44. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 25 p.

Snyder, E. B.; Namkoong, G. 1978. Inheritance in a diallel crossing experiment with longleaf pine. Res. Pap. SO-140. New Orleans, LA: U.S. Department of Agriculture Forest Service Southern Forest Experiment Station. 40 p.

Stewart, J.F.; Will, R.E.; Robertson, K.M.; Nelson, C.D. 2014. Frequent fire protects shortleaf pine (*Pinus echinata*) from introgression by loblolly pine (*P. taeda*). Conservation Genetics. DOI 10.1007/s10592-014-0669-x. [Date accessed: February 3, 2015].

Stewart, J.F.; Tauer, C.G.; Nelson, C. D. 2012. Bidirectional introgression between loblolly pine (*Pinus taeda* L.) and shortleaf pine (*P. echinata* Mill.) has increased since the 1950s. Tree Genetics and Genomics. 8: 725-735.

Tauer, C.G.; Stewart, J.F.; Will, R.E. [and others]. 2012. Hybridization leads to loss of genetic integrity in shortleaf pine: unexpected consequences of pine management and fire suppression. Journal of Forestry. 110(4): 216-224.

Timber-Mart South, 2015. Southwide average prices. http://www.timbermart-south.com/prices.html. [Date accessed: May 6, 2015].

USDA Forest Service. 1972. Forest survey handbook, FSH 4813.1. Washington, DC: U.S. Department of Agriculture. 254 p.

Walker, J.L.; Silletti, A.M. 2006. Restoring the ground layer of longleaf pine ecosystems. In: Jose, S.; Jokela, E.J.; Miller, D.L., eds. 2006. Ch. 10. The longleaf pine ecosystem--ecology, silviculture, and restoration. New York: Springer Science+Business Media, LLC: 297-325.

Wells, O.O.; Wakeley, P.C. 1970. Variation in longleaf pine from several geographic sources. Forest Science. 16(1): 28-42.

Longleaf 2

Moderator:

Ben Knapp
University of Missouri

RECENT ADVANCES IN UNDERSTANDING DUFF CONSUMPTION AND POST-FIRE LONGLEAF PINE MORTALITY

J. Morgan Varner, Jesse K. Kreye, J. Kevin Hiers, and Joseph J. O'Brien[1]

Abstract—Many longleaf pine stands across the range have suffered decades of fire exclusion, leading to declines in plant and animal biodiversity and complicating restoration and management efforts. Recent research on this topic has focused on the physiological response of overstory longleaf pines and the fuel characteristics of the surrounding forest floor. In small-scale and operational prescribed fires, post-fire pine mortality is tightly linked to basal duff (lowermost fermentation and humus forest floor horizons) consumption and, where present, crown scorch. Pines with substantial duff consumption suffered coarse root carbohydrate drain, a decline in sap flux, and reduced leaf chlorophyll content. Duff consumption in prescribed fires has been linked to duff moisture content, a difficult to predict variable in prescribed fire planning. Duff moisture varies tremendously across the forest floor and within typical burn units. Basal duff dries more rapidly than within-stand conditions. This heterogeneity in duff characteristics is further complicated by the presence of ignition vectors in the forest floor, including woody fuels and intact pine cones. When pine cones are present, ignition of underlying duff is facilitated well beyond assumed moisture thresholds of these fuels. Operational prescribed burns in long-unburned sites should focus efforts on balancing duff consumption with the need to retain overstory longleaf pines.

INTRODUCTION

Over the past two decades, forest and fire managers across the southeastern United States have reported cases of unexpected heavy overstory tree mortality following prescribed and wildfires. Mortality has been observed in several southern pines (Varner and others 2005), most notably in the normally fire-resistant longleaf pine (*Pinus palustris*). Reports of longleaf pine mortality have exceeded 75 percent, with the greatest losses occurring in larger pines (Varner and others 2005). The cause of these mortality events have been linked to long-duration heating of accumulated forest floor fermentation and humus horizons, so-called "duff" (Varner and others 2005, 2007, O'Brien and others 2010). To date, no synthesis has occurred on the cause of these duff fires and their consequences for pine survival.

Fire has been an important factor for millennia in southern pine forests in general, and longleaf pine forests and woodlands in particular. With European settlement and subsequent land use changes over the 20th century, fire diminished in its use in remnant longleaf pine forests (Jose and others 2006). Without fire, senesced leaves, bark slough, cones, and woody branches accumulate on the forest floor. This accumulated debris forms distinct forest floor horizons: surface litter (L or Oi horizon); fermentation (F or Oe horizon); and humus (H or Oa horizon). The lower, heavily decomposed fermentation and humus horizons are collectively termed "duff." The presence of duff in longleaf pine stands suggests long periods without fire (Varner and others 2005).

DUFF AND FIRE

Duff contrasts markedly with litter fuels. Whereas litter burns with rapidly spreading, intense surface fire (Fonda 2001), duff burns via smoldering combustion. Smoldering is a solid-phase or glowing form of combustion where char is generated by the thermal degradation of the fuel (Miyanishi 2001). This char mantle then ignites and glows as the smoldering front progresses. Smoldering is characterized by lower intensity, but protracted duration burning. Duff fires are capable of smoldering for hours to days following the passage of the flaming front, thereby heating the surface mineral soil, imbedded roots, and potentially tree basal cambium (Varner and others 2009).

Although widely assumed to be homogenous (the literature is replete with text describing "deep litter"), the composition and structure of duff is diverse. Duff tends to "mound" around source trees; larger trees reveal this pattern (fig. 1). These basal mounds are markedly

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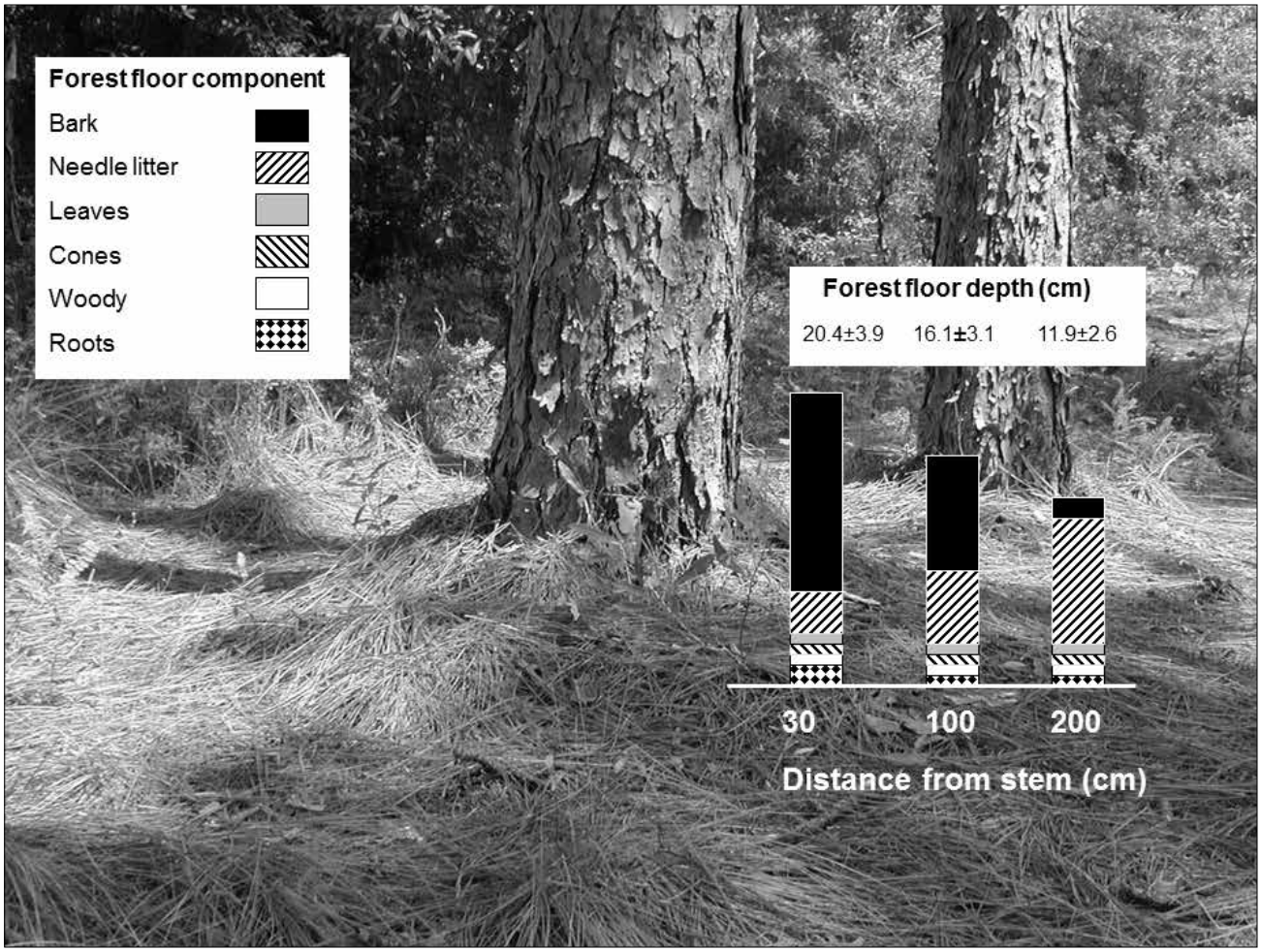


Figure 1—Forest floor accumulations in a long-unburned longleaf pine stand in northern Florida. Forest floor duff is concentrated at tree bases. The composition of duff differs with distance from stem, consisting primarily of bark slough near the bole and becoming increasingly dominated by needle litter with distance from source trees.

deeper than elsewhere in the stand, with depths near tree bases exceeding 20 to 30 cm in long-unburned stands (Varner and others 2005, Kreye and others 2014). The composition of basal mounds is dominated by sloughed bark, with senesced needle litter, cones, woody fuels, and imbedded roots comprising the remainder (fig. 1). As distance from the source tree increases, needle litter dominates the composition.

While pine and oak litter burning characteristics are somewhat understood (Fonda 2001, Kane and others 2008), there is considerably less known about duff burning. The ignition and spread rates of smoldering duff fires are controlled by moisture content, bulk density, mineral content, and depth. Ignition is limited primarily by fuel moisture (Miyanishi 2001), which is spatially heterogeneous in longleaf pine forest floor fuels (Kreye and others 2014). Duff moisture contents above 60 percent (dry-weight basis) resist ignition unless vectors are present. Ignition vectors include woody fuels and pine cones (Fonda and Varner 2004, Kreye and others 2013) that are often elevated and thus drier that the surrounding duff (Varner and others 2009). Cones, in particular, are capable of burning for long periods (mean burning time for longleaf pine cones = 49 minutes; Fonda and Varner 2004) thereby drying and pre-heating adjacent duff. Kreye and others (2014) found that the presence of cones enabled duff to ignite and smolder even when above presumed ignition thresholds.

CONSEQUENCES OF DUFF FIRES

Restoration fires across longleaf pine's range have resulted in substantial tree stress and mortality (Varner and others 2005). The cause of tree stress and mortality has been linked to duff consumption near the base of trees (Varner and others 2007, 2009, O'Brien and others 2010). In southeastern Georgia longleaf pine stands, duff consumption diminished sap flux and reduced leaf chlorophyll (O'Brien and others 2010). The duration of duff and underlying mineral soil heating in north Florida was linked to reductions in coarse root carbohydrate storage, presumably due to the drain to replenish killed

or injured fine roots (Varner and others 2009). Latewood radial growth the year following duff fires was likewise related to durations of heating (Varner and others 2009).

The proximate mechanism of these injuries is unclear. Because duff burns so slowly and can form a mantle over the underlying heated mineral soil, heat is retained for long durations following ignition (Varner and others 2009). The high density of fine roots (O'Brien and others 2010) in basal duff and in the uppermost mineral soil places these tissues at risk of fire-caused injury or consumption. This hypothesis is corroborated by results for diminished sap flux and drained carbohydrates in coarse storage roots. The limited data we have on basal heating suggests that the thick bark of longleaf pine insulates underlying cambium. Other possible contributing mechanisms include post-fire climatic water deficits, interactions with bark beetles, and root diseases (Ostrosina and others 1999, Menges and Deyrup 2001, Varner and others 2009). This blurred linkage between cause and effect, combined with the uncertainties of future climate change in the region (Mitchell and others 2014), is a hindrance to understanding and managing these ecosystems.

There are widespread examples of duff fires resulting in pine mortality. Varner and others (2005) reported cases of duff fire-caused mortality across the range of longleaf pine and across a suite of ownerships and manager experience. Using replicated prescribed fires in northwest Florida, Varner and others (2007) found that stands burned with lower duff moisture suffered markedly greater tree mortality than those burned under moist or wet duff moisture contents. In sites with heavy mortality, tree death was concentrated in the largest size classes, contrary to typical fire-caused tree mortality predictive models (Varner and others 2007). The same result of large tree mortality has been observed in other reintroduction fires in south Alabama (Kush and others 2004).

BALANCING DUFF CONSUMPTION AND OVERSTORY MORTALITY

A primary challenge for managers faced with long-unburned longleaf pine stands is to balance duff consumption with overstory pine mortality (Varner and others 2007). Burning when duff is too wet to ignite results in minimal tree mortality but a residual duff hazard. Burning when the duff is too dry results in maximal duff consumption but heavy overstory pine mortality. The balancing act between these extremes is simple in concept, but difficult in practice. One reason for the difficulty is the spatial heterogeneity of duff moisture (Kreye and others 2014) and the difficulty of measuring duff moisture (Engber and others 2013).

In spite of the difficulty, there are cases of small- and large-scale restoration treatments that achieve the balance between duff consumption and overstory pine mortality. In a series of prescribed fires in south Alabama, Kush and others (2004) burned following recent soaking rains and then extinguished (via backpack sprayers and ATV-mounted sprayers) individual pines for two days following fires. This careful burning and mop-up was successful in reducing forest floor depths (due to repeated fires) and resulted in minimal overstory pine mortality. At the large-scale, Eglin Air Force Base used operational prescribed fires in northwestern Florida to reduce duff. Using a similar prescription and data from nearby meteorological stations (Ferguson and others 2002), managers were able to consume duff, while having minimal overstory pine mortality. In a replicated prescribed fire experiment with three duff moisture treatments (dry, moist, and wet), Varner and others (2007) found that burning when duff was moist (85 percent average duff moisture content) reduced forest floor depths while resulting in minimal (less than 5 percent) overstory pine mortality. At dry duff moistures below this range, mortality was high and peaked in large pines; at wet duff moistures above this range, little duff was consumed (Varner and others 2007). There are other examples of operational prescribed fires with minimal overstory pine mortality in southern Georgia and northern Florida (Hiers and others 2005).

Finding site-specific prescriptions that result in balances between duff consumption and pine mortality is clearly possible, but larger issues impede success. The impediments to using prescribed fire in southeastern pine ecosystems are well-established (Hiers and others 2003, Kobziar and others 2015). Long-unburned stands are problematic due to the presence of deep duff, but also suffer diminished plant and animal diversity (Jose and others 2006). Prioritizing burning for sites with deep duff is possible without substantially compromising other high priority sites for burning (Hiers and others 2003).

CONCLUSIONS

The ecological consequences of fire exclusion and subsequent reintroduction of fire is a primary issue in longleaf pine ecosystem management. Current understanding of this issue suggests that the forest floor is a much more complex fuel stratum than earlier appreciated. The consequences of fires cause a cascade of physiological injuries that can result in heavy tree mortality, particularly in large pines. There are examples of successful efforts balancing duff consumption with overstory tree mortality, but challenges remain.

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LITERATURE CITED

Engber, E.A.; Varner, J.M.; Dugaw, C. [and others]. 2013. Utility of an instantaneous moisture meter for duff moisture prediction in long-unburned longleaf pine forests. Southern Journal of Applied Forestry. 37: 13-17.

Ferguson, S.A.; Ruthford, J.E.; McKay, S.J. [and others]. 2002. Measuring moisture dynamics to predict fire severity in longleaf pine forests. International Journal of Wildland Fire. 11: 267-279.

Fonda, R. 2001. Burning characteristics of needles from eight pine species. Forest Science. 72: 1-9.

Fonda, R.; Varner, J.M. 2004. Burning characteristics of cones from eight pine species. Northwest Science. 78: 322-333.

Hiers, J.K.; Laine, S.C.; Bachant, J.J. [and others]. 2003. Simple spatial modeling tool for prioritizing prescribed burning activities at the landscape scale. Conservation Biology. 17: 571-1578.

Hiers, J.K.; Gordon, D.R.; Mitchell, R.J.; O'Brien, J.J. 2005. Duff consumption and southern pine mortality. Final Report to Joint Fire Science Program. Project JFSP-01-1-3-11. 111 p.

Jose, S.; Jokela, E.J.; Miller, D.L. 2006. The longleaf pine ecosystem. Ecology, silviculture, and restoration. New York: Springer. 438 p.

Kane, J.M.; Varner, J.M.; Hiers, J.K. 2008. The burning characteristics in southeastern oaks: discriminating fire facilitators from fire impeders. Forest Ecology and Management. 256: 2039-2045.

Kobziar, L.N.; Godwin, D.; Taylor, L.; Watts, A.C. 2015. Perspectives on trends, effectiveness, and impediments to prescribed burning in the southern US. Forests. 6: 561-580.

Kreye, J.K.; Varner, J.M.; Dugaw, C.J. [and others]. 2013. Pine cones facilitate ignition of forest floor duff. Canadian Journal of Forest Research. 43: 512-516.

Kreye, J.K.; Varner, J.M.; Dugaw, C.J. 2014. Spatial and temporal variability of forest floor duff characteristics in long-unburned Pinus palustris forests. Canadian Journal of Forest Research. 44: 1477-1486.

Kush, J.S.; Meldahl, R.S.; Avery, C. 2004. A restoration success: longleaf pine seedlings established in a fire-suppressed, old-growth stand. Ecological Restoration. 22: 6-10.

Menges, E.S.; Deyrup, M.A. 2001. Postfire survival in south Florida slash pine: interacting effects of fire intensity, fire season, vegetation, burn size, and bark beetles. International Journal of Wildland Fire. 10: 53-63.

Mitchell, R.J.; Liu, Y.; O'Brien, J.J. [and others]. 2014. Future climate and fire interactions in the southeastern region of the United States. Forest Ecology and Management. 327: 316-326.

Miyanishi, K. 2001. Duff consumption. In: Johnson, E.A.; Miyanishi, K., eds. Forest fires. Behavior and ecological effects. San Diego, CA: Academic Press: 437-476.

O'Brien, J.J.; Hiers, J.K.; Mitchell, R.J.; [and others]. 2010. Acute physiological stress and mortality following fire in a long-unburned longleaf pine ecosystem. Fire Ecology. 6: 1-12.

Ostrosina, W.J.; Bannwart, D.; Roncadori, R.W. 1999. Root-infecting fungi associated with a decline of longleaf pine in the southeastern United States. Plant Soil. 217: 145-150.

Varner, J.M.; Gordon, D.R.; Putz, F.E.; Hiers, J.K. 2005. Restoring fire to long-unburned *Pinus palustris* ecosystems: novel fire effects and consequences for long-unburned ecosystems. Restoration Ecology. 13: 536-544.

Varner, J.M.; Hiers, J.K.; Ottmar, R.D.; Gordon, D.R.; Putz, F.E.; Wade, D.D. 2007. Tree mortality resulting from re-introducing fire to long-unburned longleaf pine ecosystems: the importance of duff moisture. Canadian Journal of Forest Research. 37: 1349-1358.

Varner, J.M.; Putz, F.E.; O'Brien, J.J. [and others]. 2009. Post-fire tree stress and growth following smoldering duff fires. Forest Ecology and Management. 258: 2467-2474.

LONGLEAF PINE: A LONG-ROTATION TREE IN A VERY SHORT-SIGHTED WORLD

John S. Kush[1]

Abstract—Longleaf pine is a long-rotation tree, with potential intermediate products from pine straw, frequent thinnings, hunting leases, and wildlife habitat. Despite a focus on plantations and short-rotation management, many landowners and managers are still interested in long-term management because of high-valued products derived from longer rotations. Do we have the most basic and reliable information to write management plans for long-term rotations like a 45-, 65-, or 120-year rotations? Can we improve planning for longleaf silvicultural activities of regeneration, thinning, and burning? We would argue the data exists to help answer these questions in the U.S. Forest Service's Regional Longleaf Growth Study (RLGS) established in 1964. The study's original objective was to obtain a database to develop growth and yield predictions for naturally regenerated, even-aged longleaf pine stands. We have expanded the RLGS to examine pine straw, utility pole, forage production, and more. Recent results include a site index equation and stand level growth models. The RLGS has 40+ years implementation of basal area management regimes and replication in time component that can address adaptive management and climate change issues. The future of the RLGS is doubtful and "how can this be?" is a good question. This irreplaceable investment of decades in documented management cannot be ignored and should be more relevant with longleaf pine restoration efforts. This presentation will show how the nearly 50-year old RLGS has addressed numerous questions related to longleaf pine management and the importance for its continuance.

INTRODUCTION

Robert M. Farrar, Jr. (honored posthumously)

Before I begin the discussion of my topic, the Regional Longleaf Pine Growth Study (RLGS) and what it has added to the longleaf pine knowledge base, I need to honor the memory of Dr. Robert (Bob) M. Farrar, Jr. Bob passed away in early 2014 in Starkville, Mississippi. Bob worked for the U.S. Forest Service (USFS) for 30 years, much of his time researching longleaf pine. Bob always presented at the Biennial Southern Silvicultural Research Conference (BSSRC), while with the USFS. He would show up with Mylar sheets in hand and put them on an overhead projector. These sheets were filled with numbers and equations that few of us could keep up with. Long before there was the effort to restore longleaf, Bob said this about the longleaf situation: "Everyone is "looking" but few are "seeing" what is happening to longleaf". Bob was right about that and much more.

REGIONAL LONGLEAF GROWTH STUDY

In 1964, the USFS (Bob's PhD) established the RLGS in the Gulf States (Farrar 1978). The RLGS had a rare longitudinal approach in that there were research plots from the Panhandle of Florida to western Mississippi, and north into the mountains of Alabama. The original objective of the study was to obtain a database for the development of growth and yield predictions for naturally regenerated, even-aged longleaf pine stands.

Plots were installed to cover a range of ages, densities, and site qualities. The study was UNIQUE in that it accounted for change in growth with the addition of new plots in the youngest age class every 10 years on the Escambia Experimental Forest (EEF) located just south of Brewton, Alabama. These series of plots were referred to as "Time-rep plots". The 6th set off of these were scheduled for installation in 2014.

Plot selection within the RLGS was based upon a rectangular distribution of cells formed by: 6 age classes from 20 to 120 years; 4 site quality classes from 50 to 80 feet (base age 50); 6 density classes ranging from 30 to 150 square feet/acre; and plots left un-thinned to grow. If the plot basal area had grown 7.5 square feet/acre beyond the target basal area, plots were thinned back to the previously assigned target. Thinning generally low intensity and done from below. Plots should have been prescribed burned at least every 3 years with cool, dormant season fires. The initial installation in the mid-1960's resulted in 185 sample plots. By 1981, the number of plots was down to 166

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due to hurricanes, tornadoes and trespass cutting. In 1984, Auburn University (SOFWS), in a cooperative agreement with the USFS, re-measured the plots for its 4th measurement period (20-year). Auburn has been re-measuring the RLGS since on a 5-year cycle. Today, there are 305 plots in the RLGS, with nearly 40 percent of them on the EEF.

From the first 10 years of the study, Farrar published growth predictions and produced a site-index function for naturally regenerated longleaf pine in the East Gulf area (Farrar 1979, 1981, 1985). After the addition of the next 25 years' worth of data, Lauer and Kush (2010, 2011) updated Farrar's work. Unique in the Lauer and Kush site index equation is the ability to allow the user to specify the number of years of required for tress to reach 4.5-feet or diameter at breast height (DBH). Farrar's site index added 7 years to the ring count at DBH. The stand level growth and yield model by Lauer and Kush was developed with survival curves to provide improved information for timing of thinning to capture potential mortality. In addition, the models indicate longleaf is very well-suited to frequent thinning; there is a growth response following thinning regardless of age; and the species can be managed well beyond 100 years old and the trees will still grow (the oldest trees in the RLGS are 130 years old). This information should be very helpful to the USFS if they are planning 120-year rotations for their longleaf pine stands.

Over the past 30 years, SOFWS put additional efforts beyond plot re-measurement to utilize the RLGS plots; it was more than just a growth and yield study. The RLGS was ideal for providing information on a number of critical issues: mortality; climate change – time-rep plots; utility poles; pine straw; mapping and GIS layers; impacts from hurricanes; and forage for silvopasture, wildlife, or threatened and endangered species.

Mortality

There is not much that kills longleaf pine (Kush and others 2015). Many landowners fear if they manage longleaf pine for high-value products that lightning will kill those trees. Mortality due to lightning kills less than 0.5 trees/acre/year. The largest cause of death is due to suppression. This is the result of the nature of the study to determine just how dense of a stand longleaf pine can grow in before mortality happens in the smaller diameter classes. The RLGS does show you can kill longleaf pine with fire, especially in young, dense stands and a 3-year fire return interval. The RLGS has been severely impacted by hurricanes over the decades. Hurricane's Frederick, Opal, and Ivan destroyed a total 29 plots.

Climate

Rayamajhi (1996) found parameters in Farrar's (1979, 1985) model did not remain stable when used for longer projections periods. Meldahl and others (1997) using the work from Rayamajhi found climatic factors reduced error the growth and yield models. Precipitation and minimum temperature were the most important climate variables for modeling growth changes in longleaf pine. In addition, ignoring climate resulted in bias estimates for long-term growth projections.

When the time-rep series plots were examined, basal area increment/year was significantly different between the time series (Rayamajhi and others 1998, Rayamajhi and Kush 2006). Again, these series of plots are located on the EEF where soils and management are uniform. Those plots established in the mid-1960's grew significantly less than the series of plots established in the 1980's and 1990's. Something has changed to increase growth and this is not the only growth increase documented in the past few decades (Boyer 2001, West and others 1993).

Utility pole

For every tree > 5.6" DBH the RLGS documented whether or not the tree was a pole and if so, what was the length? Nearly 75 percent of the longleaf in a "typical natural stand" made poles. The percentage of poles in a stand increased rapidly with age to 60-80 years and then decreased as trees grew out of pole size. In addition, there were a higher percentage of poles on lower to medium site quality sites when compared to better sites (site index > 80 feet at base age 50) (Shaw and others 1991). Poles have been of interest for several decades because of the price they bring when compared to sawtimber.

Pine straw

As part of the Southern Global Change Program (SGCP), needlefall was monitored on a subset of RLGS plots for three years to determine productivity (Meldahl and others 1997). Needle production decreased linearly with age and increased with site index. Dyer and others (2012) used these data to develop an equation for the number of potential bales of pine straw a landowner could anticipate from a stand.

Hurricane impacts

Kush and Gilbert (2010), and Gilbert and Kush (2013) reported on the impacts of Hurricane Ivan on the RLGS plots. Well-stocked stands suffered little damage compared to plots in, or adjacent to, openings.

Understory biomass production

The RLGS is uniquely qualified to answer a number of questions regarding foraging habitat for wildlife species and for threatened and endangered species (T & E) such as the red-cockaded woodpecker and gopher tortoise. In addition, with the renewed interest in silvopasture (bringing grazing back to the South),

the RLGS could answer how much forage a landowner could expect from a stand. A subset of plots by basal area class on the EEF was subsampled for forage production. In areas with less than 30 square feet/acre there was nearly 400 pounds/acre of forage and good longleaf pine regeneration. For plots with a basal area between 30 and 70 square feet/acre, forage production dropped to 350 pounds/acre and longleaf regeneration was found in openings in the plots. On plots with greater than 70 square feet/acre, forage production rose to 450 pounds/acre with no regeneration. Despite the higher basal area, the frequent burning on the EEF has many areas with good forage production.

THE END OF THE RLGS?

What has the RLGS provided for longleaf pine management? Initial conclusion: Not as much as it could (should) have. There have many obstacles out there. Long-term research is not well-suited to today's lifestyle and our need for instantaneous information. Many look at longleaf but do not understand its silvics. Wahlenberg (1946, page 102) in his "Problems of Natural Reproduction" chapter wrote "...mismanagement of longleaf pine has been the rule rather than the exception, due to the ignorance of the unique life history and incomplete knowledge of factors determining the life and death of seedlings and hence the succession of forest types." Not much has changed in the 70 years since he penned this sentiment.

There has been an increase in longleaf acreage on public lands. However, we have not stopped the loss of longleaf on private lands; we are losing some of our most ecologically significant forests. We are not giving private land owners and land managers the needed reasons/information to keep longleaf pine. Landowners want and need information on a number of topics: timber type/species; stocking levels; markets; potential management interactions; natural resource enterprises and small business opportunities. There is a need for growth and yield information. They provide more than amount of timber or economic value. How do you manage for individual T & E species, entire ecosystems, aesthetics, and other multiple uses, or possibly meet military mission objectives and still produce some timber products? The answer resides in growth and yield (mortality) models. And the research from the RLGS has shown that model parameters are changing.

The RLGS has been incredible resource – an underutilized wealth of information and knowledge about longleaf ecology and management. It appears that the RLGS has come to an end just as it reached its 50th-year. While not perfect, the RLGS has data difficult to find anywhere about any species – a known history of: stem mapped trees; age classes; stand density and basal area classes; soils and site classes; pine straw and pole production; thinning history; and burning history.

Definitions: Looking - have the appearance or give the impression of being; seeing - discern or deduce mentally after reflection or from information; understand. Bob was right: we are not seeing what is happening to longleaf pine and what landowners want. We are not using our knowledge of the species and its silvics. It is a long-rotation species trying to make it in a very short-sighted world.

FINAL THOUGHTS

Bob would be the first to say longleaf pine was not for everybody but it offered opportunities other species did not. He believed longleaf should be grown with landowners and land managers in mind. Bob battled decades for longleaf pine and for the RLGS. And now it appears as if the RLGS is about to pass away. At a time when there is this discussion about climate change, data from the RLGS has shown an increase in growth over the past four decades. How can we not see where the next decades go? At a time when federal and state agencies talk about longer rotations for longleaf, the RLGS has data for plots beyond 120 years old. How can we not see where these plots will go?

The dedication to Wahlenberg's (1946) book reads "Dedicated to a future in American forestry for one of the finest timber trees the world has ever known". We are a short-sighted species living in our short-sighted world. Longleaf pine has never been a short-rotation species and will never be. If it becomes that it will no longer be longleaf pine.

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LITERATURE CITED

Boyer, W.D. 2001. A generational change in site index for naturally established longleaf pine on a southern Alabama Coastal Plain site. Southern Journal of Applied Forestry. 25(2): 88–92.

Dyer, J.F.; Barlow, R.J.; Kush, J.S.; Gilbert, J.C. 2012. Pine straw production: From forest to front yard. In: Butnor, J.R., ed. Proceedings of the 16th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 100-108.

Farrar, R.M., Jr. 1978. Silvicultural implications of the growth response of naturally regenerated even-aged stands of longleaf pine (*Pinus palustris* Mill.) to varying stand age, site quality, and density and certain stand measures. Athens, GA: University of Georgia. 132 p. Ph.D. dissertation.

Farrar, R.M., Jr. 1979. Growth and yield predictions for thinned stands of even-aged natural longleaf pine. Res. Pap. SO-156. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 78 p.

Farrar, R.M., Jr. 1981. A site-index function for naturally regenerated longleaf pine in the East Gulf area. Southern Journal of Applied Forestry. 5: 150-153.

Farrar, R.M., Jr. 1985. Volume and growth predictions for thinned even-aged natural longleaf pine stands in the East Gulf area. Res. Pap. SO-220. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 172 p.

Gilbert, J.C.; Kush, J.S. 2013. Longleaf pine regeneration following Hurricane Ivan utilizing the RLGS plots. In: Guldin, J.M., ed. Proceedings of the 15th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-175. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 137-139.

Kush, J.S.; Gilbert, J.C. 2010. Impact of Hurricane Ivan on the Regional Longleaf Pine Growth Study: Is there a relation to site or stand conditions? In: Stanturf, J.A., ed. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-121. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 259-261.

Kush, J.S.; Gilbert, J.C.; Barlow, R.J. 2105. What 45 years of RLGS data has to say about longleaf pine mortality – not much. In: Holley, A.G.; Connor, K.F.; Haywood, J.D., eds. Proceedings of the 17th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-203. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 473-477.

Lauer D.K.; Kush, J.S. 2010. Dynamic site index equation for thinned stands of even-aged natural longleaf pine. Southern Journal of Applied Forestry. 34(1): 28-37.

Lauer, D.K.; Kush, J.S. 2011. A variable density stand level growth and yield model for even-aged natural longleaf pine. Special Report No. 10. Auburn, AL: Auburn University, Alabama Agricultural Experiment Station. 16 p.

Meldahl, R.S.; Kush, J.S.; Rayamajhi, J.N.; Farrar, R.M., Jr. 1997. Productivity of natural stands of longleaf pine in relation to competition and climatic factors. In: Mickler, R.A.; Fox, S., eds. The productivity and sustainability of southern forest ecosystems in a changing environment. Ecological Series 128. New York: Springer-Verlag: 231-254.

Rayamajhi, J.N. 1996. Productivity of natural stands of longleaf pine in relation to climatic factors. Auburn, AL: Auburn University. 177 p. Ph.D. dissertation.

Rayamajhi, J.N.; Meldahl, R.S.; Kush, J.S. 1998. Stability of parameters that predict growth and yield of natural stands of longleaf pine (*Pinus palustris* Mill.). In: Waldrop, T.A., ed. Proceedings of the 9th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 509-514.

Rayamajhi, J.N.; Kush, J.S. 2006. Inclusion of climatic variables in longleaf pine growth models. In: Connor, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 367-370.

Shaw, D.J.; Meldahl, R.S.; Kush, J.S. [and others]. 1991. Pole availability from naturally regenerated longleaf pine stands: Preliminary data. In: Coleman, S.S.; Neary, D.G., comps., eds. Proceedings of the 6th biennial southern silvicultural research conference. Gen. Tech. Rep. SE-70. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 260-264.

Wahlenberg, W.G. 1946. Longleaf pine: Its use, ecology, regeneration protection, growth, and management. Washington, DC: Charles Lathrop Pack Forestry Foundation in cooperation with the Forest Service, U.S. Department of Agriculture. 429 p.

West, D.C.; Doyle, T.W.; Tharp, M.L. [and others]. 1993. Recent growth increases in old-growth longleaf pine. Canadian Journal of Forest Research. 23: 846-853.

PineMap

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EFFECTS OF FERTILIZATION AND THREE YEARS OF THROUGHFALL REDUCTION ON LEAF PHYSIOLOGY OF LOBLOLLY PINE

Charles J. Pell and Lisa J. Samuelson[1]

Abstract—Climate models project decreased soil water availability in the southeastern United States, which may impact loblolly pine (*Pinus taeda* L.) productivity. In conjunction with an interdisciplinary project known as PINEMAP, the objective of this study was to investigate the interactive effects of fertilization and a 30 percent reduction in throughfall on physiological characteristics that affect forest productivity. Stand growth, leaf area index (LAI) and leaf physiology were monitored over three years of treatment. Only the most recent preliminary results are presented here for June 2014. The study is a factorial combination of throughfall reduction (30 percent versus ambient) and fertilization (fertilized versus non-fertilized) treatments in a now 9-year-old loblolly pine plantation located in Georgia. No interactive effects of treatment were significant for any variable. Fertilization increased basal area by 19 percent, basal area increment by 27 percent, and peak LAI by 29 percent. Throughfall reduction reduced peak LAI by 17 percent. Fertilization decreased stomatal conductance, likely in response to higher LAI and increased water use combined with low ambient precipitation. These results indicate fertilization can increase growth and LAI but increased LAI may lead to short-term leaf physiological sensitivity to drought.

INTRODUCTION

Climate projections for the southeastern United States (U.S.) indicate that over the next century there is likely to be increases in annual temperatures along with increased frequency and intensity of extreme precipitation events (Kunkle and others 2012). Higher levels of evapotranspiration, due to a warmer atmosphere, along with decreases in soil moisture, resulting from increased precipitation runoff, will ultimately lead to increased drought severity (IPCC 2013, Kunkle and others 2012).

Historically, water availability was considered to be the dominant factor limiting forest productivity (Gholz and others 1990). However, more recent studies have identified nutrient availability as having a greater impact on forest productivity (Jokela and others 2004). The majority of available research has been focused on the effects of nutrient addition with irrigation. These studies have found that the greatest increase in productivity is realized when nutrients are supplied while water is not limiting, due to an additive effect (Albaugh and others 1998, Samuelson and Stokes 2006). It has been well established that increased nutrient availability, especially on nutrient poor sites, can increase loblolly pine productivity. Increasing nutrient and water availability can increase height, diameter, and volume as well as total leaf area (Wear and Greis 2012). However, low water availability coupled with increased evapotranspiration may reduce productivity because of decreased retention of foliage or reductions in leaf-level gas exchange (Allen and others 2005, Fox and others 2007). Increased drought intensity and frequency could decrease the efficacy of fertilization and therefore the productivity of loblolly pine.

Due to the importance of loblolly pine as an economic species and in order to investigate the effects of climate variability on loblolly pine productivity, the Pine Integrated Network: Education, Mitigation, and Adaptation Project (PINEMAP) was established. PINEMAP's goal is to enable southern pine landowners to better manage forests under a variable climate. As a part of PINEMAP's efforts, the objective of this study was to explore the interactive effects of reduced precipitation, achieved by throughfall reduction, and fertilization on physiological factors that affect the productivity of loblolly pine. We hypothesized that increased leaf area in response to fertilization will increase the sensitivity of leaf-level gas exchange to drought imposed by throughfall reduction. Preliminary results are presented from the most recent year (2014) of an ongoing three-year analysis.

METHODS

Experimental Design

This study was conducted on a 44.5 ha loblolly pine plantation located in Taliaferro county, GA (33° 37' 35.77"N, 82° 47' 53.48"W). The 30-year average annual precipitation is 1109 mm with average daily

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maximum temperatures of 22.7 °C and minimums of 10.1 °C (NOAA National Centers for Environmental Information, Asheville, NC, Annual Summaries, Station ID COOP: 099157, http://www.ncdc.noaa.gov accessed February 2015). The Palmer Drought Severity Index was downloaded for GA Climate Division 3 (http://www7.ncdc.noaa.gov/CDO/CDODivisionalSelect.jsp# accessed February 2015). Precipitation data was collected continuously with an onsite weather station and a local nearby weather station when needed (NOAA National Centers for Environmental Information, Asheville, NC, Daily Summaries, Station ID COOP: 099157, http://www.ncdc.noaa.gov accessed February 2015). The study was designed as a 2x2 factorial combination of throughfall reduction (approximate 30 percent reduction versus ambient) and fertilization (N, P, K, and micro nutrient fertilization versus non-fertilized) replicated with four blocks. Throughfall reduction was achieved by the use of throughfall exclusion trays which covered approximately 30 percent of the ground area. The fertilization treatment was applied once in March 2012. Each treatment plot measured 0.10 ha and contained approximately 136 trees. Further details on the experimental site and treatments were provided by Samuelson and others (2014).

Growth and LAI

A year-end stand inventory was conducted in November 2014 at a stand age of 9 years. Leaf area index (LAI) was measured at peak LAI in August 2014 using a pair of LAI-2000 plant canopy analyzers (LI-COR Inc., Lincoln, NE).

Leaf Physiology

Light-saturated leaf-level photosynthesis (P_{net}), stomatal conductance (g_s), and predawn leaf water potentials (Ψ_L) were measured approximately monthly from March 2014 to October 2014. Only June 2014 data are presented. Gas exchange measurements were performed using a LI-6400 portable photosynthesis system (LI-COR Inc., Lincoln, NE) and water potentials were measured using a pressure chamber (PMS Instrument Corp., Albany, OR). Shoots were collected from fully exposed sun foliage from the upper third of three randomly selected trees within each plot. Predawn water potentials were measured on the same three trees selected for gas exchange measurements.

Statistical Analysis

The measurement order of blocks and plots within blocks was random. Data were averaged by block, and plot. All variables were analyzed with a two way analysis of variance (ANOVA) using the GLM procedure in SAS 9.3. Treatment effects were considered significant at $\alpha \leq 0.05$.

RESULTS AND DISCUSSION

In 2014, annual precipitation was 988 mm, which was lower than the 30-year average of 1109 mm resulting in a dryer than normal year (fig. 1). The PDSI indicated drought conditions for the entirety of the year (fig. 1). June, for the purpose of this report, served as a representative month to demonstrate treatment effects on leaf-level physiology. Conditions in June were dry with only a total of 50 mm of rainfall where the 30-year average for this month is 113 mm (fig. 1). The PDSI for June indicated “mild drought” conditions with mild drought being defined as PDSI values between -1.00 and -1.99 (fig. 1).

There were no interactive effects of fertilization and throughfall reduction treatments detected for any growth or physiological variable. In 2014, fertilization had a significant impact on growth while throughfall reduction had no effect on growth. Basal area and basal area increment both were increased by fertilization by 19 and 27 percent respectively (data not shown). Fertilization increased peak LAI from 2.8 $m^2 m^{-2}$ to 3.7 $m^2 m^{-2}$ (table 1). Increased growth from fertilization was likely a direct result of increased LAI (table 1). Albaugh and others (1998) determined that increased LAI from fertilization was the primary driver behind increased growth in 8 year-old loblolly pine. Because LAI represents photosynthetic surface area, LAI and growth are tightly coupled in stands before canopy closure (Allen and others 2005, Campoe and others 2013, Samuelson and others 2004, Wang and Jarvis 1990).

In addition to fertilization, a main effect of throughfall treatment on LAI was also observed. Peak LAI was reduced by 17 percent in response to throughfall reduction treatment (table 1). Needle expansion has been shown to be sensitive to water availability in conifers (Raison and others 1992). Lower predawn Ψ_L with throughfall reduction treatment (table 2) likely reduced leaf turgor pressure, which in turn retarded needle development and elongation. However, there was no significant effect of the throughfall treatment on g_s or P_{net} despite lower predawn Ψ_L in response to throughfall reduction (table 2). Lack of a throughfall treatment effect on leaf gas exchange may be because of increased water availability per unit leaf area in response to a smaller crown.

The increase in LAI with fertilization observed in this study was irrespective of water availability, as has been reported by other studies with loblolly pine when the observed drought condition was not severe (Jokela and others 2004, Samuelson and others 2004, 2014). In contrast, an interactive effect of drought and fertilization was significant in a throughfall treatment study by Tang and others (2004); 100 percent throughfall exclusion during a severe drought negated the positive effect of

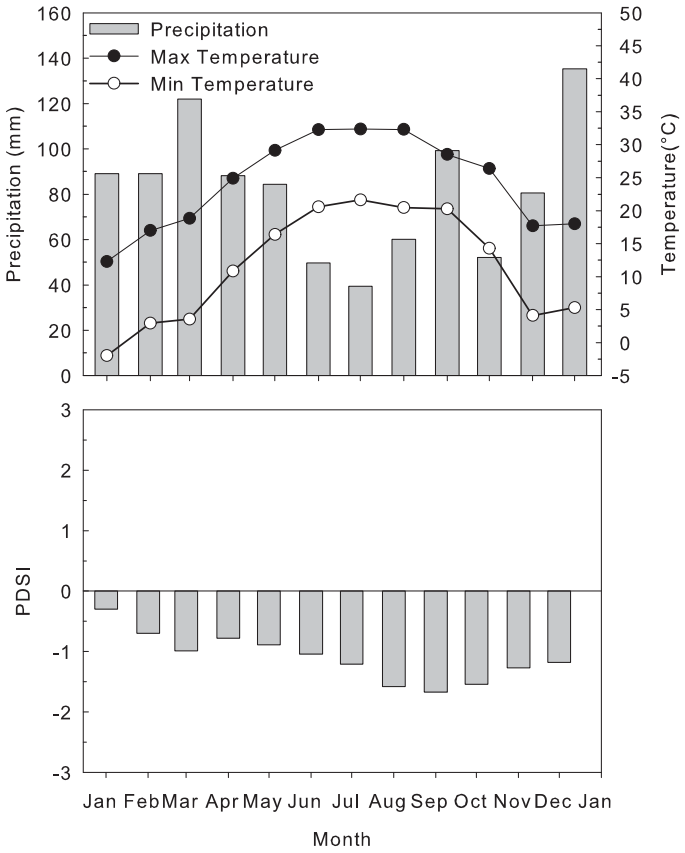


Figure 1—Monthly total precipitation, mean minimum and maximum daily temperature, and the Palmer Drought Severity Index for 2014.

Table 1—Mean (SE) responses and observed probabilities for the effects of throughfall (TR_0: ambient throughfall, TR_{30}: approximate 30% reduction) and fertilization ($Fert_0$: no fertilization $Fert_+$: one-time fertilization in March 2012) on peak LAI measured in August 2014. Bold values indicate significance at α ≤ 0.05

Treatment	Peak LAI ($m^2\ m^{-2}$)
TR_0	3.6 (0.2)
TR_{30}	3.0 (0.2)
$Fert_0$	2.8 (0.2)
$Fert_+$	3.7 (0.2)
P > F	
TR	**0.008**
Fert	**0.031**
TR x Fert	0.748

Table 2—Mean (SE) responses and observed probability values for the effects of throughfall (TR_0: ambient throughfall, TR_{30}: approximate 30% reduction) and fertilization ($Fert_0$: no fertilization $Fert_+$: one-time fertilization in March 2012) on light-saturated stomatal conductance (g_s), light-saturated leaf-level net photosynthesis (P_{net}) and predawn leaf water potential (Ψ_L) for June 2014. Bold values indicate significance at α ≤ 0.05

Treatment	g_s ($mmol\ m^{-2}\ s^{-1}$)	P_{net} ($\mu mol\ m^{-2}\ s^{-1}$)	Predawn Ψ_L (MPa)
TR_0	42.9 (6.5)	3.6 (0.3)	-0.72 (0.03)
TR_{30}	32.4 (3.1)	3.3 (0.2)	-0.83 (0.03)
$Fert_0$	45.4 (6.2)	3.8 (0.3)	-0.73 (0.04)
$Fert_+$	29.9 (2.2)	3.1 (0.2)	-0.82 (0.04)
P > F			
TR	0.126	0.361	**0.009**
Fert	**0.034**	0.062	**0.033**
TR x Fert	0.124	0.168	0.776

fertilization on leaf area production. In June 2014, the fertilization treatment reduced g_s from 45.4 mmol m^{-2} s^{-1} to 29.9 mmol m^{-2} s^{-1} (table 2), and predawn Ψ_L also decreased with fertilization (table 2). Therefore, higher LAI from fertilization and greater canopy water use induced some degree of water stress. Because of the nature of isohydric tree species such as loblolly pine, stomata close early in response to drought to reduce water loss and limit the decline in Ψ_L (Domec and Johnson 2012, Goldstein and others 2013).

In summary, fertilization had a positive effect on growth and LAI that was independent of throughfall treatment, but fertilization reduced g_s and predawn Ψ_L. In contrast, throughfall reduction did not affect leaf-level gas exchange and there was no reduction in growth in response to throughfall reduction despite a reduction in peak LAI and lower predawn Ψ_L. However, growth may decline in response to reduced LAI in subsequent years. Future analyses will include investigation of cumulative treatment effects over the three-year study, which will likely reveal more about the underlying mechanisms important to understanding the interaction between climate and growth in loblolly pine.

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LITERATURE CITED

Albaugh, T.J.; Allen, H.L.; Dougherty, M.P. [and others]. 1998. Leaf area and above-ground responses of loblolly pine to nutrient and water additions. Forest Science. 44(2): 317-328.

Allen, C.B.; Will, R.E.; McGarvey, R.C.; Coyle, D.R.; Coleman, M.D. 2005. The effect of water and nutrient availability on radiation use efficiency and gas exchange in fertigated stands of sweetgum and sycamore. Tree Physiology. 35(2): 191-200.

Campoe, O.C.; Stape, J.L.; Albaugh, T.J. [and others]. 2013. Fertilization and irrigation effects on tree level aboveground net primary production light interception and light use efficiency in a loblolly pine plantation. Forest Ecology and Management. 288: 43-48.

Domec, J.C.; Johnson, D.M. 2012. Does homeostasis or disturbance of homeostasis in minimum leaf water potential explain the isohydric versus anisohydric behavior of *Vitis nifiera* L. cultivars? Tree Physiology. 32(3): 245-248.

Fox, T.R.; Allen, H.L.; Albaugh, T.J. [and others]. 2007. Tree nutrition and forest fertilization of pine plantations in the southern United States. Southern Journal of Applied Forestry. 31(1): 5-11.

Gholz, H.L.; Ewel, K.C.; Teskey, R.O. 1990. Water and forest productivity. Forest Ecology and Management. 30: 1-18.

Goldstein, G.; Bucci, S.J.; Scholz, F.G. 2013. Why do trees adjust water relations and hydraulic architecture in response to nutrient availability? Tree Physiology. 33: 238-240.

IPCC. 2013 Climate Change 2013: the physical science basis. Contributions of working group I to the fifth assessment report of the intergovernmental panel on climate change. In: Stocker, T.F.; Qin, D.; Plattner, G.-K. [and others], eds. New York: Cambridge University Press. 1535 p.

Jokela, E.J.; Dougherty, P.M.; Martin, T.A. 2004. Production dynamics of intensively managed loblolly pine stands in the southern United States: A synthesis of seven long-term experiments. Forest Ecology and Management. 192: 117-130.

Kunkel, K.E.; Stevens, L.E.; Stevens, S.E. [and others]. 2012. Climate of the Southeast U.S. In: Regional climate trends and scenarios for the U.S. national climate assessment. NOAA Technical Report NESDIS 142-2. Washington, DC. U.S. Department of Commerce, National Oceanic and Atmospheric Administration. 98 p.

Raison, R.J.; Myers, B.J.; Benson, M.L. 1992. Dynamics of *Pinus radiata* foliage in relation to water and nitrogen stress: I. Needle production and properties. Forest Ecology and Management. 52: 139-158.

Samuelson, L.J.; Stokes, T.A. 2006. Transpiration and canopy stomatal conductance of five-year-old loblolly pine in response to intensive management. Forest Science. 52: 313-323.

Samuelson, L.J.; Johnsen, K.; Stokes, T.A. 2004. Production, allocation and stemwood growth efficiency of *Pinus taeda* L. stands in response to 6 years of intensive management. Forest Ecology and Management. 192: 59-70.

Samuelson, L.J.; Pell, C.J.; Stokes, T.A. [and others]. 2014. Two-year throughfall and fertilization effects on leaf physiology and growth of loblolly pine in the Georgia Piedmont. Forest Ecology and Management. 330: 29-37.

Tang, Z.; Sword-Sayer, M.A.; Chambers, J.L.; Barnett, J.P. 2004. Interactive effects of fertilization and throughfall exclusion on the physiological responses and whole-tree carbon uptake of mature loblolly pine. Canadian Journal of Botany. 82: 850-861.

Wang, Y.P.; Jarvis, P.G. 1990. Influence of crown structural properties and PAR absorption, photosynthesis, and transpiration in Sitka spruce: application of a model (MAESTRO). Tree Physiology. 7: 297-316.

Wear, D.N.; Greis, J.G. 2012. The southern forest futures project: summary report. Gen. Tech. Rep. SRS-GTR-168. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 54 pp.

PineMap
(Continued)

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TRADE-OFF BETWEEN FOREST PRODUCTIVITY AND CARBON SEQUESTRATION IN SOIL

A. Noormets[1]

With the growing fraction of the world's forests being intensively managed plantations, these ecosystems will increasingly be relied upon to provide other ecosystem services, in addition to merchantable timber. Schemes proposing the use of managed forests to mitigate climate change by sequestering carbon, however, are yet to be tested for feasibility and cost. In the current study we will review literature on forest responses to key climate and management factors on the inputs and outputs to the soil C pool in the context of expected changes under management. Recent global reviews indicate that there are distinct tradeoffs in allocation of assimilated carbon to different plant parts with productivity, and that the greater aboveground productivity comes on the account of belowground allocation, particularly on account of fine roots. Despite the proportional decrease in belowground fluxes in proportion to GPP, the absolute respiration fluxes are higher in managed than unmanaged forests, resulting in greater soil C loss. The balance between annual soil C inputs (leaf and fine root production plus mortality of tissue) and losses (soil heterotrophic respiration) indicates that while the imbalance is global, it is greater in the more productive managed than unmanaged forests. This shift triggered by high productivity exaggerates the elevated respiration losses caused by the greater frequency of harvests and physical disturbance of soil, outlining the greater vulnerability of soil C in managed forests.

Increasing global population and expanding land use mean that an ever greater percentage of human needs for wood products is being met by managed forests (Birdsey and Pan 2015, Foley and others 2005, Hansen and others 2013). Currently, about 7 percent of world's forests are plantations and 57 percent are secondary forests recovering from anthropogenic disturbance (FAO 2010). From 2000 to 2005 the rate of increase in the area of planted forests was 2 percent yr-1 and is accelerating (FAO 2009), whereas total forest area decreased at a rate of about 2 percent per decade. The intensive management practices developed over the past 5 decades have increased productivity of some plantation species by nearly 5-fold (Fox and others 2007), whereas the background increase globally is estimated 15-20 percent over the past century, attributable to forest regrowth, increased atmospheric CO2 and N deposition (Friedlingstein and others 2006, Piao and others 2011). Whether these changes translate long-term carbon sequestration depends on the allocation and chemical composition of the material, as well as the activity of heterotrophs that decompose the plant-derived detritus. Recent analyses highlight global patterns in allometry, whereby larger and faster growing trees shift their allocation patterns over different phases of canopy development (Chen and others 2013), which differ between natural and managed forests (Noormets and others 2015). Furthermore, there is growing evidence suggesting a decline in soil C pool in the world's forests, particularly in intensively managed plantations (Noormets and others 2015, Noormets and Nouvellon 2015). However, quantitative information about the role of forest management in this remains limited, and further explicit tests and improved understanding of belowground carbon dynamics are required. Although managed have both smaller soil C pool and higher Rh (Noormets and others 2015), the functional relationship to plant allocation and specific management practices is unclear and needs to be elucidated. In the current study, we will explore the relationship between soil C loss as quantified with heterotrophic respiration and belowground productivity.

This study is based on the literature review and the analysis of two global databases published earlier (Noormets and others 2015). The databases include the soil respiration database (SRDB) (Bond-Lamberty and Thomson 2010) and NPP database (Luyssaert and others 2009).

Soil C losses through heterotrophic respiration (Rh) increased in proportion with productivity, whether expressed on total or belowground basis (fig. 1). Although proportional allocation shifts with increasing plant size and NPP in favor or woody tissues and foliage, and away from fine roots (Chen and others 2013), BNPP may increase in absolute terms due to isometric effects. The increase in BNPP brings with it an increase in Rh (fig. 1, top), as a fraction of belowground allocation supports root exudation and the activity of

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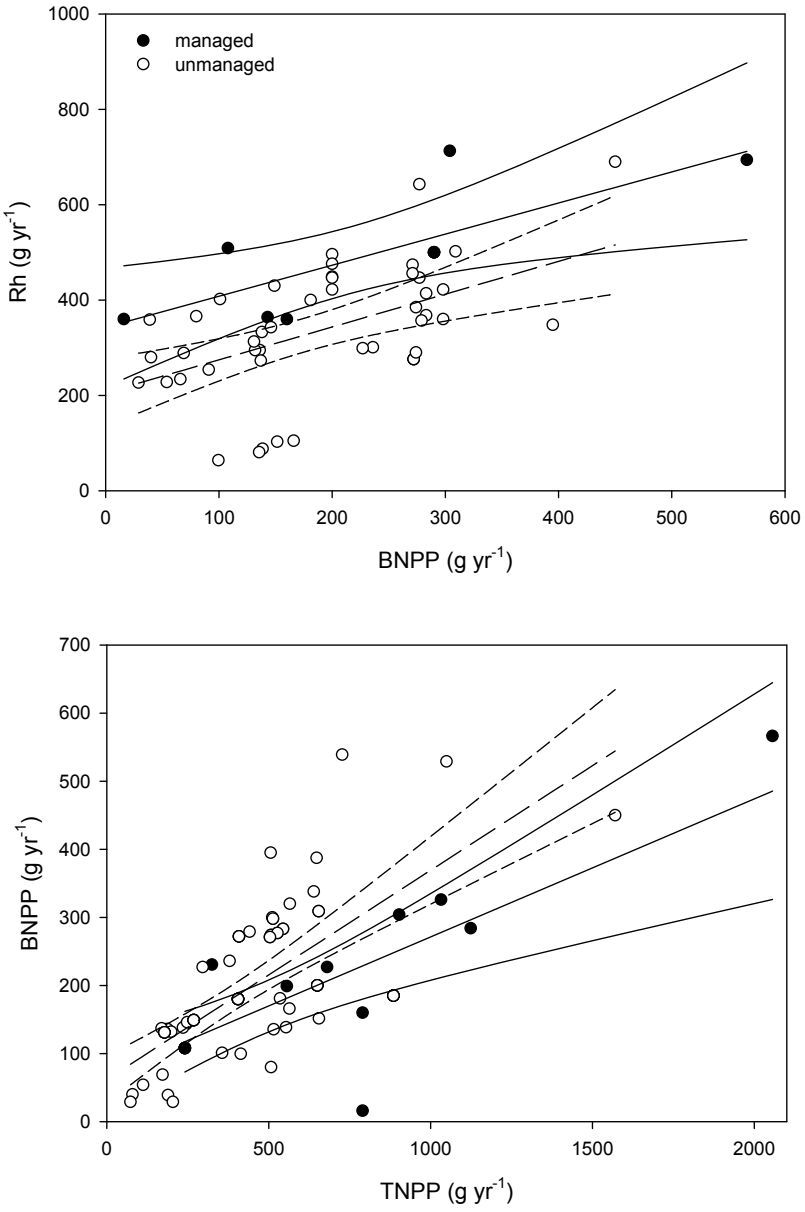


Figure 1—Top: The relationship between heterotrophic respiration (Rh) and belowground net primary production (BNPP) in managed (filled symbols and solid lines) and unmanaged (open symbols and dashed lines) temperate forests. Bottom: Relationship between BNPP and total net primary production (TNPP) in managed and unmanaged temperate forests. The linear regression fits include 95% confidence.

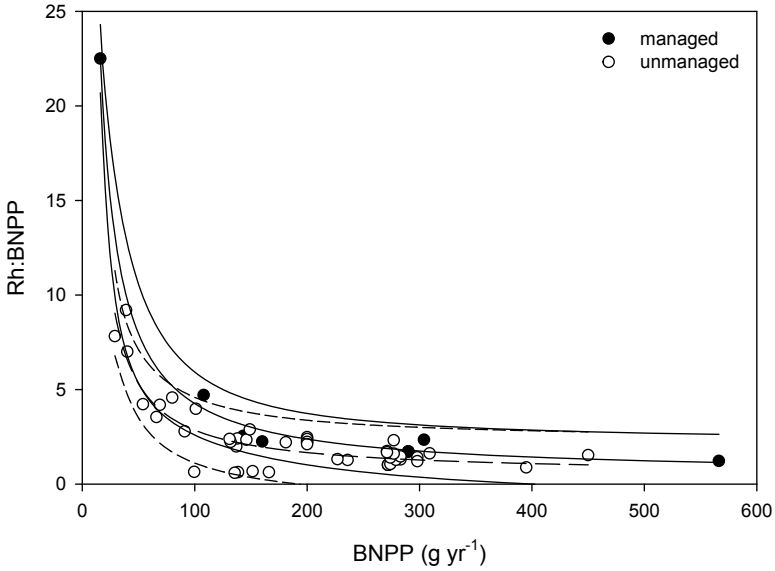


Figure 2—The relationship between the ratio of heterotrophic respiration to belowground net primary production (Rh:BNPP) to BNPP in managed (filled symbols and solid lines) and unmanaged (open symbols and dashed lines) temperate forests. The inverse second order polynomial models are plotted with 95% confidence intervals.

both root-associated and free-living microbes. This observation is consistent with the report by Chen and others (2014). Furthermore, Chen and others (2014) also showed that as the ratio of Rh:GPP (gross primary production) decreased with increasing productivity, that of root respiration increased.

Given the dependence of both belowground allocation and root exudation, as well as of microbial activity and Rh on GPP-derived carbohydrate substrates, respiration and GPP interact through both positive and negative feedback loops (Chen and others 2014, Noormets and others 2015). Despite lower proportional allocation to BNPP (fig. 1, bottom) the overall Rh was about 150 g m-2 yr-1 greater in managed than unmanaged forests (fig. 1, top). The ratio of Rh:BNPP decreased with BNPP, particularly at low productivity levels (young, recently disturbed stands), but still exceeded unity in most stands, even at high BNPP (fig. 2). Notably, even though Rh overall was higher in managed than unmanaged forests (fig. 1, top), the ratio of Rh:BNPP did not clearly differ by management status in any but the youngest stands (fig. 2).

These findings indicate that factors that contribute to increased Rh can lead to a decline in soil C, and that unadjusted Rh was greater in managed than unmanaged forests. On the other hand, managed forests that receive fertilizer inputs, have also been reported to have lower fine root production and exudation to the rhizosphere (Janssens and Luyssaert 2009, Maier and others 2004), which in turn suppresses microbial activity and reduces Rh (Fog 1988, Högberg and others 2003, Janssens and others 2010). As the result of the combined effect of increased photosynthesis, decreased belowground allocation and decreased root respiration, higher nutrient availability results in higher biomass production efficiency (defined as the ratio of NPP to GPP) (Vicca and others 2012). Consequently, managed forests may potentially sequester greater amounts of carbon belowground than their unmanaged counterparts (Noormets and others 2015), provided that the ongoing losses through Rh are offset by the harvest residue input at the end of the rotation cycle.

Factors contributing to increased productivity also increase belowground carbon flux and Rh. The observed net change reflects the counteracting effects of shifts in allocation patterns and isometric increases in absolute fluxes. Altered allocation patterns in managed forests decrease detritus production, as the improved nutrient availability and genetic selection favor preferential allocation to woody tissues and foliage over fine roots and root exudation. The isometric increase in total belowground C flux, on the other hand, may promote Rh through priming. The greatest contributor to Rh, however, is the harvest-related disturbance. The realized carbon sequestration in managed forests depends on the extent to which the effect of management activities on Rh can be minimized.

LITERATURE CITED

Birdsey R.; Pan, Y. 2015. Trends in management of the world's forests and impacts on carbon stocks. Forest Ecology and Management. 355: 83-90.

Bond-Lamberty, B.; Thomson, A. 2010. A global database of soil respiration data. Biogeosciences. 7: 1915-1926.

Chen, G.; Yang, Y.; Robinson, D. 2013. Allocation of gross primary production in forest ecosystems: allometric constraints and environmental responses. New Phytologist. 200: 1176-1186.

Chen, G.S.; Yang, Y.S.; Robinson, D. 2014. Allometric constraints on, and trade-offs in, belowground carbon allocation and their control of soil respiration across global forest ecosystems. Global Change Biology. 20: 1674-1684.

Food and Agriculture Organization. 2009. State of the world's forests 2009. Food and Agriculture Organization of the United Nations. Rome: Italy. 152 p.

Food and Agriculture Organization. 2010. Global Forest Resources Assessment 2010: progress towards sustainable forest management. Food and Agriculture Organization of the United Nations. Rome: Italy: 320-378.

Fog, K. 1988. The effect of added nitrogen on the rate of decomposition of organic-matter. Biological Reviews of the Cambridge Philosophical Society. 63: 433-462.

Foley, J.A.; DeFries, R.; Asner, G.P. [and others]. 2005. Global consequences of land use. Science. 309: 570-574.

Fox, T.R.; Jokela, E.J.; Allen, H.L. 2007. The development of pine plantation silviculture in the southern United States. Journal of Forestry. 105: 337-347.

Friedlingstein, P.; Cox, P.; Betts, R. [and others]. 2006. Climate-carbon cycle feedback analysis: results from the (CMIP)-M-4 model intercomparison. Journal of Climate. 19: 3337-3353.

Hansen, M.C.; Potapov, P.V.; Moore, R. [and others]. 2013. High-resolution global maps of 21st-century forest cover change. Science. 342: 850-853.

Högberg, M.N.; Bååth, E.; Nordgren, A. [and others]. 2003. Contrasting effects of nitrogen availability on plant carbon supply to mycorrhizal fungi and saprotrophs – a hypothesis based on field observations in boreal forest. New Phytologist. 160: 225-238.

Janssens, I.A.; Dieleman, W.; Luyssaert, S. [and others]. 2010. Reduction of forest soil respiration in response to nitrogen deposition. Nature Geoscience. 3: 315-322.

Janssens, I.A.; Luyssaert, S. 2009. Nitrogen's carbon bonus. Nature Geoscience. 2: 318-319.

Luyssaert, S.; Inglima, I.; Jung, M. 2009. Global forest ecosystem structure and function data for carbon balance research. http://daac.ornl.gov/. Oak Ridge, TN: Oak Ridge National Laboratory Distributed Active Archive. DOI: 10.3334/ORNLDAAC/949. [Date accessed: November 05, 2013].

Maier, C.A.; Albaugh, T.J.; Allen, H.L.; Dougherty, P.M. 2004. Respiratory carbon use and carbon storage in mid-rotation loblolly pine (*Pinus taeda* L.) plantations: the effect of site resources on the stand carbon balance. Global Change Biology. 10: 1335-1350.

Noormets, A.; Epron, D.; Domec, J.C. [and others]. 2015. Effects of forest management on productivity and carbon sequestration: a review and hypothesis. Forest Ecology and Management. 355: 124-140.

Noormets, A.; Nouvellon, Y. 2015. Introduction for special issue: Carbon, water and nutrient cycling in managed forests. Forest Ecology and Management. 355: 1-3.

Piao, S.; Ciais, P.; Lomas, M. [and others]. 2011. Contribution of climate change and rising CO_2 to terrestrial carbon balance in East Asia: a multi-model analysis. Global and Planetary Change. 75: 133-142.

Vicca, S.; Luyssaert, S.; Penuelas, J. [and others]. 2012. Fertile forests produce biomass more efficiently. Ecology Letters. 15: 520-526.

THE EFFECTS OF DECREASED WATER AVAILABILITY ON LOBLOLLY PINE (*PINUS TAEDA* L.) PRODUCTIVITY AND THE INTERACTION BETWEEN FERTILIZER AND DROUGHT

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Abstract—As part of the regional PINEMAP (Pine Integrated Network: Education, Mitigation, and Adaptation project) funded by the NIFA - USDA, we established a factorial study in McCurtain County, OK near Broken Bow. This study examined the effects of fertilization and ~30 percent reduction in throughfall on an seven-year-old loblolly pine (*Pinus taeda* L.) plantation. The objective of this study was to determine effects of decreased water availability and fertilization on tree growth. We measured tree and stand development as well as a suite of ecophysiological variables. Tree growth was reduced in throughfall exclusion treatments and increased in fertilizer treatments. At various times during the measurement period, throughfall exclusion reduced net photosynthesis, stomatal conductance, and mid-day leaf water potential. These results indicate that lower availability of soil moisture reduces leaf gas exchange and slows growth.

INTRODUCTION

Over the last half century, increases in pine plantation productivity and management in the southeastern United States has enabled these forests to produce more wood than any other region in the country (Fox and others 2007, Cubbage and others 2007). Loblolly pine have a greater growth rate than other commercially important southern pines and consist of over 50 percent of pine plantations across the Southeast are loblolly pine (*Pinus taeda* L.), thus making it the most commercially important tree species and a large driver of commercial timber production and habitat for numerous wildlife species (Baker and Langdon 1990, Dipesh and others 2015). Across the Southeast it is predicted that a shift towards a warmer and drier climate will likely occur. Increasing temperatures and decreasing or more variable precipitation is expected to impact forests productivity in the Southeast (IPCC 2007). More specifically, larger precipitation events with longer drying periods are predicted to lead to an increased intensity and duration of drought (Easterling and others 2000, Walsh and others 2014).

As part of the regional PINEMAP (Pine Integrated Network: Education, Mitigation, and Adaptation project) funded by the NIFA – USDA, this study examined the effects of fertilization and ~30 percent reduction in throughfall on a loblolly pine plantation over 2012, 2013, and 2014 growing seasons. The objective of this study was to determine effects of decreased water availability and fertilization on leaf gas exchange and tree growth. We hypothesized that fertilizer added to a water stressed loblolly pine plantation will lead to tree growth similar to that of an unfertilized, non-water stressed loblolly pine plantation.

SITE

This study was conducted on a privately owned loblolly pine plantation near Broken Bow in southeastern OK planted in January 2008. Planting density for the site was 2 by 3 m (~1650 trees ha^{-1}) Elevation for the site was 150 m. Soil consisted of Ruston fine sandy loam with 3 to 8 percent slopes, very deep, and well drained (U.S Department of Agriculture 2015). Depth to the water table is greater than 2000 mm. Precipitation over the three years was 1026 mm in 2012, 1312 mm in 2013, and 1289 mm in 2014. Average temperature was 15.4 °C with average maximum and minimum temperatures of 22.6 and 8.8 °C (Mesonet Weather 2015).

PROCEDURES

Experimental Design

The study consisted of four blocks (16 plots). Each block consisted of four treatment plots designed as a two by two factorial combination of fertilization and throughfall reduction. Plots had a 0.03 ha measurement plot for tree growth and yield within a 0.10 ha treatment plot. For leaf gas exchange, each plot had five measurement trees (80 total). Treatments include fertilization (optimum nutrition), ~30 percent reduction in throughfall, fertilization plus ~30 percent reduction in throughfall, and control with ambient throughfall and no fertilization. The elemental fertilization treatment

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consisted of common operational applications of 224 kg ha^{-1} nitrogen, 27 kg ha^{-1} phosphorus, and 56 kg ha^{-1} potassium. Nitrogen was applied as 432 kg ha^{-1} urea, phosphorus was applied as 140 kg ha^{-1} DAP, and potassium was applied as 112 kg ha^{-1} potash. To prevent micronutrient deficiency, granular oxysulfate micronutrient mix (Southeast Mix, Cameron Chemicals, Inc., Virginia Beach VA) was applied at a rate of 22.4 kg ha^{-1}. Throughfall reduction treatments consisted of precipitation throughfall exclusion troughs placed between tree rows designed to cover 30 percent plot ground area. Exclusion troughs were installed in late spring/early summer 2012 and were built of lumber and covered with clear U.V. stabilized plastic sheeting consisting of two layers co-extruded polyethylene and one layer of high strength polyester string (Tuff-Scrimm™ Poly 12, Americover Inc., Escondido, CA).

Measurements and Analysis

Volumetric soil water content (VWC) was measured every 4 to 6 weeks at 0 to12 cm with a HydroSense Soil Water Measurement System (Campbell Scientific, Inc, Logan, UT). Tree height and diameter growth was measured in January 2012 before treatments were implemented and in December following each growing season. Height growth was measured initially using a height pole. Once trees were taller than the height pole, tree height was measured with a hypsometer (Laser Technology, Inc., Centennial, CO, USA). Diameter growth was measured at DBH by averaging two caliper measurements taken at right angles. Similar to height growth, once trees outgrew the use of calipers, diameter tape was used. Total height and diameter growth was analyzed as a two by two factorial combination (n = 4). Leaf gas exchange was measured every 4 to 6 weeks from October 2012 through September 2014 on two fascicles taken from the upper third of tree using a portable photosynthesis system (LiCor 6400, LiCor Inc., Lincoln, NE). Mid-day leaf water potential was measured using a pressure chamber (PMS, Instrument Corp., Corvallis, OR) and coincided with leaf gas exchange measurements. Leaf gas exchange measurements and mid-day leaf water potential measurements were analyzed using a repeated measures analysis with an AR1 covariance structure (n = 4).

RESULTS AND DISCUSSION

Soil Moisture

Throughfall reduction reduced VWC. Ambient precipitation plus fertilizer treatments maintained greater VWC than control treatments. Across all plots, VWC at 0 to12 cm was reduced from 10.8 to 8.5 percent by throughfall excluders.

Growth

Fertilizer increased tree growth and throughfall reduction decreased tree growth. Throughfall reduction decreased total height growth over 2012, 2013, and 2014 growing seasons ($p < 0.0001$) compared with ambient precipitation treatments (fig. 1A). Throughfall reduction decreased total diameter growth over 2012, 2013, and 2014 growing seasons ($p = 0.0009$) and fertilization increased total diameter growth ($p = 0.0003$), compared with ambient precipitation and non-fertilized treatments, respectively (fig. 1B). Fertilization compensated for drier conditions in diameter growth, where throughfall reduction plus fertilization had similar growth as control treatments (fig. 1B).

Leaf Gas Exchange

Throughfall reduction decreased net photosynthesis and stomatal conductance ($p < 0.0001$). Fertilization decreased stomatal conductance ($p = 0.006$) and had no effect on net photosynthesis. Throughfall reduction caused increasingly negative mid-day leaf water potentials ($p < 0.0001$) and fertilization caused less negative mid-day leaf water potentials ($p = 0.02$).

Effects of Fertilization and Reduced Precipitation

These results indicate that lower availability of soil moisture reduces leaf gas exchange and slows growth. However, it was shown that fertilization can compensate for throughfall reduction. Tree growth was increased by fertilization and decreased by throughfall reduction. Additive effects of throughfall reduction and fertilization indicate that positive effects of fertilization are not eliminated when growth is reduced by throughfall reduction. Differences between the two responses determined whether volume growth decreased or increased when responses were combined, thus determining the net effect on growth. Throughfall reduction decreased net photosynthesis and stomatal conductance. These changes are related to increasingly negative mid-day leaf water potential and higher degree of tree water stress. Fertilizer decreased stomatal conductance and led to less negative mid-day leaf water potential, indicating how fertilization can be beneficial in loblolly pine plantations experiencing reduced water availability.

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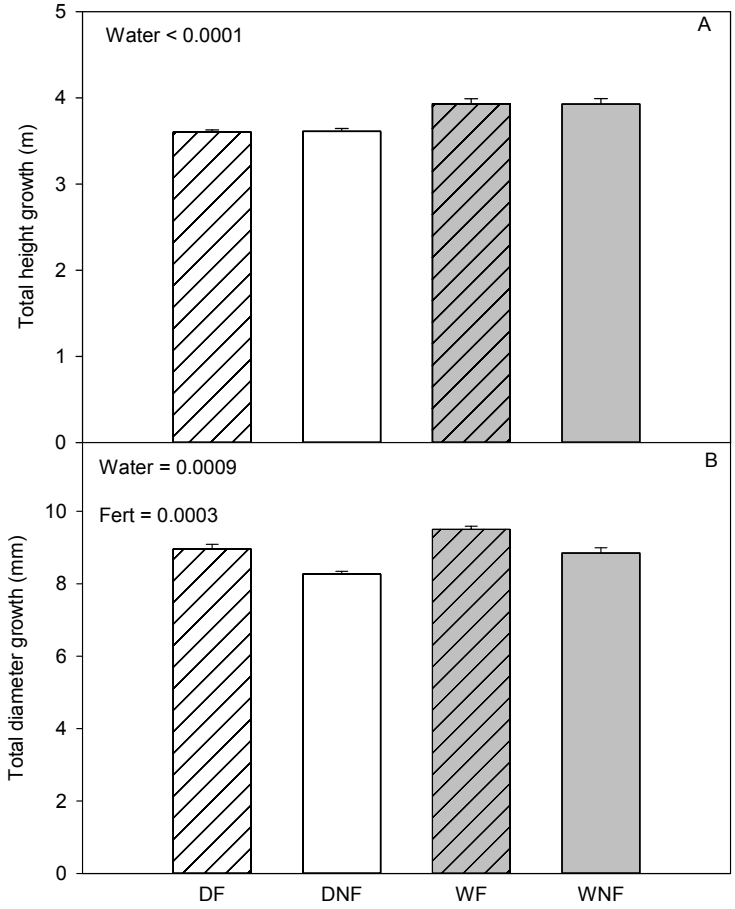


Figure 1—Total height growth (A) and diameter growth (B) by treatment over three growing seasons. DF = 30 percent throughfall reduction plus fertilization, DNF = 30 percent throughfall precipitation reduction with no fertilization, WF = ambient precipitation plus fertilization, and WNF = ambient precipitation with no fertilization.

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LITERATURE CITED

Baker, J.B.; Langdon, O.G. 1990. *Pinus taeda* L.-Loblolly Pine. In: Burns, R.M.; Honkala, B.H., tech. cords. Silvics of North America, Vol. 1. Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 497-512.

Cubbage, F.; Donagh, P.M.; Sawinski, J. [and others]. 2007. Timber investment returns for selected plantations and native forests in South America and the southern United States. New Forests. 33: 237-255.

Dipesh, K.C.; Will, R.E.; Lynch, T.B. [and others]. 2015. Comparison of loblolly, shortleaf, and pitch X loblolly pine plantations growing in Oklahoma. Forest Science. 61(3): 540-547.

Easterling, D.R.; Meehl, G.A.; Parmesan, C.; Changnon, S. 2000. Climate extremes: observations, modeling, and impacts. Science. 289: 2068-2074.

Fox, T.R.; Jokela, E.J.; Allen, H.L. 2007. The development of pine plantation silviculture in the southern United States. Journal of Forestry. 105: 337-347.

IPCC, 2007: Summary for policymakers. In: Climate change 2007: The physical science basis. Solomon, S.; Qin, D.; Manning, M. [and others]. eds. Contribution of working group I to the 4th assessment report of the intergovernmental panel of climate change. New York: Cambridge University Press: http://www.ipcc.ch/publications_and_data/ar4/wg1/en/spm.html [Date accessed: May 7, 2015].

Mesonet Weather. 2015. Daily data retrieval. http://www.mesonet.org/index.php/weather/daily_data_retrieval [Date accessed: May 7, 2015].

U.S. Department of Agriculture. 2015. Official series description-Ruston series. https://soilseries.sc.egov.usda.gov/OSD_Docs/R/RUSTON.html [Date accessed: May 1, 2015].

Walsh, J.; Wuebbles, D.; Hayhoe, K. [and others]. 2014. Ch.2 Our changing climate. In: Melillo, J.M.; Richmond, T.C. and Yohe, G.W., eds. Climate change impacts in the United States: The 3rd National Climate Assessment. Washington, DC: U.S. Global Change Research Program: 19-67.

Hardwood Management

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STEM DIAMETER DYNAMICS UNDER VARYING SHELTERWOOD TREATMENTS IN AN UPLAND HARDWOOD FOREST ON THE CUMBERLAND PLATEAU ESCARPMENT

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Abstract—Managing forests on the Cumberland Plateau escarpment for select desirable species can be particularly onerous due to the high diversity of dominant tree species. We implemented 5 treatments to alter species composition and structure in an effort to favor *Quercus* and maintain its dominance in the stands. Treatments were shelterwood prescriptions that in the first stage retained a percentage of the basal area (100, 75, 50, 25 and 0 percent retentions). After 9 growing seasons, the residual trees in all but the 0 percent retention treatment were removed. We installed permanent vegetation measurement plots and recorded species and diameter for all trees 1.5 inches dbh and greater in 2001 (pretreatment); 2002 (first growing season post stage one harvest); 2009 (8 years post stage one harvest), 2011 (first growing season post stage two harvest) and in 2014 (4 years post stage two harvest). None of the treatments increased *Quercus* stems. In the 0 percent retention, or clearcut, *Quercus* stems changed from 37 stems per acre (SPA), to 5 SPA immediately after harvest, to 24 SPA thirteen years post-harvest; while *Liriodendron tulipifera* stems increased from 16 SPA pretreatment to 523 SPA thirteen years later. In the 75 percent retention treatment (midstory herbicide in first stage; residual commercial harvest in final stage), for all species, SPA declined from 320 to 35; there were no *Quercus*, *Acer saccharum* or *L. tulipifera* stems found in 2014. The residual stems were *Carya ovalis*, *Fagus grandifolia*, *Fraxinus americana* and *Cercis canadensis*. Clearcutting and the 25 percent retention shelterwood showed the highest potential for recruiting *Quercus* into competitive size classes; additional intervention may be needed to control non-*Quercus* competitors.

INTRODUCTION

Cumberland Plateau forests, which includes those found in northeastern Alabama, are dominated by either *Quercus-Carya* upland types on the broad tabletops or intermediate mixed mesophytic and *Quercus-Carya* types on the side slopes, or escarpment (Braun 1950). These classifications result from local topographic and edaphic conditions, a consequence of geological uplifting and subsequent erosion. Over 30 canopy species can be found in the highly biodiverse forests of the Cumberland Plateau (Hinkle and others 1993). Myriad disturbances have influenced these forests and most stands are considered second or third growth (Hart and Grissino-Mayer 2008). Over 50 to 100 years, the result of these large-scale disturbances was a massive intrusion of oaks (*Quercus* spp.), yellow-poplar (*Liriodendron tulipifera* L.), ash (*Fraxinus* spp.), and other important species.

Today's upland hardwood forests in the Tennessee Valley of north Alabama and adjacent regions contain a mixture of species with wide ranges of shade tolerance and growth rates. Failure of *Quercus* to regenerate and recruit into smaller sapling size classes and the concurrent shift in dominance by mesophytic species remains a concern here as in other eastern forests (Nowacki and Abrams 2008). Manipulating light levels by reducing overstory and/or midstory stem densities is commonly recommended to promote oak over its competitors (Brose and others 2008, Loftis 1990a, Loftis 1990b, Parker and Dey 2008, Schweitzer and Dey 2011). The disturbance intensity and regime needed to accomplish this remains unknown, as prescriptions need to be site-specific. High intensity disturbances such as clearcutting may result in a conversion of stands to *L. tulipifera* (Beck and Hooper 1986, Groninger and Long 2008, Jenkins and Parker 1998, Loftis 1990b). Intermediate-intensity density reductions via shelterwood prescriptions have been tested as a means to alter light to favor *Quercus* over non-*Quercus* species (Johnson and others 2002, Loftis 1990a, Sander 1972, Schlesinger and others 1993, Schweitzer and Dey 2011, Spetich and others 2002).

The analysis of stand structure following differing levels of disturbance allows us to quantify changes in the residual structure and composition. As clearcutting may often have adverse social impacts, the two-phase harvesting associated with shelterwood prescriptions may result in a less severe visual impact

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and greater social acceptance. What is unknown in upland hardwood forests on the escarpment of the Cumberland Plateau is the level of disturbance needed to regenerate desired species, and the changes in the residual stands as prescriptions are implemented. In this paper, we evaluate the changes in sapling-sized and larger stems over two-phases of variable-retention shelterwood treatments, and clearcutting. The efficacy of these prescriptions on the regeneration response will be presented in elsewhere. The management goal is to maintain a similar species composition as current by mimicking the disturbance regime that gave us the stands we have today. The challenge is recruiting desirable species into competitive positions by altering the disturbance intensity.

METHODS

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

We used a randomized complete block design with three replications of five treatments. Each site (block) comprised one replication of five treatments established along the slope contour. One replication, located on Miller Mountain (34º 58' 11" N, 86º 12' 21" W) had a southwestern aspect and a mean elevation of 1600 feet. Two replications located at Jack Gap (34º 56' 30" N, 86º 04' 00" W) had northern aspects. One Jack Gap replication was located at 1496-feet elevation and the other at 1200-feet elevation. Treatments were randomly assigned to 10-acre areas within each replicated block because the land owner at the time of study implementation, Mead Corporation, mandated the maximum size of upland hardwood clearcuts at 10 acres. Dominant canopy tree taxa on both sites included *Quercus* that represented 46 percent of pretreatment basal area, including *Q. velutina* Lamarck, *Q. rubra* L., *Q. alba* L., *Q. montana* L. *Carya* species was 15 percent pretreatment basal area, *Acer saccharum* Marsh. was 13 percent pretreatment basal area and *L. tulipifera* was 9 percent pretreatment basal area. Common understory species included *Cornus florida* L., *Cercis canadensis* L., and *Oxydendrum arboretum* DC.

The treatments consisted of five levels of BA retention percentages: 0 (clearcut), 25, 50, 75, and 100 (untreated control; not harvested ≥ 40 years) (table 1). Shelterwood prescriptions were implemented in two phases. In the first phase, tree harvest for the 0, 25 and 50 percent retention treatments was accomplished by chain saw felling and grapple skidding, and done from fall 2001 through winter 2002. Trees in the 25 percent and 50 percent retention stands were retained on the basis of species [favoring oak, ash (*Fraxinus* spp.), and persimmon (*Diospyros virginiana*)], vigor class, and crown position. Trees were marked to be retained using guidelines outlined originally by Putnam and others (1960), and recently updated by Meadows and Skojac (2008). The 75 percent retention stands were treated using a herbicide (Arsenal®, active ingredient imazapyr) by means of tree injection technique in fall of 2001 to deaden the midstory. Rates of application were within the range recommended by the manufacturer. Watered solutions were made in the laboratory and then trees received application via waist-level hatchet wounds using a small, handheld sprayer. One incision was made per 3 inches of diameter and each incision received approximately 0.15 fluid ounce of solution. Herbicide treatments were completed in autumn 2001, prior to leaf fall. The goal was to minimize the creation of overstory canopy gaps while removing 25 percent of the basal area in the stand midstory. All injected trees were in lower canopy positions, reducing the creation of canopy gaps.

All 15 stands grew for eight years, prior to phase II, the final harvest in the fall of 2010. Merchantable trees (primarily those greater than 5.5 inches diameter at breast height, dbh) in the 25, 50, 75, and 100 percent retention stands were then removed through chainsaw felling and grapple skidding. The 0 percent retention treatment, clearcut in 2001, was not retreated. The second phase of treatment resulted in four forest cohorts: (1) nine-year old regenerating clearcut, (2) released regeneration with older non-merchantable residuals from the first phase harvest (25 and 50 percent retentions); this cohort differed due to phase I treatment, and included a well-established mid and understory in conjunction with scattered residual canopy trees, (3) released regeneration from the 75 percent retention, which lacked a well-developed midstory (mostly small-diameter sugar maple) and lacked scattered residuals and large sprouts, (4) new "clearcut", formerly the 100 percent retention which under phase I was not treated and under phase II had all overwood removed.

Prior to treatment, five measurement plots were systematically located in each treatment area. Plot

Table 1—Shelterwood retention study treatment descriptions and time frames for upland hardwood stands located in Jackson County, Alabama. Residual basal area targets were for merchantable stems, 5.6 inches dbh and greater

Treatment designation	History 2001	November 2001-December 2002 Phase I	2002 Data collection (growing seasons post Phase 1)	2009 Data collection (growing seasons post Phase 1)	2010 Phase II	2011 Data collection (growing seasons post Phase II)	2014 Data collection (growing seasons post Phase II)
Control	Pretreatment	No cut	1	8	All residual trees cut	1	4
75 percent retention	Pretreatment	Herbicide midstory	1	8	All residual trees cut	1	4
50 percent retention	Pretreatment	Cut to 40 ft^2/a basal area	1	8	All residual trees cut	1	4
25 percent retention	Pretreatment	Cut to 30 ft^2/a basal area	1	8	All residual trees cut	1	4
Clearcut	Pretreatment	Cut to 5 ft^2/a basal area	1	8	No additional cut	1	4

centers were permanently marked with a 24-inch piece of reinforcing steel, and GPS coordinates were recorded. At each plot center, a 0.025-acre plot was established and all trees 1.5 inches dbh and greater were monumented (distance and azimuth measured and recorded from plot center, each tree tagged with a numbered aluminum tag) and species and dbh recorded. Data were recorded in late summer of 2001, 2002, 2009, 2011, and 2014 (table 1). For this analysis, we combined all *Quercus* (*Q. alba*, *Q. veluntina*, *Q. montana*, *Q. rubra*, *Q. muehlenbergii* Englem.). We used an analysis of variance (ANOVA) by implementing PROC MIXED in the SAS 9.0 system (SAS Institute 2000) and we specified a random effect (block) and a repeated statement (time) with the type of covariance matrix assigned unstructured by TYPE=UN option specified as stand(treat). Differences in stems per acre and basal area and stems were assessed using Tukey's HSD test with significance set at α 0.05.

RESULTS AND DISCUSSION

Initially, stands contained 26 species and composition was typical for Cumberland Plateau escarpment forests. Pretreatment inventories showed that stands were fully stocked and had basal areas between 119.4 and 147.6 square feet per acre (ft^2/a) for all trees 1.5 inches and greater dbh, averaging 129.2 ft^2/a (table 2). Diameters ranged from 1.5 to 28.3 inches dbh. Trees with diameters greater than 20 inches dbh included *Q. alba*, *Q. rubra*, *Q. veluntina*, *Q. montana*, *L. tulipifera*, and *F. grandifolia*. Stem densities ranged from 291 to 347 stems per acre (SPA) (table 2). Across all stands, stem densities were dominated by *A. saccharum* (30.7 percent), *Quercus* (11.1 percent) and *L. tulipifera* (4.6 percent). Distribution of stems by diameter class for all stands resembled a typical inverse J-shaped curve (fig. 1). On average, 60.2 percent of the stems were between 1.5 and 5.5 inches dbh.

From 2001 through 2009, the control treatment accrued 24 additional SPA, and 14.4 ft^2/a of basal area (figs. 2 and 3). Ingrowth species were *F. grandifolia*, *Cercis canadensis* L., and *Magnolia acuminata* L. Control treatment basal area did not differ from that of the 75 percent retention treatment across phase I and phase II of the shelterwood treatments (table 2). In 2010, the control stands were essentially clearcut, resulting in 26.5 ft^2/a of residual basal area and 104 SPA. In 2014, four growing seasons post cut, the control stands had no stems greater than 15.5 inches, and 62.4 percent of the stems were 1.5-3.5 inches dbh (fig. 4). There were 37 SPA of *L. tulipifera* in the smallest diameter class, 13 SPA of *A. saccharum*, and no *Quercus*.

In the 75 percent retention treatment, the herbicide treatment targeted midstory trees and not overstory

Table 2—Stems per acre (SPA)(std) and basal area (BA ft²/a)(std) for all species 1.5 inches dbh and greater, under five shelterwood retention treatments, over four time periods, for upland hardwood stands located in Jackson County, Alabama. Times are as follows: 2001, pretreatment; 2002, one-year post treatment; 2009, 8 growing seasons post treatment; 2011, one year post final harvest; 2014, four years post final harvest. Phases of treatments and descriptions are given in Table 1

	2001		2002		2009		2011		2014	
	SPA	BA	SPA	BA	SPA	BA	SPA	BA	SPA	BA
Control	291a (112)	126.6a (71.8)	280a (105)	126.6a (71.8)	315a (100)	141.0a (80.1)	104c (135)	26.5a (37.5)	141c (186)	27.1a (38.9)
75 percent retention	320a (138)	120.6a (77.4)	117b (78)	94.6ab (77.4)	88b (50)	101.6ab (70.3)	19c (40)	14.1a (44.4)	35c (77)	14.4a (44.5)
50 percent retention	347a (100)	147.6a (79.3)	112b (111)	53.3bc (79.1)	336a (369)	58.6bc (91.0)	555b (428)	46.4a (94.0)	632b (438)	52.1a (94.9)
25 percent retention	331a (112)	119.4a (53.4)	136b (118)	43.6c (53.6)	419a (218)	52.6bc (57.6)	532b (415)	27.2a (22.0)	600b (407)	32.0a (23.1)
Clearcut	331a (156)	131.8a (70.0)	88b (148)	19.6c (28.0)	427a (206)	20.6c (21.2)	1021a (435)	43.1a (19.4)	1208a (527)	54.5a (21.2)

Different letters within columns indicate significant difference among treatments at α 0.05

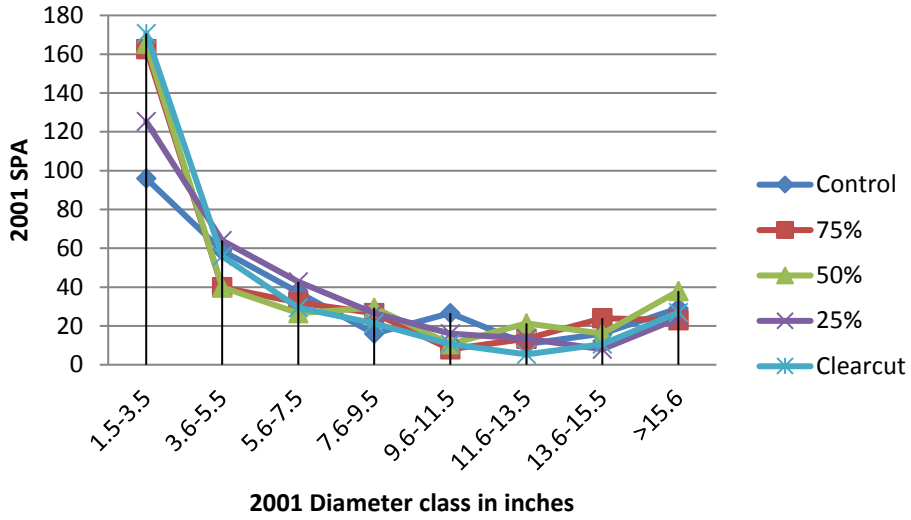


Figure 1—Pretreatment diameter distributions for all woody stems 1.5 inches dbh and greater in stands targeted for five silvicultural prescriptions on the Cumberland Plateau escarpment in Jackson County, AL. Phase I treatment assignments were Control (no treatment); 75 percent (shelterwood with initial midstory herbicide treatment to retain 75 percent of the basal area); 50 percent and 25 percent (shelterwood treatments with commercial harvests to retain 50 and 25 percent of the basal area); Clearcut (removal of all merchantable stems).

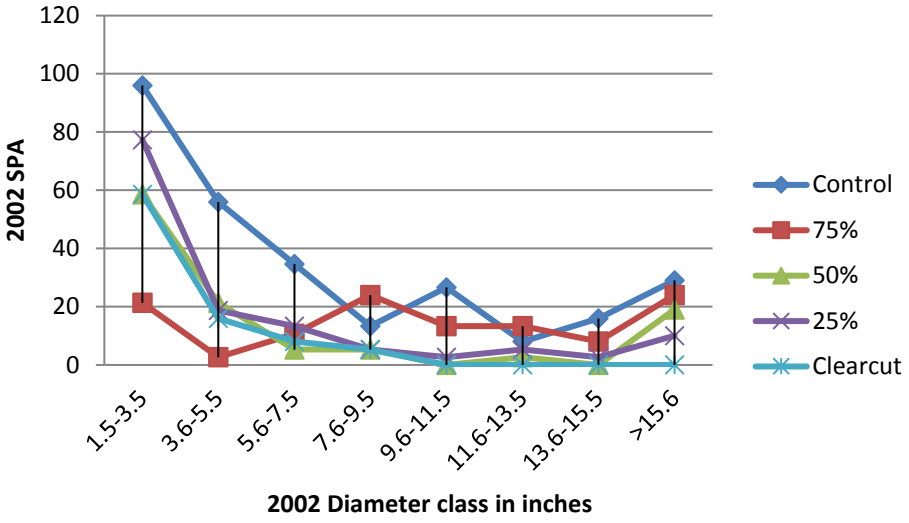


Figure 2—First-year diameter distributions following Phase I treatment for all woody stems 1.5 inches dbh and greater in stands under five silvicultural prescriptions on the Cumberland Plateau escarpment in Jackson County, AL. Phase I treatments were Control (no treatment); 75 percent (shelterwood with initial midstory herbicide treatment to retain 75 percent of the basal area); 50 percent and 25 percent (shelterwood treatments with commercial harvests to retain 50 and 25 percent of the basal area); Clearcut (removal of all merchantable stems).

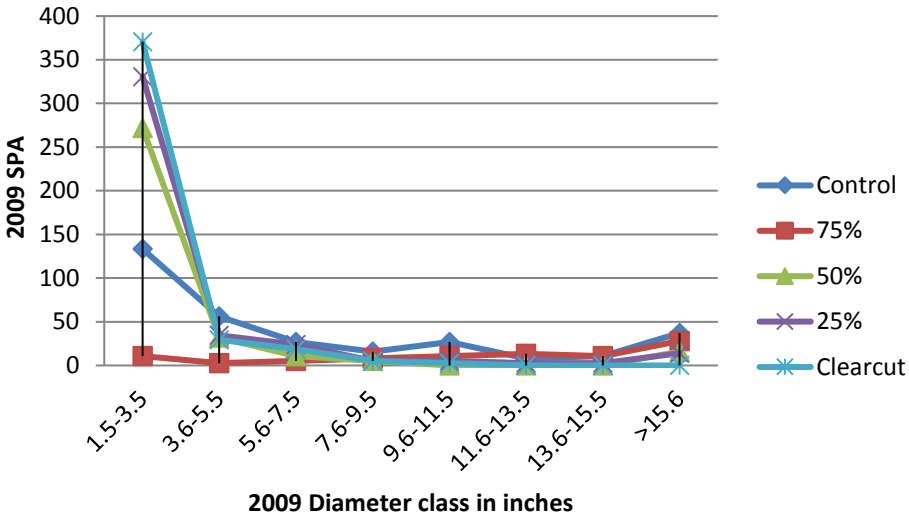


Figure 3—Eight-year diameter distributions following Phase I treatment for all woody stems 1.5 inches dbh and greater in stands under five silvicultural prescriptions on the Cumberland Plateau escarpment in Jackson County, AL. Phase I treatment assignments were Control (no treatment); 75 percent (shelterwood with initial midstory herbicide treatment to retain 75 percent of the basal area); 50 percent and 25 percent (shelterwood treatments with commercial harvests to retain 50 and 25 percent of the basal area); Clearcut (removal of all merchantable stems).

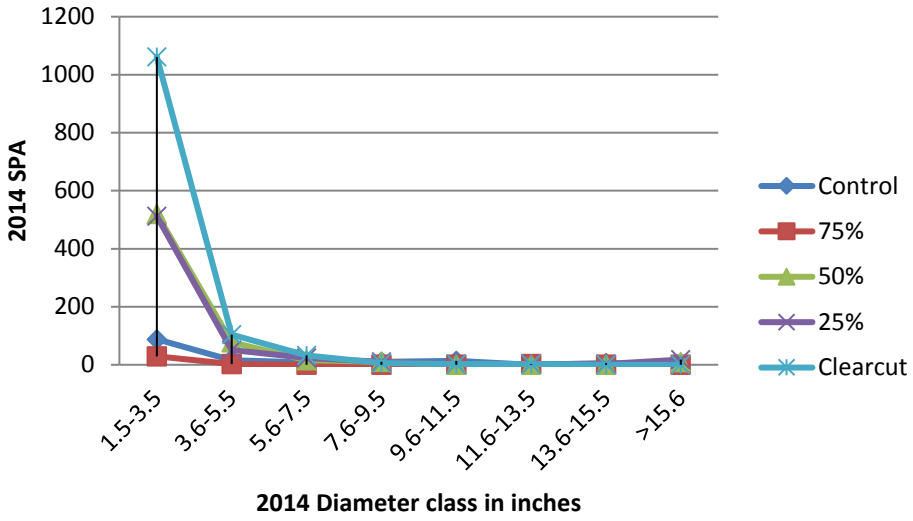


Figure 4—Four-year post final harvest (Phase II) diameter distributions for all woody stems 1.5 inches dbh and greater in stands under five silvicultural prescriptions on the Cumberland Plateau escarpment in Jackson County, AL. Phase I treatment assignments were Control (no treatment); 75 percent (shelterwood with initial midstory herbicide treatment to retain 75 percent of the basal area); 50 percent and 25 percent (shelterwood treatments with commercial harvests to retain 50 and 25 percent of the basal area); Clearcut (removal of all merchantable stems). Phase II removed all merchantable stems in the Control, 75 percent, 50 percent, and 25 percent retention treatments.

trees. We herbicided 202 SPA, ranging from 1.5 inches dbh up to 10.5 inches. The amount of basal area removed in this treatment was 19.4 ft^2/a, or 16.1 percent of the total. Initially, residual basal area differed from only the clearcut treatment, and stem densities only differed from that of the control treatment (table 2). Nine species were targeted in the herbicide treatment, with *A. rubrum* L. the primary species for removal (56 SPA treated), followed by *A. saccharum* (53 SPA treated) and *Nyssa sylvatica* Marsh. (40 SPA treated). Following the midstory removal, the diameter distribution curve changed, especially in comparison to the other treatments (fig. 2). There were 27 SPA of *Quercus* following the herbicide treatment, 13 SPA in the 7.5 to 19.5 inch diameter class, and 14 SPA greater than 19.5 inches dbh. There were also 13 SPA of *A. saccharum*, all less than 13.5 inches dbh, and 3 SPA of *L. tulipifera* in the 7.6 to 9.5 inch dbh classes. There were no stems of ingrowth recorded in the eight growing seasons post-herbicide. Prior to phase II, the final overstory removal harvest, these stands had 101.6 ft^2/a of basal area and 88 SPA, and after the harvest had 14.5 ft^2/a of basal area and 19 SPA (figs. 3 and 4). The residual stems were *Carya ovalis* Sarge., *F. grandifolia*, *Fraxinus americana* L. and *C. canadensis.* The *C. ovalis* was the largest diameter tallied at 11.5 inches dbh, while the other stems were all less than 4.0 inches dbh. These results were exactly the same four years after the final harvest (2014) as in 2011 (one year post harvest).

The 50 and 25 percent retention treatments reacted similarly to the initial and final harvest, and SPA and basal area did not differ over the course of study (table 2). When considering all stems 1.5 inches dbh and larger, the residual basal area for both these treatments was 36 percent of the pretreatment basal area. In the 50 percent retention treatment, stems were still distributed among the diameter classes, but there were no *Quercus* stems less than 15.5 inches dbh, and 11 SPA of *Quercus* between 15.6 and 19.5 inches dbh. Although total SPA continued to increase with time, *Quercus* SPA did not. In 2009, there were 91 SPA of ingrowth, and 37 of these were *L. tulipifera*. After the final harvest and four growing seasons, 520 SPA of the total 632 SPA were in the 1.5 to 3.5 inch diameter class, and 164 SPA of the 520 were *L. tulipifera*. There were over 100 SPA in the 3.6 to 9.5 diameter class, and 9 SPA of large canopy trees that were not removed (greater than 7.6 inches dbh). However, this is slightly misleading as we had one entire survey plot that was not harvested, and that plot accounted for all residual stems over 21.6 inches dbh, which were *Q. alba*, *Q. rubra* and *L. tulipifera*. The 25 percent retention treatment increased stem densities over time and those stems were also dominated by the smallest diameter class size, 512 SPA out of 600 at four years post final harvest. This treatment did result in the recruitment of 11 SPA of *Quercus* in the 1.5 to 5.5 inch dbh class; prior to treatment there were no *Quercus* less than 5.5 inches dbh (figs. 3 and 4).

Clearcutting resulted in the most dramatic change to diameter distributions, as well as increasing basal area by 34.9 ft^2/a following 13 growing seasons. Immediately after the harvest, there were no stems greater than 9.5 inches dbh, but eight years later there were 6 SPA greater than 9.5 inches dbh (fig. 2). The greatest number of ingrowth stems occurred in this treatment, with 227 new stems tallied eight growing seasons postharvest (fig. 3). These stems were dominated by *L. tulipifera*, *Robinia pseudoacacia* L., and *Prunus serotina* Ehrh. Clearcut stands had significantly greater SPA in 2014 (table 2) than the other four treatments in this study. Of the 1208 SPA, 1061 SPA were in the 1.5-3.5 inch dbh class, and 485 SPA of these were *L. tulipifera* (fig. 4). *Quercus* density changed from 38 SPA pretreatment, somewhat evenly distributed among diameter classes but with none in the smallest size class, to 24 SPA after 13 growing seasons, with 16 SPA in the 1.5 to 5.5 inch dbh class. *Liriodendron tulipifera* SPA changed from 16 SPA pretreatment, all under 9.5 inches dbh, to 522 SPA in this same size range. *A. saccharum* density in the 1.5 to 5.5 dbh class changed little, from 98 SPA pretreatment to 88 SPA 13 growing seasons after treatment.

Reducing stem density from below using herbicides has been reported to create conditions necessary for growth of small *Quercus* stems (Loftis 1990a). Loftis (1978) noted that this treatment also removes a source of sprout competition, which was obvious in our study. Four years after final overstory removal, the stands remained sparsely populated by woody stems. We did tally 9380 SPA of reproduction (1 foot tall up to 1.5 inches dbh), which was 9.8 percent *Quercus*, 10 percent *A. saccharum*, and 21 percent *L. tulipifera*, with over 32 percent of the stems greater than 4.5 feet tall and less than 1.5 inches dbh. While there is potential for these stands to regenerate to desirable woody species, the resultant stand structure four years after final harvest appears decimated and resembles a shrubby abandoned field. The interim visual appearance following phase II of the oak shelterwood may be perceived negatively and its initial structure and composition should be detailed to landowners. We found that the increase in light created by deadening the midstory was ephemeral, lasting only a few growing seasons prior to that space being occupied by *A. saccharum*. (Schweitzer and Dey 2011). In order for this low intensity shelterwood treatment to provide increased light over a longer time frame, we suggest deadening or removing a few dominant canopy trees during phase I to create gaps in the canopy. In these systems with both shade intolerant *L. tulipifera* and tolerant *A. saccharum*, the amount of light will need to be carefully controlled. Not enough light will stimulate *A. saccharum* in the understory, while too much light will stimulate *L. tulipifera*. We did note an increase in *L. tulipifera* in the understory after the initial treatment, but these newly emerged seedlings slowly faded out as the *A. saccharum* occupied the midstory space. The *L. tulipifera* positively responded to the final harvest and are quickly dominating, and the majority of stems will soon be in sapling sizes.

In these aggregating hardwood forests, the absence or low density of small stems of *Quercus* strongly suggests that these species will decline in importance in the next stand. The emergence of *Quercus* 25-30 years after stand initiation may occur, however, even if there are relatively few *Quercus* in subordinate positions early in stand development. This *Quercus* development could be stimulated by a targeted intermediate treatment to control *Quercus* competitors in the succeeding stands. In clearcut upland hardwood forests in southern Indiana, Morrissey and others (2008) found that *Quercus* persisted during the stem exclusion stage (21-35 years after harvesting) in association with *L. tulipifera*. If the shelterwood method is going to be successful in regenerating *Quercus*, established *Quercus* must be of competitive size, such as the 5 foot height given by Sander (1972). If we use 1.5 inches dbh as the acceptable cut-off size for competitive *Quercus*, which is reasonable on these higher productivity sites that have an abundance of early-successional and fast growing *L. tulipifera*, none of the shelterwoods tested in this study would meet a desired stocking goal for *Quercus*, regardless of the end density. In upland hardwood stands in North Carolina treated with two-phase shelterwood harvests, Loftis (1983) found 16 years after harvest oak seedling density increased, but tolerant hardwoods dominated. In our stands, we also have the threat of tolerant *A. saccharum* moving into larger and more competitive size classes. As with Groninger and Long (2008), we found that clearcutting was more effective for maintaining *Quercus* compared to any of the two-stage shelterwoods, but not without a concurrent stimulation of both *L. tulipifera* and *A. saccharum*. On a productive site in the southern Appalachians of North Carolina, Loftis (1978) reported that four years after clearcutting stands treated with a preharvest herbicide treatment resulted in the same number of desirable stems 0.5 inch dbh and greater as a complete clearcut with no pretreatment. Without post-clearcutting competition control, *Quercus* will most likely lose dominance in Cumberland Plateau escarpment forests.

CONCLUSIONS

In an attempt to mimic the perfect storm of disturbances that resulted in the forest composition we have today, we are testing various residual basal area retention shelterwoods to ascertain how a change in basal area, stem density, and subsequent light levels and competition, impact Cumberland Plateau stands. Although studies in similar areas have reported that the conditions created by the first phase of the

shelterwoods were conducive to promoting desired species such as *Quercus*, few have followed stand structure and associated dynamics after the final harvest. We should use caution in interpreting results at any intermediary stage. In our study, desirable stems of *Quercus* have not recruited beyond the regeneration stratum. However, in terms of esthetics and wildlife habitat creation, these successive disturbances may appeal to some land owners.

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LITERATURE CITED

Bailey, R.G. 1995. Description of the ecoregions of the United States. 2d ed., rev. Misc. Publ. 1391. Washington, DC: U.S. Department of Agriculture. 108 p. + map.

Beck, D.E.; Hooper, R.M. 1986. Development of a Southern Appalachian hardwood stand after cutting. Southern Journal of Applied Forestry. 10: 168-172.

Braun, E.L. 1950. Eastern Deciduous Forests of North America. Philadelphia: The Blakiston Company: 596 p.

Brose, P.H.; Gottschalk, S.B.; Horsley, P.D. [and others]. 2008. Prescribing regeneration treatments for mixed-oak forests in the Mid-Atlantic region. Gen. Tech. Rep. NRS-33. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 100 p.

Fenneman, N.M. 1938. Physiography of Eastern United States. New York: McGraw-Hill Book Company. 714 p.

Groninger, J.W.; Long, M.A. 2008. Oak ecosystem management considerations for central hardwood stands arising from silvicultural clearcutting. Northern Journal of Applied Forestry. 25(4): 173-179.

Hart, J.L.; Grissino-Mayer, H.D. 2008. Vegetation patterns and dendroecology of a mixed hardwood forest on the Cumberland Plateau: Implications for stand development. Forest Ecology and Management. 255: 1960-1975.

Hinkle, C.R.; McComb, W.C.; Safely Jr.; J.M., Schmalzer, P.A. 1993. Mixed mesophytic forests. In: Martin, W.H., Boyce, S.G., Echternacht, A.C., eds. Biodiversity of the Southeastern United States: upland terrestrial communities. New York: John Wiley: 203-253.

Jenkins, M.A.; Parker, G.R. 1998. Composition and diversity of woody vegetation in silvicultural openings of southern Indiana forests. Forest Ecology and Management. 109: 57-74.

Johnson, P.S.; Shifley, S.R.; Rogers, S. 2002. The ecology and silviculture of oaks. New York: CABI Publishing. 503 p.

Loftis, D.L. 1978. Preharvest herbicide control of undesirable vegetation in southern Appalachian hardwoods. Southern Journal of Applied Forestry. 2(1): 51-54.

Loftis, D.L. 1983. Regenerating Southern Appalachian mixed hardwoods with the shelterwood method. Southern Journal of Applied Forestry. 7(4): 212-217.

Loftis, D.L. 1990a. A shelterwood method for regenerating red oak in the Southern Appalachians. Forest Science. 36(4): 917-929.

Loftis, D.L. 1990b. Predicting post-harvest performance of advance red oak reproduction in the southern Appalachians. Forest Science. 36: 908-916.

Meadows, J.S.; Skojac, D.A., Jr. 2008. A new tree classification system for southern hardwoods. Southern Journal of Applied Forestry. 32(2): 69-79.

Morrissey, R.C.; Jacobs, D.F.; Seifert, J.R. [and others]. 2008. Competitive success of natural oak regeneration in clearcuts during the stem exclusion stage. Canadian Journal of Forest Research. 38: 1419-1430.

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the eastern United States. BioScience. 58(2): 123-138.

Parker, W.C.; Dey, D.C. 2008. Influence of overstory density on ecophysiology of red oak (*Quercus rubra*) and sugar maple (*Acer saccharum*) seedlings in central Ontario shelterwoods. Tree Physiology. 28: 797-804.

Putnam, J.A.; Furnival, G.M.; McKnight, J.S. 1960. Management and inventory of southern hardwoods. Agric. Handb. 181. Washington, DC: U.S. Department of Agriculture. 102 p.

Sander, I.L. 1972. Size of oak advance reproduction: key to growth following harvest cutting. Res. Pap. NC-79. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 6 p.

SAS Institute Inc. 2000. SAS Version 8.01. Cary, NC: SAS Institute Inc. 1686 p.

Schlesinger, R.C.; Sander, I.L.; Davidson, K.R. 1993. Oak regeneration potential increased by shelterwood treatments. Northern Journal of Applied Forestry. 10(4): 149-153.

Schweitzer, C.J.; Dey, D.C. 2011. Forest structure, composition, and tree diversity response to a gradient of regeneration harvests in the mid-Cumberland Plateau escarpment region, USA. Forest Ecology and Management. 262: 1729-1741.

Smalley, G.W. 1982. Classification and evaluation of forest sites on the Mid-Cumberland Plateau. Gen. Tech. Rep. SO-38. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Research Experiment Station. 58 p.

Spetich, M.A.; Dey, D.C.; Johnson, P.S.; Graney, D.L. 2002. Competitive capacity of *Quercus rubra* L. planted in Arkansas' Boston Mountains. Forest Science. 48: 504-517.

STAND DEVELOPMENT PATTERNS FOR YOUNG PLANTED OAK STANDS ON BOTTOMLAND HARDWOOD RESTORATION SITES

Jonathan Stoll, Brent R. Frey[1]

Abstract—Over the last several decades, bottomland restoration efforts have established hundreds of thousands of acres of planted hardwood stands throughout the Southeastern U.S. Much of the initial research focused on planting approaches and early growth and survival, contributing to more effective establishment methods. However, less research attention has been focused on stand development and treatment options for these planted stands as they age. Given that many afforestation stands are approaching 20 years of age or greater and undergoing crown closure, an improved knowledge of stand conditions is needed to evaluate opportunities for silvicultural treatments aimed at enhancing growth, and/or structural or compositional diversity. An increasing demand for such knowledge by landowners is becoming evident particularly by those that have participated in hardwood planting initiatives including the Conservation Reserve Program and Wetlands Reserve Programs (CRP and WRP respectively). This study investigates tree and stand development within young (10 to 20 year old) planted oak stands across a range of stand ages and site conditions in the Lower Mississippi Alluvial Valley. Growth and yield and stand development were evaluated using field surveys of stand characteristics (species, diameter, density, height, and crown measurements) and destructive sampling of individual trees. Our results show that as stands reach 20 or more years of age individual trees begin to interact, crown closure occurs, and conditions approach full stocking (the "A-line") (Goelz 1995). At this point, stems have generally reached merchantable diameters for pulpwood, and self-pruning has, for a stand fully stocked, progressed to one log-length in height (17.3 feet). These results suggest that thinning treatments could be merchantable and desirable from a tree growth perspective, in addition to potentially enhancing desirable stand conditions for wildlife habitat. As such, this information can provide a basis for informing silvicultural treatments aimed at improving stand conditions. Improving our knowledge of stand development, and growth and yield, could prove critical for ensuring the continued commitment of landholders to the management of their hardwood plantings and ongoing participation in these restoration programs.

INTRODUCTION

The Lower Mississippi Alluvial Valley (LMAV), locally known as "The Delta," was once an almost continuous hardwood forest spanning 24 million acres from southern Illinois to Louisiana (Twedt 2004). In the last two centuries, the LMAV has been largely cleared for agriculture, with less than a quarter of the original forest cover remaining today (Twedt 2004). Recognizing the widespread loss and fragmentation of LMAV bottomland forests and the ecosystem services they provide, private and government interest in afforesting marginal agricultural land has risen sharply in the last several decades (Allen 1997, Twedt 2004). Since the 1980's, there have been hundreds of thousands of acres of marginal-agricultural land in the Southeastern United States planted back to hardwood forest for the purposes of soil conservation and enhancement, water quality, timber production, and wildlife habitat often through Conservation Reserve Program and Wetland Reserve Program contracts (CRP and WRP respectively) (Gardiner and others 2004, Twedt 2004). Early efforts at hardwood afforestation on marginal agricultural lands focused on heavy-seeded species such as oaks (*Quercus* spp.) and pecan (*Carya illinoinensis* (Wangenh.) K. Koch) to improve the speed at which succession can take place; the goal being to return land to diverse, structurally complex forests that can provide economic, wildlife, and ecosystem values to the region (Stanturf and others 2000). Decades later, few studies (noted above) have assessed whether the recent forest restoration is progressing toward these goals and a definition for success remains elusive (Stanturf and others 2001).

Many old field sites in the LMAV have limited natural seed sources due to decades of intensive agriculture that have left remnant trees and forest patches at the extreme edges of fields, leaving large agricultural fields well beyond the dispersal distances of most tree species (Battaglia and others 2008). Left alone, the

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typically large agricultural fields in the Delta would take decades or even centuries for heavy seeded species to naturally invade large portions of the field. Because of this, reforestation sites have typically been planted with one to three heavy seeded species, with the expectation of natural recruitment by better dispersed species. Oaks (Quercus spp.) are some of the most valuable species both economically and functionally to the ecosystem and have been, by far, the most heavily planted group of species in hardwood afforestation efforts in the LMAV (Schoenholtz and others 2001). Early afforestation efforts also included direct seeding (at around 4000 acorns per acre), but direct seeding often had poor success and lacked predictability relative to planting (Allen 1990, Lockhart and others 2003, Twedt and Wilson 2002). Due to the poor survival of many direct seeded sites, planting with 1-0 bare root seedlings has been the preferred approach. However, due to the costs of hardwood afforestation, and the expectation of natural recruitment from other species, most stands, at least those established through CRP or WRP contracts, were planted at a relatively low spacing of 12 x 12 or 10 x 15 feet (Allen 1997, Meadows and Goelz 2001, Twedt and Wilson 2002). Unfortunately, in many cases, this afforestation approach has produced many nearly pure oak stands, particularly where survival is high and natural recruitment of other species is limited (Allen 1997).

Many stands are now well beyond the establishment phase and undergoing crown closure, and concerns have been raised about development of these well-stocked, mono-dominant oak stands, including concerns about a lack of species diversity, poor vertical structure, and poor self-pruning. Unlike natural stands which establish at much higher densities, trees in lower density, planted stands do not appear to be naturally pruning at a young age and thus have yet to produce much clear wood, which may reduce timber quality at maturity. This study aims to describe structural conditions within young (10-20 year old) planted stands with a focus on crown development. Oaks, like many hardwoods, are generally decurrent in form and respond to light by expanding their crowns into open growing space (Oliver and Larson 1996); this complicates plantation management. How individual oak trees and their crowns develop in relatively low-density, mono-dominant conditions has not been well described and represents a particular knowledge gap for management of afforestation stands. By taking inventory of stands across an age range within which canopy closure occurs, we hope to describe crown development in individual trees as it relates to stands conditions. With such information, better guidelines for silvicultural treatments such as thinning can be developed.

METHODS

The study used two approaches: a field survey of stand conditions, and an analysis of individual trees using destructive sampling. Sampling sites included state Wildlife Management Areas (WMA), mitigation areas, and private lands enrolled in WRP. The LMAV sites are typically characterized by heavy soils, often Sharkey clay, that would otherwise be difficult to farm. Other than one site in Louisiana, all of these forests are outside the Mississippi River levee system and are rarely inundated by more than one foot of flood water for extended periods. Of the range of possible stands to inventory, the focus for this study was on oak dominated stands, 10-20 years of age that had good survival to date. Plots were established randomly within the selected stands, at least 50 feet from the stand edge. Species composition in the sampled stands was most commonly a single oak species or a mix of Nuttall (*Quercus texana* Buckley) water (*Quercus nigra* L.), willow (*Quercus phellos* L.) or cherrybark (*Quercus pagoda* Raf.) oak.

Within each stand, 20th acre plots were used to inventory planted oak trees and 100th acre sub-plots were used for natural regeneration. All oaks within the plot were measured for diameter at breast height (DBH) and identified to species. On each plot, the two closest dominant or co-dominant trees to plot center were measured for height, height to live crown, and crown diameter in two directions. Where age data were not reliably available for a given stand, it was determined by extracting an increment core from the base of a dominant tree within the stand.

A subset of individual trees were destructively sampled for stem growth analysis and to develop allometric models of bole, branch, and leaf biomass. Trees were felled and total height, crown height, and crown diameter were measured. Trees were then sampled at ground level, 1.1 feet, 4.5 feet, and then in 3.3 foot sections thereafter. Within each section, all branches were removed and basal diameters were recorded. The branches (with leaves attached) were then bagged and weighed by section. The remaining 3.3 foot section of bole was then weighed and an approximately one inch cross section was cut off the butt of each section, placed in an air tight bag in a cooler, and then returned to the lab for processing. Discs were weighed green and then placed in a drying oven at 60°C to constant weight. Resulting moisture ratios from the wet and dry weights were applied to the green weights of stems measured in the field.

Weight

A weight equation for green tons/acre was calculated using stem weights of the felled trees that were sampled. A total of nine trees were used: one

cherrybark oak, one water oak, and seven Nuttall oaks. The equation used was as follows:

$$W = b_0 + b_1(D^2H) \qquad \text{Equation 1}$$

where W is weight in tons, D is diameter at breast height in inches, and H is total height in feet.

Statistical analyses were performed using analysis of variance (ANOVA) and simple linear regression. Regressions were used to find trends between different factors in multiple stands of different ages. Weight equations from green weights of the stem were regressed against height and diameter.

RESULTS

Stand Dynamics

For the range of stands sampled, average diameters ranged from slightly below three inches, to almost nine inches. This diameter range was within a density range, at the plot level, from approximately 150 to 500 trees per acre (TPA) (fig. 1). There was no evident relationship ($R^2 = 0.03$, $P = 0.29$) between diameter and stand density for the range of stand conditions sampled.

To investigate stocking levels for these stands, plot averages were plotted on a bottomland stocking guide (Goelz 1995). Stocking ranged from less than 50 percent to over 100 percent of "full" stocking, with only a few stands falling below the "B-line" ("understocked") or exceeding the "A-line" ("overstocked"). The variation in stocking was attributable to the wide range in basal area (BA) (from 15 ft^2/acre to over 100 ft^2/acre) which was largely a function of the relatively wide range in stand developmental conditions across the age range (fig. 2). It is notable that even the younger stands (>10 years) are mostly above the "B-line" for minimal stocking reflecting the selection of better stocked stands.

Total heights for these plots ranged from 25 to 60 feet, and were positively correlated with diameter (fig. 3). Based on the regression, a tree at either end of this height spectrum would be 2.5 and 10.2 inches DBH respectively. The corresponding live crown height for this same range would be approximately 6 to 16 feet. As expected, the live crown height increased with height-dbh growth, although the average live crown ratio remained at approximately 70 percent across the sampled size range. Both total height (TH) and height to live crown (HLC) showed substantial variance across the size range, likely reflecting differences in site condition and stand establishment history. The regression data

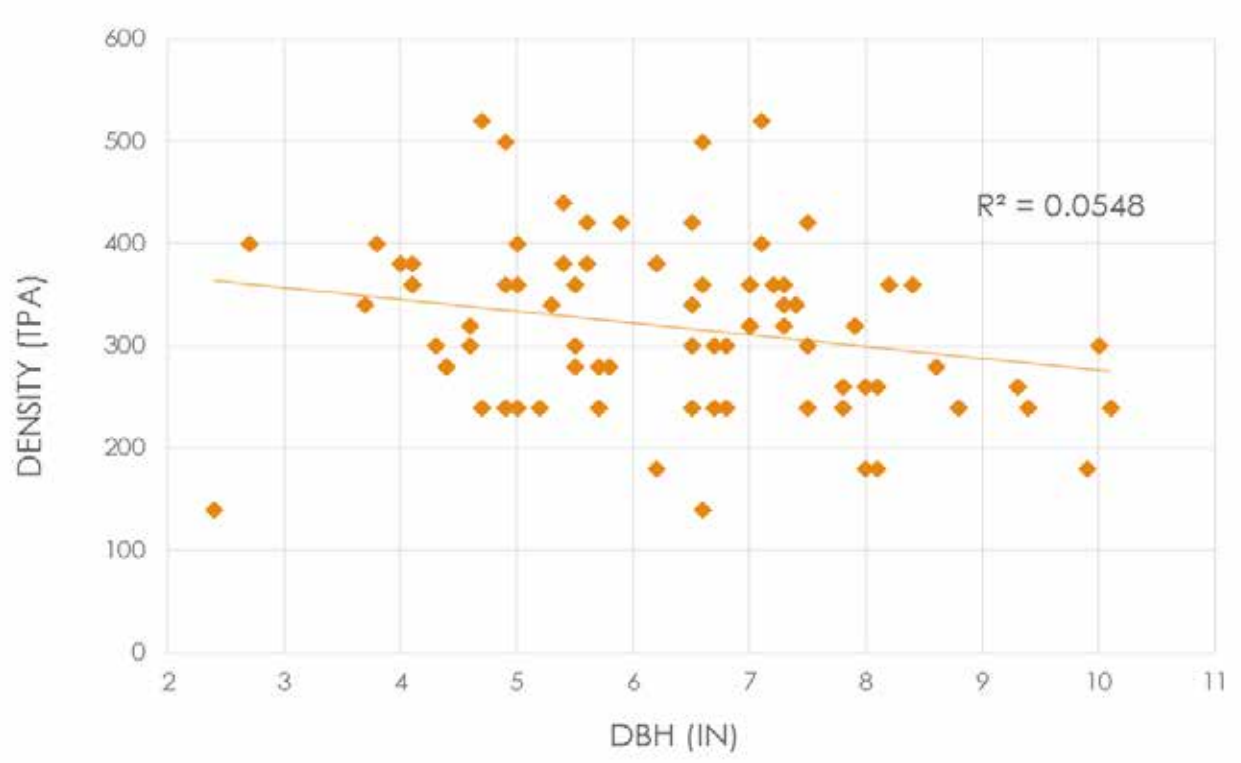


Figure 1—Density/diameter relationship for oak stands in the LMAV.

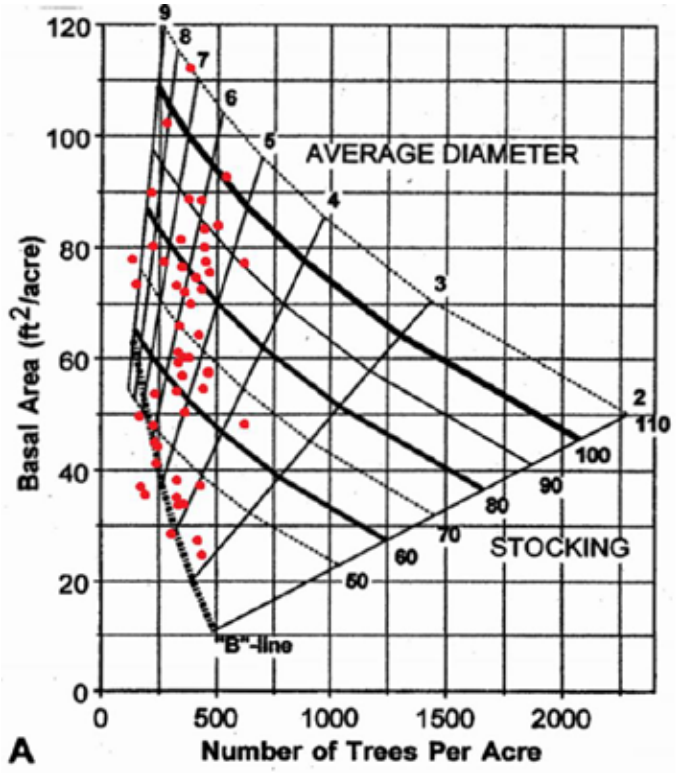


Figure 2—Bottomland stocking diagram (Goelz 1995) with plot-level averages indicated in red.

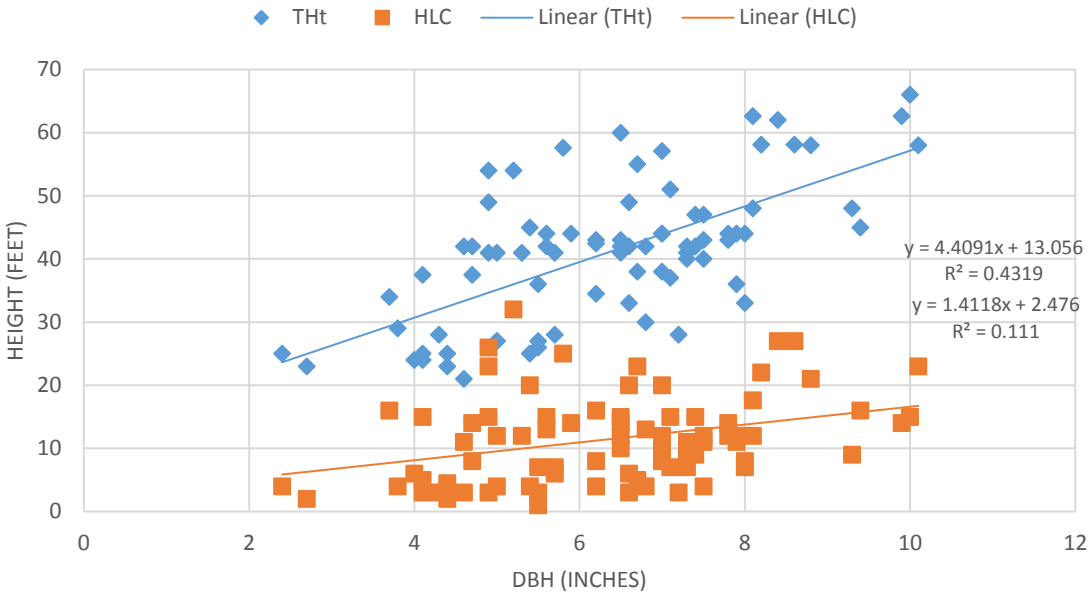


Figure 3—Total height (TH) and representative height to live crown (HLC) for oak plantations in the LMAV.

suggests that a well-pruned butt-log (16 feet) may be expected from an average density stand approaching 10 inches in diameter.

Weight

The linear regression of the nine destructively sampled trees, using equation 1 yielded the following weight equation:

$$W = - 34.671 + 1.859(D^2H) \quad \text{Equation 1.1}$$

This simple linear regression model yielded an R^2 of 0.90 (fig. 4). The sample trees ranged in weight from 0.015 to 0.17 tons. Analysis of variance for this regression model shows the relationship to be very significance ($P < 0.001$). The single tree model was then applied to plot average diameter and height and multiplied by the density of that plot and converted to estimate per acre values for volume. The plot level values fitted against height show that a stand with good survival at 25 feet would be between 8 and 12 tons per acre of green stem weight. Well stocked stands approaching 60 feet should carry up to 70 tons of green stem weight per acre (fig. 5). We emphasize again that these are values for well stocked stands between 200 and 500 trees per acre (TPA).

DISCUSSION

While the main priority of most bottomland afforestation efforts is for enhancing wildlife habitat or other ecosystem services, timber quality is an important element that will provide revenue and management options to meet these goals. As such, improving our knowledge of stand development is critical for supporting the goals of afforestation efforts.

In this study we examined conditions in young, well-stocked, oak-dominated stands representing an age range from 10-20 years old. The primary species measured were water, willow and Nuttall oak. These species represent the most frequently planted bottomland species of the 1985-1998 period by the Natural Resource Conservation Service (NRCS), WMA's, and U.S. Fish and Wildlife Service (USFWS) in the LMAV (Schoenholtz and others 2001). While some studies such as Schweitzer (1998) have indicated very poor establishment, we specifically evaluated a range of stands with high-stocking. These oak dominated stands are of particular interest to managers because of their lack of diversity (Allen 1997). Furthermore, where oak survival has been high, the relatively wide initial spacing (302 trees per acre) has likely delayed crown closure and competition among individual trees. This poses concerns for stem quality because of potentially delayed self-pruning that reduces merchantable bole length (Clatterbuck and Hodges 1988).

Most sampled stands fell in the range of 20 to 90 square feet of basal area per acre. Based on the bottomland stocking guide it is apparent that most older stands measured fell above the minimum level for "full stocking" (i.e. "B-line") (Goelz 1995). This suggests stands are generally approaching or have reached crown closure, although a wide range of stand conditions were apparent given the range of stand ages evaluated in this study. Few sampled stands approached the 100 percent line that would presumably qualify them as "overstocked" (Goelz 1995). This same stocking guide predicted that hardwood stands established at around 300 TPA would approach 100 percent stocking just before 8 inches average DBH which corresponds to the 20 plus year old stands in this study (Goelz and Meadows 1997). Stands approaching 100 percent stocking are likely experiencing reduced diameter increment and thus thinning would be desirable to maintain growth and vigor of individual trees (Goelz 1995; Goelz and Meadows 1997).

Height growth trends were similar to those reported in other studies in the southeastern U.S. evaluating oak performance during the first 20-30 years (Carlson and Goelz 1998, Krinard and Johnson 1988, Krinard and Kennedy 1987, Roth and others 1993, Stine and others 1995). Our results are comparable to those of Carlson and Goelz (1998) where both water and Nuttall oak approached 45 to 50 feet and 6 to 8 inches in diameter at age 20 for 12 x 12 foot spacing in a minor bottom in Arkansas. Our study incorporated a larger range of site indices and so had more variation in both height and diameter. It is also apparent that despite Nuttall oak's reputation for fast growth, on average, water oak out performed both Nuttall and willow oak in height and diameter. Similarly, Rousseau (2008) found that 20 year-old willow, water, and Nuttall oak approached 60, 63, and 55 feet tall, respectively, in a selection trial situated on a moderately productive site in western Kentucky. Average diameters for these same three species were 7.3, 7.9, and 6.3 inches respectively. These growth rates and height diameter relationships correspond with the upper range of performance values in our data, which is to be expected given the more controlled environments of species trials.

Average live crown ratios remained similar throughout the range of diameters sampled. A crown ratio of 70 percent or greater may be favorable to diameter growth but is less desirable for clear wood production. Stand density, particularly crown competition from adjacent trees, is one of the primary factors determining self-pruning and thus frequency of defects in trees in young hardwood stands (Sonderman 1985, Ward 1964). Density, along with age and species were the greatest

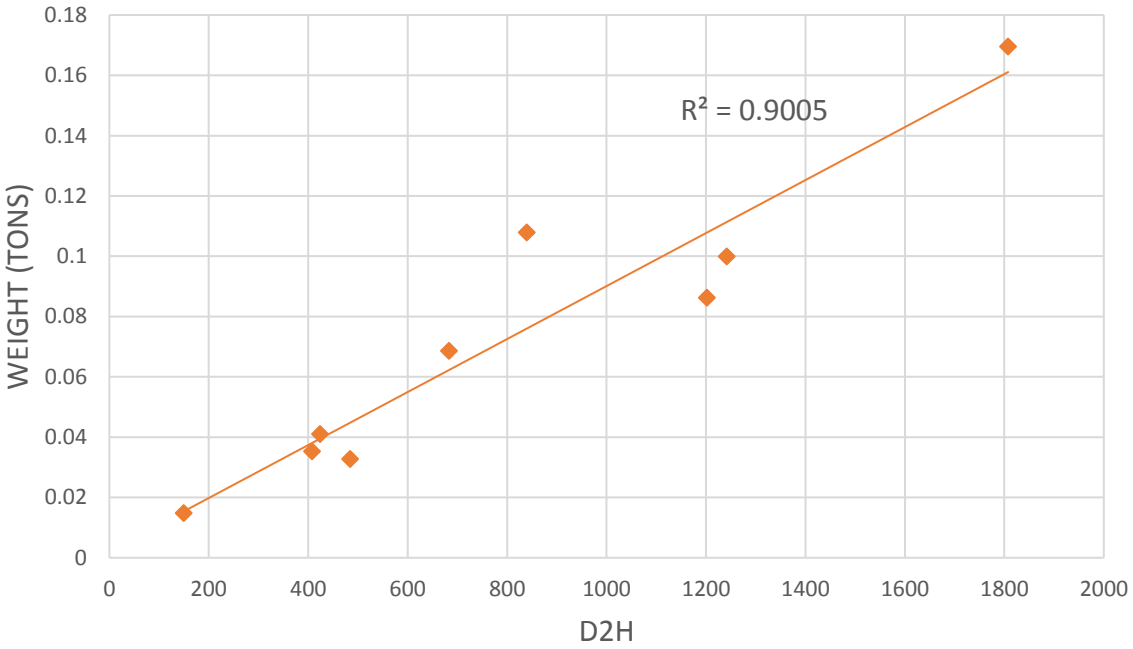


Figure 4—Weight regression model using nine felled trees.

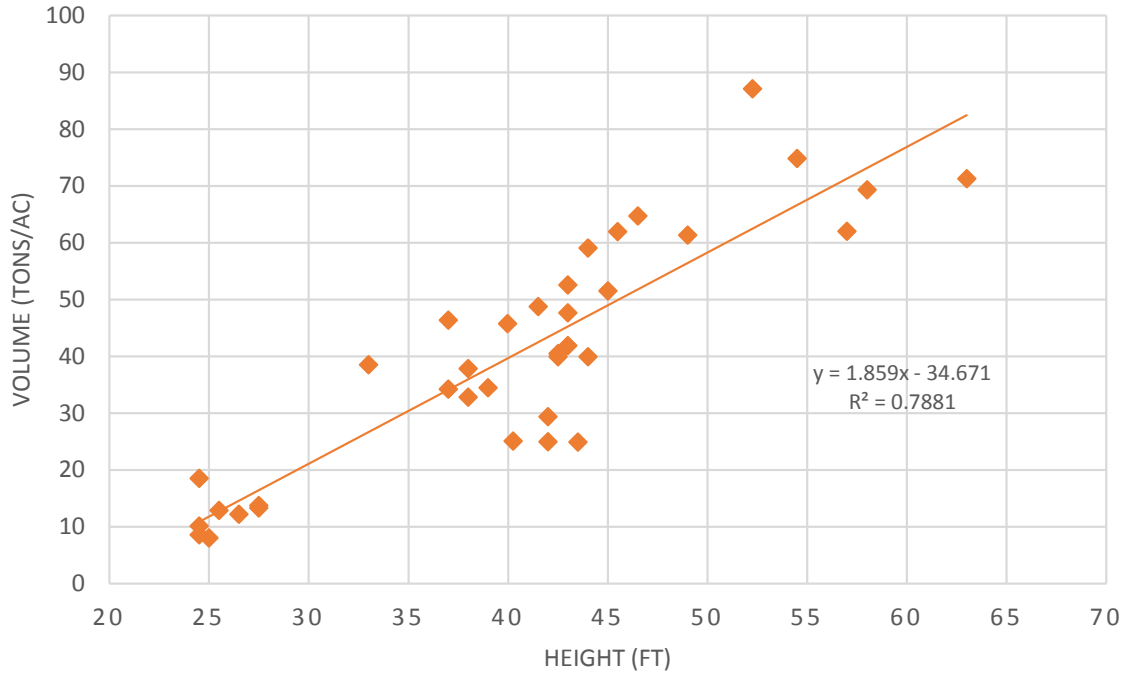


Figure 5—Weight model applied to plot-level averages(diameter and height) and plotted against height.

factors in determining natural pruning and defect, especially in the second 8 feet of the butt log. Live crown height does not exceed one log (17.3 feet) until a stand has grown to around 55 to 60 feet tall which would correspond to around 70 square feet of basal area at 300 TPA (fig. 2, 3). Given the low establishment density and lack of natural recruitment, it is not surprising that self-pruning has not progressed rapidly. However, it appears that as stands approach the upper limits of full-stocking, this process is well underway.

As stand weight approaches 70 tons per acre in the 20 year old stands, thinning is becoming economically feasible. It has been suggested that up to one third of the volume can be removed while still leaving desirable stocking for both tree and stand growth (Meadows and Goelz 2001). The critical tradeoff is providing sufficient time for stems to undergo self-pruning while limiting declines in tree vigor and diameter increment. If diameter growth slows, it will take much longer for wounds to close over prolonging the development of clear wood (Della-Bianca 1983). However, if thinning is applied too early (e.g. precommercial thinning) it may limit self-pruning and future stem quality (Heitzman and Nyland 1991). Based on our study, this tradeoff will vary between stands depending on stocking conditions and, most likely, species composition. Further study is needed to evaluate species-specific differences in these processes, particularly in older planted stands. Improving our knowledge of tree growth and crown development will be important for refining thinning approaches and their timing, particularly as we aim to provide environmental benefits (i.e. wildlife habitat and other ecosystem services) while maintaining future stand management options.

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LITERATURE CITED

Allen, J.A. 1990. Establishment of bottomland oak plantations on the Yazoo National Wildlife Refuge complex Mississippi USA. Southern Journal of Applied Forestry. 14(4): 206-210.

Allen, J.A. 1997. Reforestation of bottomland hardwoods and the issue of woody species diversity. Restoration Ecology. 5(2): 125-134.

Battaglia, L.L.; Minchin, P.R.; Pritchett, D.W. 2002. Sixteen years of old-field succession and reestablishment of a bottomland hardwood forest in the lower Mississippi Alluvial Valley. Wetlands. 22(1): 1-17.

Battaglia, L.L.; Pritchett, D.W.; Minchin, P.R. 2008. Evaluating dispersal limitation in passive bottomland forest restoration. Restoration Ecology. 16(3): 417-424.

Carlson, D.W.; Goelz, J.C.G. 1998. Hardwood plantations after 20 years on a minor stream bottom in southeast Arkansas: survival, growth, and volunteer ingrowth. In: Waldrop, T.A., ed. 1998. Proceedings of the 9th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 555-559

Clatterbuck, W.K.; Hodges, J.D. 1988. Development of cherry bark oak and sweetgum in mixed, even aged bottomland stands in central Mississippi, U.S.A. Canadian Journal of Forest Resources. 18(1): 12-18.

Heitzman, E.; Nyland, R.D. 1991. Cleaning and early crop-tree release in northern hardwood stands: a review. Northern Journal of Applied Forestry. 8(3): 111-115.

Della-Bianca, L. 1983. Effect of intensive cleaning on natural pruning of cove hardwoods in the southern Appalachians. Forest Science. 29(1): 27-32.

Gardiner, E.S.; Stanturf, J.A.; Schweitzer, C.J. 2004. An afforestation system for restoring bottomland hardwood forests: biomass accumulation of Nuttall oak seedlings interplanted beneath eastern cottonwood. Restoration Ecology. 12(4): 525-532.

Goelz, J.C.G. 1995. A stocking guide for southern bottomland hardwoods. Southern Journal of Applied Forestry. 19(3): 103-104.

Goelz, J.C.G.; Meadows, J.S. 1997. Stand density management of southern bottomland hardwoods. In: Meyer, D.A., ed. Proceedings of the 25th annual hardwood symposium: 25 years of hardwood silviculture: a look back and a look ahead. 1997 May 7-10. Cashiers, NC. Memphis, TN: National Hardwood Lumber Association: 73-82.

Krinard, R.M.; Johnson R.L. 1988. Stand parameters of a 27-year-old water oak plantation on old field loessial soils. Res. Note SO-348. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 4 p.

Krinard, R.M.; Kennedy, H.E. Jr. 1987. Fifteen year growth of six planted hardwood species in Sharkey clay soil. Res. Note SO-336. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 4 p.

Lockhart, B.R.; Keeland, B.; McCoy, J.; Dean, T.J. 2003. Comparing regeneration techniques for afforesting previously farmed bottomland hardwood sites in the lower Mississippi Alluvial Valley, USA. Forestry: An International Journal of Forest Research. 76(2): 169-180.

Meadows, J.S.; Goelz, J.C.G. 2001. Fifth-year response to thinning in a water oak plantation in north Louisiana. Southern Journal of Applied Forestry. 25(1): 31-39.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. New York: John Wiley and Sons, Inc. 520 p.

Roth, F. A.; Wheeler, G. L.; Colvin, R.J. 1993. Growth of a cherrybark oak plantation after 29 years on a Coastal Plain creek-bottom site. In: Brissette, J.C., ed. Proceedings of the 7th biennial southern silvicultural research conference. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 479-483

Rousseau, R. J. 2008. Twenty-year results from a bottomland oak species comparison trial in western Kentucky. In: Jacobs, D.F.; Michler, C.H., eds. 2008. Proceedings of the 16th central hardwood forest conference. Gen. Tech. Rep. NRS-P-24. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 223-232.

Schoenholtz, S.H.; James, J.P.; Kaminski, R.M. [and others]. 2001. Afforestation of bottomland hardwoods in the lower Mississippi Alluvial Valley: status and trends. The Society of Wetland Scientists. 21(4): 602-613.

Stanturf, J.A.; Gardiner, E.S.; Hamel, P.B. [and others]. 2000. Restoring bottomland hardwood ecosystems in the lower Mississippi Alluvial Valley. Journal of Forestry. 98(8): 10-16.

Stanturf, J.A.; Schoenholtz, S.H.; Schweitzer, C.J.; Shepard, J.P. 2001. Achieving restoration success: Myths in bottomland hardwood forests. Restoration Ecology. 9(2): 189-200.

Schweitzer, C.J.; 1998. What is restoring bottomland hardwood forests? A study from the Lower Mississippi Alluvial Valley. Transactions of the 63rd North American wildlife and natural resources conference; 1998 March 20-25; Orlando, FL. Washington, DC: Wildlife Management Institute: 147-155.

Sonderman, D.L. 1985. Stand density: a factor affecting stem quality of young hardwoods. Res. Pap. NE-561. Broomall, PA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 8 p.

Stine, M.; Chambers, J.L.; Wilson, M.; Ribbeck, K. 1995. Twenty-year survival and growth of six bottomland hardwood species. In: Boyd, M.B., ed. Proceedings of the 8th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-1. Asheville, NC. U.S. Department of Agriculture Forest Service, Southern Research Station: 500-502

Twedt, D.J. 2004. Stand development on reforested bottomlands in the Mississippi Alluvial Valley. Plant Ecology. 172: 251-263.

Twedt, D.J.; Wilson, R.R. 2002. Development of oak plantations established for wildlife. Forest Ecology and Management. 162(2-3): 287-298.

Ward, W.W. 1964. Live crown ratio and stand density in young, even-aged, red oak stands. Forest Science. 10(1): 56-65.

HERBICIDE OPTIONS FOR HARDWOOD MANAGEMENT

Andrew W. Ezell and A. Brady Self[1]

Abstract—The use of herbicides in hardwood management presents special problems in that many of the most effective herbicides are either designed to control hardwoods or the product is not labeled for such applications. Numerous studies involving herbicide application in hardwoods have been completed at Mississippi State University. This paper is a compilation of results from research projects which provides the latest information for such applications which we have to make available.

INTRODUCTION

Mississippi landowners intent on growing hardwoods are fortunate to have one of the most accommodating sets of environmental conditions in the nation. Long growing seasons, great climatic conditions, and the wide variety of appropriate species make establishment and maintenance of hardwood stands more attainable than some other regions of the country provided the appropriate silvicultural methods are used. Even a casual observer of nature notes the ease with which various woody and herbaceous species colonize most sites in the South. Consequently, land managers are faced with a multitude of problems frequently associated with unwanted vegetative competition. These problems typically involve reduced growth in established stands and often include reduced survival in planted stands. Any given tract of land may possess a vast array of native plant species. Many of these species are not problematic and do not warrant special consideration in their role associated with hardwood management. Typically, they are controlled with herbicides during the establishment period, or are deemed to have insignificant negative effects on hardwood survival and growth. However, some species offer special problems in the management of hardwoods.

The use of herbicides has proven beneficial in management efforts by lowering establishment costs, increasing growth, reducing mortality, and lessening vine induced stem deformation. This publication informs the reader of options available in the treatment of undesirable vegetation in hardwood management efforts. Adherence to labeled application rates and timing is extremely important in hardwood management due to the negative impacts many of the most effective herbicides can have if applied improperly. Although not intended to be an all-encompassing listing, product names, rates, and application timings proven to have greatest efficacy in both operational forest herbicide work and research are reviewed.

ARTIFICIAL REGENERATION

Chemical Site Preparation For Cutover Sites

Pine and hardwood silviculture differ in many aspects, but both systems benefit from proper competition control. While post-planting herbicide options differ greatly between pines and hardwoods, chemical site preparation treatments are very similar. Applications using imazapyr (e.g. Arsenal® AC, Chopper® Gen2, .etc.) and glyphosate (e.g. Accord® Concentrate, Accord® XRT II, .etc.) are the "gold standard" for use on cutover sites. There has been no noted documentation of phytotoxic symptoms on planted hardwoods due to residual soil activity of imazapyr when herbicide applications are performed at least 12 weeks prior to planting. Glyphosate, a foliar-active only herbicide, has no soil activity and has no residual effects. Due to prohibitive costs and a lack of growing season herbaceous control, very little chemical site preparation is performed in hardwood management.

Treatment applications of this nature are intended to aid in control of species that cannot be eliminated through use of a herbicide application after trees are planted.

If chemical site preparation should prove necessary, currently the standard recommendation is:

> 28-32 ounces/acre Chopper® GEN2 + 4 – 5 quarts/acre of a forestry labeled glyphosate product

Application should occur in August – October late prior to planting.

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Chemical Site Preparation For Retired Agricultural Fields

Retired agricultural areas encompass a unique set of environmental variables not present in most cutover sites. While chemical site preparation on cutover sites primarily targets woody competition, former agricultural fields often do not have a woody component. Woody species present in former fields are often beneficial to planted seedlings, with improvement to desirable stem quality outweighing possible associated negative effects. A forester, or other natural resource professional, is needed to aid in determining when non-planted woody plant density reaches a level that is detrimental to planted stems.

Site preparation will not provide control of herbaceous competition following planting unless a product with residual soil activity like Oust® XP is added to the mixture. Again, chemical site preparation should only be considered if necessary for control of existing onsite vegetation that cannot be eliminated using growing season herbaceous weed control. Unless necessitated for control of one or more of these problem species, the expense of chemical site preparation is not justified.

If necessary, chemical site preparation on retired field sites is typically applied in bands 4 to 6 feet wide. Banding herbicides in this manner reduces overall treatment cost and maintains both cover and food sources for wildlife. Current application recommendations that vary by situation are:

- 5 – 6 quarts/acre of a forestry labeled glyphosate product + an appropriate surfactant
- 14 – 16 ounces/acre Arsenal® AC + 4 – 5 quarts/acre of a forestry labeled glyphosate product + an appropriate surfactant
- 16 ounces/acre Arsenal® AC + an appropriate surfactant
- 28-32 ounces/acre Copper® GEN2 + an appropriate surfactant

Applications should be performed August – October.

Herbaceous Weed Control

Herbaceous weed control (HWC) is a treatment designed solely for control of herbaceous competition during the first growing season after planting. Research over the last 25 years regarding herbicide rates and application timing has developed prescription recommendations. Currently, the typical HWC recommendation in hardwood plantings (excepting cottonwood) is:

2 ounces Oust® XP/acre (Applications should be performed prior to bud break (February – March) (Consideration should be given to soil pH. See below)

Applied as a pre-emergent, Oust® XP (sulfometuron methyl) provides excellent control of most grass and broadleaf species when applied in bands or broadcast applications. Special consideration is warranted if treating areas with higher pH soils (applications of 1 ounce Oust® XP/acre should be used if pH ≥ 6.5) or areas that flood March - July. Other products are effective, but result in off-label use when used in hardwood management.

If HWC is necessary in areas that typically flood, Oust® XP should be avoided. Oust® XP will move with water, and may result in wasted application expense as well as off-target impact if flood waters reach application areas. Goal® 2XL (oxyfluorfen) is a good alternative in these situations. Long used in cottonwood HWC, Goal® treated hardwood seedlings have exhibited no phytotoxicity. Like Oust® XP, Goal® 2XL provides best results when used as a pre-emergent, but exhibits less residual soil activity and controls fewer species. Of particular note, Goal® can be applied post-emergent, but does not control most grasses. Standard use rate recommendations are:

- Pre-emergent = 64 ounces/acre Goal® 2XL+ an appropriate surfactant
- Post-emergent = 32 ounces/acre Goal® 2XL + an appropriate surfactant

Application timings for Goal® 2XL vary by targeted species. The herbicide label should be consulted when determining appropriate timing for treatment.

Occasionally situations arise that dictate the use of a grass-only herbicide for HWC. Two products labeled for grass control in hardwoods are Select® 2EC (clethodim) and Fusilade® DX (fluazifop-P-butyl). Both herbicides are foliar active and effective only on grass species. These two herbicides offer the best option for control of “problem” grasses such as bermudagrass or johnsongrass with Select® generally exhibiting greater efficacy in single application treatments. Nonionic surfactants should be used if utilizing either product. Spray rates and appropriate application timings vary dependent upon growth stage and target species.

Different herbicides exhibit different lengths of vegetative control when applied on targeted vegetation. None of the herbicides used in HWC will provide complete control of vegetation for an entire growing season. The intent of these applications is not to achieve complete growing season herbaceous control, but rather to provide an adequate time period for establishment of planted seedlings in a “free-to-grow” status.

NATURAL REGENERATION

Light is the one of the most important factors in establishing and recruiting natural regeneration of hardwoods. Many regeneration attempts fail when established shade-tolerant midstory stems outcompete newly germinated shade-intolerant seedlings. Even the best planned and timed overstory treatments combined with a proper harvesting schedule will fail to produce adequate natural regeneration if available light levels are insufficient for germinant survival. Midstory injection provides an effective means of control for undesirable stems competing for light in the midstory. Light is increased at ground level and young shade-intolerant stems are provided with adequate levels of light for growth.

Several compounds have been tested and are labeled for injection. Historically, much injection work was performed through girdling trees using frill cuts and "painting" a solution of Tordon® (picloram) into the cut. Tordon® products worked well in this capacity, but problems with non-target species being impacted were common. More recently, Arsenal® AC has become the standard for midstory injection work due to its effectiveness at controlling a wide variety of woody species while reducing labor requirements. Care should be taken to eliminate ground contact with imazapyr, as soil activity could result in non-target impact of hardwoods. Current midstory injection recommendations are:

> Arsenal® AC mixed at 20 percent vol/vol with water. Apply 1ml of solution per cut and use 1 cut per 3 inches of stem diameter

Efficacy of midstory injection with imazapyr varies depending on timing with early fall applications (September – October) proving most effective with slightly less control from November – February and July – August applications. Avoid injection during March – June.

While injection works typically works very well, other methods/compounds may be justified at times due to species resistance and numbers. Stand conditions may exist where the number of target small diameter stems precludes injection due to expense. Additionally, some species are naturally resistant to imazapyr (e.g. elms, locusts, eastern redbud, waxmyrtle). Control in these situations warrants use of a compound other than imazapyr. If either stand condition is encountered, and most target stems are not greater than 30 feet in height, skidder applications of glyphosate or Garlon® 3A (triclopyr - amine) may be possible. Under these conditions, standard application recommendations are:

- 4 percent vol/vol of a forestry labeled glyphosate product + an appropriate surfactant
- 4 percent vol/vol of a forestry labeled glyphosate product + 3 percent vol/vol Garlon® 3A + an appropriate surfactant (if species are present that glyphosate will not control)

Invasive/Noxious Species

Many species have extremely aggressive growth rates and may become a severe threat to young hardwoods, especially if no chemical site preparation is used on an area. If possible, managers should endeavor to control these species prior to hardwood establishment. Herbicidal control of several of these "problem" species is detailed below.

Morning-glory—Morning-glory is a small herbaceous vine that often climbs planted hardwoods, pulling seedling tops over. In agricultural fields this vine can quickly overtake a large area causing severe stem deformity and mortality of planted seedlings. In addition, seedling leaves are often shaded to a point were insufficient levels of light are available thus resulting in increased seedling mortality. Seven species of morning-glory are native to Mississippi, and Goal® 2XL will control the five most common. Pre-emergent treatment of morning-glory with Goal® is not effective, but post-emergent applications have proven to have great control. Currently the treatment recommendation for morning-glory is:

> 32 ounces/acre Goal® 2XL + an appropriate surfactant (post-emergent)

Timing of application will vary dependent upon species involved and growing conditions, but should occur when vine leaders are between one and three feet in length.

Kudzu—This introduced native of Asia has resulted in timber management problems across the state for decades. Several compounds are labeled for use in kudzu control, but Escort® XP (metsulfuron methyl) typically yields the best results. If treating areas with seedlings/saplings, Escort® XP may cause damage in hardwoods. Transline® (clopyralid) is a more appropriate compound for controlling kudzu in areas with smaller hardwoods. Efficacy from Transline® is not as good as in Escort® XP, but damage to hardwoods is avoided. A relative newcomer in kudzu control is Streamline® (aminocyclopyrachlor + metsulfuron methyl), which provides excellent control. Treatment recommendations include:

- 4 ounces/acre Escort® XP + an appropriate surfactant
- 21 ounces/acre Transline® + an appropriate surfactant

- 10 ounces/acre Streamline® + an appropriate surfactant

Application timing for all three products should be in July – October, with overall spray volumes in the 50 to 100 gallons per acre (GPA) range for adequate coverage.

Wisteria—Wisteria is a vine species that impacts hardwood seedling/saplings in a manner similar to morning-glory. The tops of smaller trees are pulled over and shaded by wisteria foliage. Additionally, this species may compete with larger trees in the same manner. Many products have been tested for control of wisteria in hardwood areas with little success for most compounds. Transline® has demonstrated good control and the treatment recommendation is:

21 ounces/acre Transline® + an appropriate surfactant

For greatest efficacy, herbicide applications should occur during the months of May – July.

Trumpet Creeper—Trumpet creeper, also known as "cow itch vine," is another vine species often having severe impacts on seedling growth and survival in regeneration efforts. Glyphosate has proven very effective in trumpet creeper control. Careshould be taken to avoid herbicide contact with hardwood foliage as glyphosate will damage or kill the trees. The standard treatment recommendation for trumpet creeper is:

- 4 quarts/acre of a forestry labeled glyphosate product + an appropriate surfactant

Herbicide applications should occur during late summer months (August – September).

Redvine—Redvine is a perennial woody vine common to bottomland sites with clay soils. The species can be difficult to control and competes with hardwood seedlings/saplings much in the same manner as morning-glory and wisteria. Heavy coverage of redvine often occurs resulting in planting failures due to intense competition for light. Dicamba HD (diglycolamine) can be used to provide good control in non-seedling areas. Chemical site preparation is the only option for redvine control on properties to be regenerated as non-target seedling damage will be prohibitive. If control of redvine is desired in older hardwood stands, recommended treatment is:

2 quarts/acre Dicamba HD® + an appropriate surfactant

Application timing should be during late summer months (August – September).

Peppervine—Peppervine is another vine species that is very problematic and very difficult to control. Little can be done to control peppervine competition after seedlings are planted or naturally regenerated. Multiple compounds have been tested with little success. The best option available for managers wishing to regenerate hardwoods in areas where peppervine occurs is to attempt control of the species in site preparation efforts.

Japanese climbing fern—Another introduced vine native to southeastern Asia, Japanese climbing fern was introduced into Florida and has spread throughout Florida and into the southern portions of Gulf Coast states. Currently, the spread of the species has reached central Mississippi. Japanese climbing fern engulfs young trees underneath a dense canopy of vegetation. Damage and mortality to tree species is similar to that of other problem species discussed above. Current application recommendations are:

- 1 ounce/acre Escort® XP + an appropriate surfactant (may damage seedlings)
- 5 percent vol/vol Clearcast® + an appropriate surfactant (will damage seedlings)

Applications should be performed from July - October.

Cogongrass—Cogongrass is native to southeastern Asia and was first reported in the United States in the early 1900s. The species outcompetes most native species resulting in their suppression or elimination from the system. Cogongrass spread is promoted through the lack of cleaning roadside/farming/logging equipment when moving between sites. Combinations of imazapyr and glyphosate have been used in control attempts, but multiple-application treatments are needed for total control. Additionally, Streamline® has proven very effective in cogongrass control. Control recommendations in hardwoods are:

- 10 ounces/acre Streamline® + an appropriate surfactant (may damage seedlings)

Applications should take place during April.

Trifoliate Orange—Trifoliate orange, sometimes called "mock orange," "lemon tree," or "Chinese bitter orange," is native to Asia and is closely related to citrus species. This species can form dense thickets capable of reducing available light to a point where native hardwood species cannot be regenerated under a canopy of trifoliate orange in the understory. Control can be difficult, but options are available in the form of basal and foliar treatments. Options for control include Garlon® 4 (triclopyr – ester), Garlon® 3A, and Perspective® (aminocyclopyrachlor + chlorsulfuron):

Basal sprays (large stems):

- 20-25 percent vol/vol Garlon® 4 in basal oil
- 10 percent vol/vol Perspective® in basal oil

Foliar spray (small stems):

- 4 percent vol/vol Garlon® 3A + an appropriate surfactant (August)

Eastern Baccharis—Eastern baccharis, also known as baccharis, saltbush, and groundsel tree, is a native invasive shrub species historically found in wetland and coastal areas. The species has become problematic throughout much of the state due to the ease with which its seeds are transported via wind. Eastern baccharis readily establishes on sites with exposed mineral soil and will completely occupy cutover areas or old fields. Dense canopies are often formed in severe infestations of the species, with other species being eliminated or suppressed for some period of time. Timing of herbicide treatment varies depending on the compound used. Control recommendations include:

- 10 ounces/acre Streamline® + an appropriate surfactant (July - September)
- 6 quarts/acre Garlon® 4 + an appropriate surfactant (November – February) (non-dormant hardwoods may be damaged or killed)

Chinese Privet—Similar to many exotic species, Chinese privet was introduced as a landscaping plant. Originally, this shrub species was used as an ornamental and to form privacy hedges. It quickly spread due to seed dispersal from birds and root sprouts, and can now be found across the state. Native plant species are quickly suppressed in the dense thickets that often form when Chinese privet grows unchecked. Varying degrees of control have been achieved using numerous herbicides including glyphosate, imazapyr, triclopyr, metsulfuron, and hexazinone. Of these, glyphosate products have produced the best results with lowest costs. The standard recommended treatment for Chinese privet in hardwood stands is:

- 5 percent vol/vol of a forestry labeled glyphosate product + an appropriate surfactant (by hand)
- 4 quarts/acre of a forestry labeled glyphosate product + an appropriate surfactant (aerial)

Hand applications can be performed year round due to the evergreen status of Chinese privet. Aerial applications should be performed after hardwood leaf drop on non-target vegetation (late winter). However, best efficacy for both treatments is achieved during the month of February in Mississippi. Optimal timing during February may be early in the month for south Mississippi and later in the month for northern portions of the state.

Chinese Tallowtree—Another non-native woody species native to Asia, Chinese tallowtree was introduced as an ornamental. Other names commonly associated with Chinese tallowtree include: Florida aspen, chicken tree, popcorn tree, and tallowtree. The species exhibits very quick growth, produces large amounts of seeds at as early as three years of age, possesses extremely high seed germination rates, and decaying leaves are toxic to other species of plants. This listing of attributes enables Chinese tallowtree to serve in an extremely damaging capacity in hardwood systems. Recommended treatments for control are:

- 64 ounces/acre Clearcast® + Methylated Seed Oil (MSO) (aerial)
- 2 percent vol/vol Clearcast® + MSO (by hand)
- 30 percent vol/vol Krenite® S + an appropriate surfactant
- 3 percent vol/vol Garlon® 4 + an appropriate surfactant

All treatments should be applied July – October and will damage hardwood seedlings.

Bamboo—Several species of bamboo have been introduced into southern ecosystems. All form extremely dense thickets where low levels of light preclude survival of native species. Most species reach heights that prevent “over-the-top” applications of herbicides by hand. Additionally, stem density and the quantity of foliage typical of escaped bamboo make obtaining adequate control through aerial applications of herbicides difficult. When control of established bamboo stands is desired, the best option is to cut/bulldoze the stand down. After stems resprout and are 1 -3 feet tall, herbicidal treatment is appropriate. Satisfactory control is obtained through the use of Velpar® (hexazinone) products. Standard treatment prescriptions for bamboo control are:

- 8 - 10 quarts/acre Velpar®L + an appropriate surfactant
- 85 -107 ounces/acre Velpar® DF + an appropriate surfactant

Switchcane—Switchcane, also known as giant cane and river cane, include three species of native bamboos. Historically, these species formed dense cane brakes where other plant species could not survive. While the presence of switchcane is somewhat reduced from pre-European levels, the species can still prove problematic in hardwood management efforts.

If troublesome quantities of are encountered, the recommended treatment is:

> 4 - 5 percent vol/vol of a forestry labeled glyphosate product + an appropriate surfactant

On occasion, larger cane brakes may need bulldozer/cutting treatment similar to that described in the bamboo section.

Saw Palmetto—Saw palmetto is a small, slow growing palm that grows on a variety of sites across the southeast portion of the nation. The species competes with other native species for moisture and nutrients and can reach densities that prohibit growth and survival of desired hardwoods. Best treatment options include:

> 2 quarts/acre Garlon® 4 + 2 ounces/acre Escort® XP + 1 – 2 percent vol/vol MSO

Due to the dense and extensive nature of saw palmetto vegetation, spray volumes should be 20 – 40 GPA. Typical control will be approximately 90 percent and repeat spot treatment should be applied after 8 to 12 months.

CONCLUSIONS

The practice of using herbicides in hardwood systems is increasing. Planning herbicide use in hardwood management can be difficult compared to that required in similar pine systems. However, most targeted species can be controlled through careful consideration of effective herbicides and appropriate application timing. Land managers should be cautious when utilizing herbicides in any forest management effort, but current herbicide options for use in hardwoods make suppressing unwanted vegetation both efficient and cost effective.

FIELD TEST OF FOLIAR-SPRAY HERBICIDES TO CONTROL MOUNTAIN LAUREL IN MATURE MIXED-OAK FORESTS IN WESTERN MARYLAND

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Abstract—Successful oak (*Quercus* spp.) regeneration requires the presence of competitive sources of oak reproduction before parent oaks are harvested. Mountain laurel (*Kalmia latifolia*) in the understory of many Appalachian forests prevents new oak seedlings from receiving adequate sunlight to survive and grow into competitive size classes. This study examined the efficacy of three herbicides (triclopyr, glyphosate, or imazapyr) applied as a foliar spray on mountain laurel in a mature mixed-oak forest. Each herbicide was applied at three rates, expressed as quarts per acre. The herbicides were mixed in water with a surfactant added to achieve application rates within U.S. Environmental Protection Agency label limits. The nine herbicide/rate combinations were applied once each in four different months (April, June, August, or October). Efficacy was quantified by the percentage of mountain laurel foliage controlled 12 months after treatment on 200 randomly assigned 0.005-acre plots. Triclopyr applied at 4.8 quarts per acre provided > 85 percent foliage control in all four months. Glyphosate applied at 4.8 quarts per acre also provided > 85 percent foliage control in April, June, and August. Imazapyr provided significantly less foliage control, and some hardwood trees were damaged adjacent to plots treated with imazapyr. Study results for all 36 herbicide/rate/month combinations are provided.

INTRODUCTION

Numerous research studies have indicated that successful oak (*Quercus* spp.) regeneration depends on the presence of competitive sources of oak reproduction when overstory harvests or other stand-replacing disturbances occur (Dey 2014, Loftis 2004). One reason why large, competitive advanced reproduction does not develop in mature mixed-oak forests is that interfering plants in the understory limit the availability of sunlight (Brose and others 2008, Rebbeck and others 2011). Examples of interfering plants include ferns, grasses, suppressed shade-tolerant trees, and various evergreen or deciduous shrubs. Fire suppression, heavy deer browsing, and repeated partial harvest practices have contributed to the development of interfering plants (Nowacki and Abrams 2008, Rooney 2009, Schuler 2004). Such interference often develops and persists over many decades. Without a natural or prescribed intervention to reduce the interference, successive acorn crops rarely become competitive advanced seedlings to sustain the oak component in the next stand (Loftis 1985, 1990).

Remedial measures for controlling interfering plants include mechanical methods, herbicides that involve either broadcast or stem-applied methods, and prescribed fire. Mechanical methods include the use of brush saws, chainsaws, girdling, disking, or simply crushing with large machinery to reduce low shade that may be suppressing desirable reproduction (Lof and others 2012, Nyland and others 2006). Hand methods can effectively target individual plants but can be relatively labor intensive and costly. None of the mechanical methods mentioned, however, eliminates sprouting from cut stumps or damaged roots. If the controlled plants sprout and become reestablished in just a few years, small advanced oak reproduction may not have enough time to grow into competitive size classes (Brose 2011, Miller and others 2014).

Herbicide treatments can also be applied to control interfering plants. Hack-and-squirt methods are effective at controlling small trees (Miller and others 2004, Schweitzer and Dey 2015). Basal bark methods are effective at controlling small woody stems, although

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the cost becomes prohibitive where prescriptions involve numerous stems over large acreages (Miller 1990, Zedaker 1986). Broadcast foliar-spray methods are often more efficient than target-specific methods for dense coverage of interference whose height does not exceed 15 to 20 feet (Horsley and Bjorkbom 1983, Nyland and others 2006). It is often necessary to apply stem injection treatments as a follow-up to foliar-spray treatments to control taller interference. For root-sprouting species, the cut-stump method can control thousands of stems with a relatively small volume of herbicide per acre (Kochenderfer and others 2013). Dense interference from grasses and ferns can be controlled with broadcast treatments of glyphosate and sulfometuron methyl (Engelman and Nyland 2006, Horsley 1991).

Prescribed fires are often recommended for reducing woody interference and preparing the seedbed for a future acorn crop in mature oak stands (Brose and others 2014). Such treatments are effective at reducing low woody interference, while taking care to avoid damage to the overstory trees. Historically, mountain laurel (*Kalmia latifolia*) was likely controlled under mature mixed-oak forests by repeated wildfires (Monk and others 1985). More recently, however, effective landscape-scale fire exclusion has promoted the development of low woody interference in the majority of mature oak stands, including vast areas of mountain laurel and other ericaceous shrubs (Brose and others 2014, Nowacki and Abrams 2008). Although prescribed fires can be applied to control mountain laurel interference in oak stands, forest managers have experienced three major drawbacks with this approach. First, fires applied in mature stands with dense mountain laurel interference tend to be relatively severe with much higher flame lengths and faster rates of spread that cause real or perceived damage to the value of merchantable overstory timber. Second, in stands where advanced oak reproduction is sparse or too small to be resistant, fire can eliminate any advanced oak reproduction that is present until the next acorn crop. Third, mountain laurel can sprout quickly after the fire treatment and regain its interference status before adequate competitive oak reproduction develops (Elliott and others 1999, Moser and others 1996).

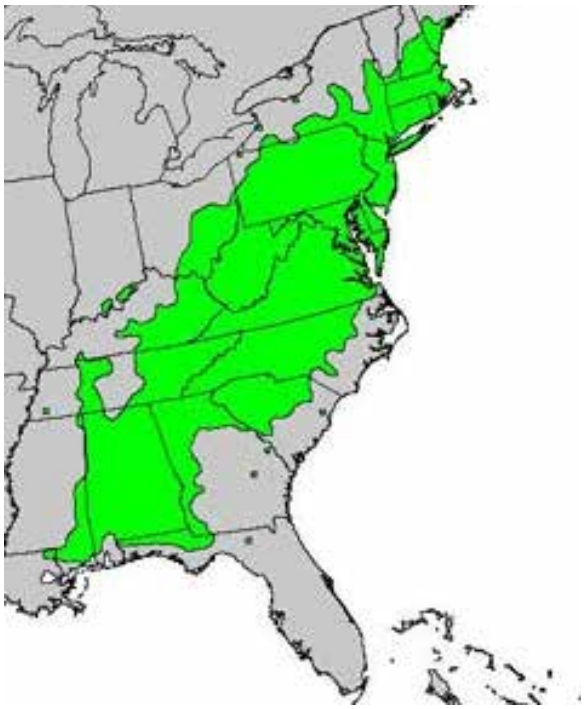


Figure 1—Native range of mountain laurel found mostly on dry, acid soils often associated with mixed-oak forests.

A broadcast foliar-spray herbicide treatment may be a better method for reducing mountain laurel interference if preservation of timber value and control of sprouting are needed throughout the oak regeneration process. The objective of this study was to quantify the effects of three foliar-spray herbicides applied at three rates per acre and at four different times of the year on the percentage of mountain laurel foliage controlled 12 months after treatment. The results would allow forest managers to formulate a mechanized broadcast prescription to successfully control mountain laurel, with some flexibility in the prescription to be consistent with other management objectives.

Background on Mountain Laurel

Mountain laurel is an evergreen shrub that develops dense thickets over many decades beneath mature forests in the Appalachian Mountains (fig. 1). If allowed to spread without the controlling influence of wildfires or other disturbances, individual plants can live as long as 40 to 60 years (McNab and Clinton 2004). It is generally associated with dry, acid soils where it can cover large areas and interfere with tree regeneration mainly by shading the forest floor and competing for moisture (Chastain and Townshend 2008, Kaeser and others 2008). This species of *Kalmia* is not considered to be allelopathic (Eppard and others 2005). It reproduces mostly by sprouting and layering, although it is capable of reproducing by seed (Malek and others 1989, Rathcke 2003).

Leaf longevity of mountain laurel can affect its sensitivity to foliar-spray herbicide treatments. Twig and foliar growth occurs in early April through June, and flowering usually occurs in May (Richardson and O'Keefe 2009). Mature leaves develop a thick, waxy cuticle that may inhibit absorption of water-soluble herbicides unless a surfactant is added to improve spread and penetration. New leaves remain on the plant for 2 to 3 years, with most dropping between spring and autumn of the second year. As a result, about half of the leaves present from spring to autumn are new

leaves, while the other half are 1 to 2 years old (Monk and others 1985).

Background on Herbicides and Surfactant

Triclopyr is a selective systemic herbicide that mimics auxin, a plant hormone involved in cell growth (Tu and others 2001). Once absorbed, triclopyr causes disorganized cell growth and vascular tissue destruction. It is effective at controlling sprouting because it stays active in target plants until they die. The ester formulation used in this study is relatively non-toxic to terrestrial animals, but can be toxic to fish and aquatic invertebrates if it reaches water systems. However, it is generally not mobile in soil, so applications should be planned to avoid movement through surface runoff or melting snow.

Glyphosate is a non-selective systemic herbicide that interferes with the formation of amino acids used to synthesize proteins, and thereby kills the plant by disrupting metabolism (Carlisle and Trevors 1988). Glyphosate has relatively low toxicity to birds and mammals (Evans and Batty 1986). The Rodeo® form of glyphosate used in this study is rated for aquatic use. It rapidly binds to soil particles, thus making it relatively immobile and incapable of absorption by non-target plant root systems (Feng and Thompson 1990). Degradation occurs mainly by microbial metabolism in the soil (Carlisle and Trevors 1988).

Imazapyr is a selective systemic herbicide that inhibits the enzyme acetolactate synthase, which catalyzes the production of amino acids essential for cell growth (Tu and others 2001). Unlike the other herbicides used in this study, imazapyr does not bind well to soil particles (McDowell and others 1997). It can leach from the roots of treated plants and damage nearby non-target plants by absorption through their root systems (DiTomaso and Kyser 2007, Kochenderfer and others 2001, Lewis and McCarthy 2008). It exhibits relatively low toxicity to birds, mammals, and fish (Patten 2003, Tatum 2004). Imazapyr is not readily metabolized in plants and provides relatively slow control of target vegetation (Tu and others 2001).

Haller and Stocker (2003) reported the relative toxicity of several adjuvants, including the limonene non-ionic surfactant Cide-Kick® that was added to the herbicides used in this study. They concluded that acutely toxic concentrations would be avoided where label recommendations were followed under normal use conditions, as was done in this study.

STUDY SITE

The study site is a 49-acre mature, mixed-oak forest located within the Potomac-Garrett State Forest in western Maryland. Basal area averaged 128 square feet per acre, relative density was 106 percent, and the quadratic mean diameter at breast height (d.b.h.) was 10 inches. The overstory trees date to the early 1920s and are composed of 35 percent chestnut oak (*Q. montana*), 20 percent northern red oak (*Q. rubra*), 16 percent red maple (*Acer rubrum*), 16 percent blackgum (*Nyssa sylvatica*), 5 percent sweet birch (*Betula lenta*), 4 percent white oak (*Q. alba*), 2 percent scarlet oak (*Q. coccinea*), and 2 percent sassafras (*Sassafras albidum*). Fire was excluded from the study site since the 1920s. The species composition was typical of the mixed mesophytic forest described by Braun (1950), with a site index of 65 for northern red oak, base age 50 years. Elevation on the study site ranges from 2,780 to 2,860 feet, with 0 to 15 percent slopes on a ridgetop or slightly west aspect. Annual precipitation averages 45 inches and is evenly distributed throughout the year. Few advanced oak seedlings were present, and oak regeneration stocking was zero as defined by Brose and others (2008). Stocking of tall woody interference and low woody interference was 70 percent and 65 percent, respectively. The low woody interference was composed mainly of mountain laurel ranging in height from 3 to 11 feet. Soils are described as Dekalb very stony sandy loam and Leetonia very stony sandy loam (Stone and Matthews 1974).

METHODS AND DESIGN

The treatments consisted of three herbicides, applied at three rates per acre, and applied once each in four different months. The herbicides tested in this study were triclopyr as Garlon® 4 Ultra, glyphosate as Rodeo®, and imazapyr as Chopper® Gen2™. Cide-Kick® surfactant was added to each herbicide to improve spread and penetration. The herbicide solutions, including appropriate volumes of undiluted herbicide, water, and surfactant, were mixed under controlled conditions off-site (table 1). The application rates were 1.6, 3.2, and 4.8 quarts per acre for triclopyr and glyphosate, and 0.8, 1.6, and 2.4 quarts per acre for imazapyr. Product labels approved by the U.S. Environmental Protection Agency specify a maximum of 8 quarts per acre for triclopyr and glyphosate and 3 quarts per acre for imazapyr. Previous guidelines for controlling mountain laurel with foliar-spray treatments were considered when setting the application rates in this study (Jackson and Finley 2005, Kochenderfer and others 2012). Finally, the treatments were applied in April, June, August, or October 2013 to determine if efficacy varied by month of application.

The study was based on a completely randomized design. Two hundred permanent circular plots were established where a somewhat ubiquitous understory cover of mountain laurel was present.

Table 1—Herbicide solution components and total solution volume applied to each 0.005-acre plot and the equivalent herbicide application rate per acre for each treatment

Herbicide name & concentration	Herbicide volume	Water volume	Surfactant volume[1]	Total solution volume	Application rate
	---------- milliliters per plot ----------				quarts per acre
1% triclopyr[2]	7.6	745.6	3.8	757	1.6
2% triclopyr	15.2	738.0	3.8	757	3.2
3% triclopyr	22.8	730.4	3.8	757	4.8
1% glyphosate[3]	7.6	745.6	3.8	757	1.6
2% glyphosate	15.2	738.0	3.8	757	3.2
3% glyphosate	22.8	730.4	3.8	757	4.8
0.5% imazapyr[4]	3.8	749.4	3.8	757	0.8
1.0% imazapyr	7.6	745.6	3.8	757	1.6
1.5% imazapyr	11.4	741.8	3.8	757	2.4
Control	0	0	0	0	0

[1]Non-ionic surfactant as Cide-Kick® (100% active ingredient [a.i.]); [2]triclopyr as Garlon® 4 Ultra (60.5% a.i.); [3]glyphosate as Rodeo® (53.8% a.i.); [4]imazapyr as Chopper® Gen2™ (26.7% a.i.).

Each plot was 8.3 feet in radius, thus covering 0.005 acre. Plots were separated by untreated buffer zones of at least 25 feet to isolate the effect of the herbicide treatments between plots. The center of each plot was marked with a permanent rod, and a nearby witness tree was painted with a colored ring to facilitate locating the plots. Five plots were randomly assigned to each of the 36 treatments (180 plots) and untreated controls (20 plots).

Before treatment, the perimeter of each plot was marked with colored flagging at several points on the ground to indicate the correct radius from the center point. The initial percent cover of mountain laurel foliage within each plot was determined to the nearest 5 percent and recorded. The volume of herbicide solution for each treatment was pre-measured and put in separate bottles such that each plot received an equal amount of herbicide solution. The herbicide treatments were applied with a hand-held garden sprayer, the entire content of each pre-measured bottle was applied to a plot, and the foliage within a plot was completely wetted. Care was taken to clean and rinse each garden sprayer before and after each treatment. Treatments were applied on days when the weather was dry, and no rain was expected for at least 48 hours after treatment. The difference between the initial cover of foliage and the cover of foliage 12 months later was used to quantify the efficacy of the treatments.

ANALYSES

Two analyses were conducted. The first analysis was a completely randomized design to compare the percentage of foliage controlled on the treated plots to that observed on the control plots.
The second analysis focused on a completely randomized factorial (3×3×4) design to examine the effect of the herbicide (factor 1), the application rate per acre (factor 2), and the month of application (factor 3) on the percentage of foliage controlled on the treated plots, excluding the control plots. The resulting fixed effect model has the form:

$$Y_{ijk} = \mu + \alpha_i + \beta_j + \theta_k + (\alpha\beta)_{ij} + (\alpha\theta)_{ik} + (\beta\theta)_{jk} + (\alpha\beta\theta)_{ijk} + \varepsilon_{ijk}$$

where,

Y = the percentage of mountain laurel foliage controlled,

μ = the overall mean,

α = the effect of the herbicide,

β = the effect of the application rate per acre,

θ = the effect of month of application,

ε = the random error.

Herbicide, application rate per acre, and month of application were fixed effects; the remaining terms represent the interaction of factors in the full model. The generalized linear mixed model routine in SAS was used for all statistical analyses (SAS Institute Inc. 2011). Initially, percent cover of mountain laurel in the pre-treatment inventory of plots was evaluated as a possible covariate in the model, but was not statistically significant. A pseudo-likelihood estimation technique via PROC GLIMMIX was used to model the efficacy of the treatments. A beta distribution and a logit link function were used because the response variable was a percentage with many small and large values. The Tukey-Kramer least squares mean separation test was used for all multiple comparisons. The differences between treatment means were considered to be statistically significant when $p < 0.05$. For each analysis, the residuals were tested for normality by the Shapiro-Wilk test and for homogeneity of variance by the Levene test.

RESULTS

In the first analysis, all treatments significantly reduced the cover of mountain laurel foliage compared to untreated controls ($p < 0.01$) (fig. 2). In the second analysis, foliage control differed among herbicides ($p < 0.01$), application rates ($p < 0.01$), and month of application ($p < 0.01$). The interactions of factors in the full model were statistically significant as well ($p < 0.01$), thus indicating a dose response for each herbicide that varied by month (table 2). In general, efficacy increased at higher application rates of each herbicide tested.

Mountain laurel was most sensitive to herbicide treatments in the June applications. New leaves emerge in May and June and have less waxy coating than mature leaves present at other times of the year. Foliage control was > 90 percent for both triclopyr and glyphosate applied at 1.6, 3.2, and 4.8 quarts per acre in June (table 2). With imazapyr, foliage control was > 55 percent at 1.6 and 2.4 quarts per acre in June, but was < 42 percent for all other treatments (table 2).

The October treatments of all herbicides and application rates exhibited the lowest foliage control compared to other months, but a dose response was still evident. For example, foliage control increased from 25 to 86 percent as the triclopyr application rate increased from 1.6 to 4.8 quarts per acre in October (table 2). Similarly, foliage control increased from 13 to 29 percent for glyphosate and from 10 to 22 percent for imazapyr as the application rates increased from low to high.

Among the three herbicides tested, triclopyr exhibited the most consistent control of mountain laurel foliage, with > 75 percent control at 3.2 quarts per acre and > 85 percent control at 4.8 quarts per acre in each of the months tested (table 2). Glyphosate provided similar foliage control compared to triclopyr in June and August, but significantly less control in April and October. Imazapyr provided > 55 percent control at 1.6

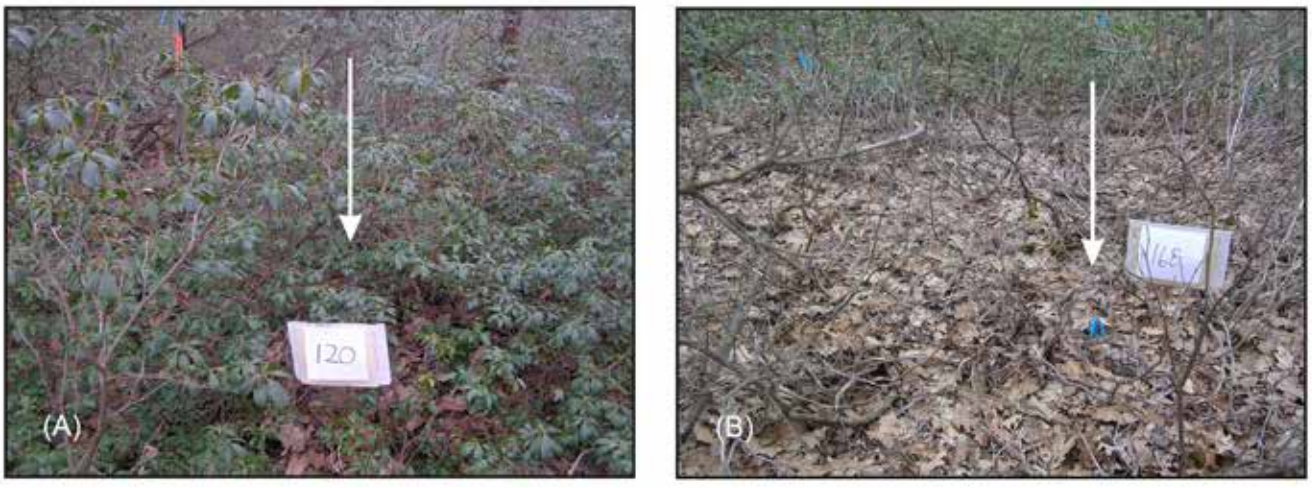


Figure 2—Mountain laurel foliage cover 12 months after (A) no treatment and (B) treatment with 4.8 quarts per acre Garlon® 4 Ultra. White arrows indicate plot centers. (photos by Gary W. Miller)

Table 2—Mean percentage of mountain laurel foliage controlled 12 months after foliar-spray treatments, n=5 observations for each entry, one standard error in parentheses, followed by results of multiple comparisons[1]

Herbicide	Application rate	Month of application 2013			
		April	June	August	October
	quarts per acre	---------- percentage of foliage controlled ----------			
triclopyr[2]	1.6	61.5 (9.8)e	94.8 (0.1)b	81.1 (5.0)d	24.6 (5.8)i
	3.2	87.0 (6.8)cd	94.7 (0.1)b	76.7 (4.5)d	75.8 (1.9)d
	4.8	99.0 (1.0)a	91.8 (1.9)c	92.1 (1.3)c	86.3 (6.0)cd
glyphosate[3]	1.6	18.0 (3.4)gh	92.6 (2.1)bc	56.1 (6.2)e	12.6 (1.2)h
	3.2	73.3 (9.5)d	97.9 (1.3)a	92.0 (1.3)c	27.6 (6.8)fg
	4.8	89.8 (4.9)cd	97.8 (1.4)a	91.8 (1.6)c	29.2 (5.9)fg
imazapyr[4]	0.8	40.0 (9.9)ef	17.2 (2.6)gh	17.7 (5.1)gh	10.4 (0.3)j
	1.6	41.5 (7.8)ef	57.1 (13.4)e	36.1 (8.0)fg	14.7 (2.2)hi
	2.4	41.1 (6.8)ef	62.7 (14.9)de	36.5 (3.9)f	21.6 (1.7)g
Control		0	0	0	0

[1]Entries followed by the same letter are not significantly different at α = 0.05; [2]triclopyr as Garlon® 4 Ultra (60.5% a.i.); [3]glyphosate as Rodeo® (53.8% a.i.); [4]imazapyr as Chopper® Gen2™ (26.7% a.i.).

quarts per acre and > 60 percent control at 2.4 quarts per acre in June, but significantly less control in August and October.

DISCUSSION

Each of the treatments tested in this study offers at least one superior attribute, depending on management objectives. For example, imazapyr provided less foliage control than triclopyr and glyphosate, but small conifer seedlings are not sensitive to imazapyr at the application rates tested. The forest manager might choose to accept less control of mountain laurel foliage in cases where less damage to conifer seedlings is an acceptable tradeoff. Similarly, glyphosate is the least expensive herbicide of the three tested in this study, and it is also effective in other foliar-spray, injection, and cut-stump treatments. The forest manager might choose to apply glyphosate to control mountain laurel because it is versatile, and any remaining herbicide can be used for many other herbicide application prescriptions. Triclopyr provided superior foliage control in each month tested, and it is the least mobile of the three herbicides tested, both within the vascular system of the target plant and within the soil and environment. However, Garlon® 4 Ultra is relatively volatile at temperatures > 80 degrees Fahrenheit, thus increasing the risk of damage to non-target plants on hot days. Triclopyr and imazapyr also provide greater control of post-treatment sprouting of the target plant compared to glyphosate (Kochenderfer and others 2012).

Regarding timing, April treatments are superior because most other desirable vegetation is still dormant in early spring. Only the mountain laurel foliage and other evergreen plants are likely to be damaged by foliar-spray treatments applied in April. October is another time when many desirable plants are dormant, but efficacy declined significantly in autumn when mountain laurel growth is waning (table 2). Although mountain laurel was most sensitive to the herbicide treatments applied in June, some management objectives may be inconsistent with late spring-early summer operations. Many species of songbirds are actively nesting and fledging young in mountain laurel at that time of year. Forest managers might avoid June treatments to minimize disturbance to songbird breeding activity. Desirable tree seedlings and most herbaceous species are actively growing by mid-June, so herbicide treatments at that time may result in unacceptable collateral damage to non-target species

The percentage of foliage control is expected to increase beyond 12 months for each of the herbicides tested. Efficacy exceeded 85 percent for most of the medium and high application rates for triclopyr and glyphosate after 12 months. Foliage control and plant mortality will likely approach 100 percent in the second

growing season after treatment. Imazapyr in particular is documented to take longer than the other herbicides to control target vegetation, due in part to its slower metabolism.

For each treatment, foliage control did not extend beyond the perimeter of the plots. Foliage outside the plots was not affected, even where control reached 100 percent. The study site had an obvious "spotted" appearance where mountain laurel foliage was missing in somewhat perfect circles 12 months after treatment. Although mountain laurel reproduces in part by layering, this observation indicates that the herbicide treatments had limited mobility through root networks, and that herbicide contact with foliage was the primary mechanism of absorption. When planning mechanized broadcast treatments, it is important to obtain adequate herbicide contact with foliage and sufficient absorption of active ingredient to achieve the desired degree of control.

A damage assessment was conducted in mid-September 2014, about 11 to 17 months after the treatments, to determine if non-target trees on or adjacent to the treated plots had suffered any herbicide damage. No damage was associated with plots treated with triclopyr or glyphosate. All plots treated with imazapyr in April, June, or August had damaged trees on or within 13 feet of the plot. Only two plots treated with imazapyr in October had nearby damaged trees 11 months after treatment, but additional damage may become apparent in later assessments. Damaged trees were either completely defoliated or exhibited deformed or chlorotic leaves. Species damaged around plots treated with imazapyr included red maple, blackgum, sassafras, and sweet birch. One 15-inch d.b.h. chestnut oak had deformed leaves in its upper crown. Damaged trees ranged in size from 3 to 16 inches d.b.h., and it was assumed that imazapyr was absorbed through root uptake within the treated plots. Assessments will be continued for several years to determine the full extent of damage to non-target plants.

The results of this study indicate the need for further research on foliar-spray herbicide control of mountain laurel. In this study the herbicide treatments were applied by hand, assuring that foliage was thoroughly wetted and the volume of herbicide delivered per unit area was consistent. Further testing is needed at operational scales using mechanized broadcast spray equipment to define how the results may vary under real-world conditions. In addition, the treated plots will be monitored for several years to quantify any increases in efficacy that may occur in subsequent growing seasons, the degree to which mountain laurel plants become reestablished after the herbicide treatments, and whether collateral damage or additional mountain laurel mortality occurs outside the treated plots. Finally, additional testing is needed on other sites, where soil and other ecological conditions may differ, to see if the results reported here can be expected over a larger geographic region.

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LITERATURE CITED

Braun, E.L. 1950. Deciduous forests of eastern North America. Philadelphia: Blakiston. 596 p.

Brose, P.H. 2011. A comparison of the effects of different shelterwood harvest methods on the survival and growth of acorn-origin oak seedlings. Canadian Journal of Forest Research. 41: 2359-2374.

Brose, P.H.; Dey, D.C.; Waldrop, T.A. 2014. The fire-oak literature of eastern North America: synthesis and guidelines. Gen. Tech. Rep. NRS-135. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 98 p.

Brose, P.H.; Gottschalk, K.W.; Horsley, S.B. [and others]. 2008. Prescribing regeneration treatments for mixed-oak forests in the Mid-Atlantic region. Gen. Tech. Rep. NRS-33. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 100 p.

Carlisle, S.M.; Trevors, J.T. 1988. Glyphosate in the environment. Water, Air, and Soil Pollution. 39: 409-420.

Chastain, R.A.; Townsend, P.A. 2008. Role of evergreen understory shrub layer in the forests of the central Appalachian Highlands. Journal of the Torrey Botanical Society. 135(2): 208-223.

Dey, D.C. 2014. Sustaining oak forests in eastern North America: regeneration and recruitment, the pillars of sustainability. Forest Science. 60(5): 926-942.

DiTomaso, J.M.; Kyser, G.B. 2007. Control of *Ailanthus altissima* using stem herbicide application techniques. Arboriculture and Urban Forestry. 33: 55-63.

Elliott, K.J.; Hendrick, R.L.; Major, A.E. [and others]. 1999. Vegetation dynamics after a prescribed fire in the southern Appalachians. Forest Ecology and Management. 114: 199-213.

Engelman, H.M.; Nyland, R.D. 2006. Interference to hardwood regeneration in northeastern North America: assessing and countering ferns in northern hardwood forests. Northern Journal of Applied Forestry. 23(3): 166-175.

Eppard, H.R.; Horton, J.L.; Nilsen, E.T. [and others]. 2005. Investigating the allelopathic potential of *Kalmia latifolia* L. (Ericaceae). Southeastern Naturalist. 4(3): 383-392.

Evans, D.D.; Batty, M.J. 1986. Effects of high dietary concentrations of glyphosate on a species of bird, marsupial and rodent indigenous to Australia. Environmental Toxicology and Chemistry. 5: 399-401.

Feng, J.C.; Thompson, D.G. 1990. Fate of glyphosate in a Canadian forest watershed: 2. Persistence in foliage and soils. Journal of Agricultural Food Chemistry. 38: 1118-1125.

Haller, W.T.; Stocker, R.K. 2003. Toxicity of 19 adjuvants to juvenile *Lepomis macrochirus* (bluegill sunfish). Environmental Toxicology and Chemistry. 22(3): 615-619.

Horsley, S.B. 1991. Using Roundup and Oust to control interfering understories in Allegheny hardwood stands. In*:* McCormick, L.H.; Gottschalk, K.W., eds. Proceedings of the 8th central hardwood forest conference. Gen. Tech. Rep. NE-148. Radnor, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station: 281-290.

Horsley, S.B.; Bjorkbom, J.C. 1983. Herbicide treatment of striped maple and beech in Allegheny hardwood stands. Forest Science. 29(1): 103-112.

Jackson, D.R.; Finley, J.C. 2005. Herbicides and forest vegetation management. University Park, PA: Pennsylvania State University, College of Agricultural Sciences Extension Publication. 31 p.

Kaeser, M.J.; Gould, P.J.; McDill, M.E.; Steiner, K.C.; Finley, J.C. 2008. Classifying patterns of understory vegetation in mixed-oak forests in two ecoregions of Pennsylvania. Northern Journal of Applied Forestry. 25(1): 38-44.

Kochenderfer, J.D.; Kochenderfer, J.N.; Miller, G.W. 2012. Manual herbicide application methods for managing vegetation in Appalachian hardwood forests. Gen. Tech. Rep. NRS-96. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 59 p.

Kochenderfer, J.D.; Kochenderfer, J.N.; Miller, G.W. 2013. Extending the time interval for applying herbicide in cut-stump treatments on American beech. Northern Journal of Applied Forestry. 30(3): 118-124.

Kochenderfer, J.D., Zedaker, S.M.; Johnson, J.E.; Smith, D.W; Miller, G.W. 2001. Herbicide hardwood crop tree release in central West Virginia. Northern Journal of Applied Forestry. 18: 46-53.

Lewis, K.; McCarthy, B. 2008. Nontarget tree mortality after tree-of-heaven (*Ailanthus altissima*) injection with imazapyr. Northern Journal of Applied Forestry. 25(2): 66-72.

Lof, M; Dey, D.C.; Navarro, R.M.; Jacobs, D.F. 2012. Mechanical site preparation for forest restoration. New Forests. 43(5-6): 825-848.

Loftis, D.L. 1985. Preharvest herbicide treatment improves regeneration in southern Appalachian hardwoods. Southern Journal of Applied Forestry. 9(3): 177-180.

Loftis, D.L. 1990. A shelterwood method for regenerating red oak in the Southern Appalachians. Forest Science. 36(4): 917-929.

Loftis, D.L. 2004. Upland oak regeneration and management. In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 163-167.

Malek, A.A.; Blazich, F.A.; Warren, S.L.; Shelton, J.E. 1989. Influence of light and temperature on seed germination of mountain laurel. Journal of Environmental Horticulture. 7(4): 161-162.

McDowell, R.W.; Condron, L.M.; Main, B.E.; Dastgheib, F. 1997. Dissipation of imazapyr, flumetsulam, thifensulfuron in soil. Weed Research. 37: 381-389.

McNab, W.H.; Clinton, B.D. 2004. *Kalmia latifolia* L. In: Francis, J.K., ed. Wildland shrubs of the United States and its Territories: thamnic descriptions: volume 1. Gen. Tech. Rep. IITF-26. San Juan, PR: U.S. Department of Agriculture Forest Service, International Institute of Tropical Forestry; Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 407-410.

Miller, G.W.; Kochenderfer, J.N.; Gottschalk, K.W. 2004. Effect of pre-harvest shade control and fencing on northern red oak seedling development in the central Appalachians. In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 182-189.

Miller, G.W.; Kochenderfer, J.N.; Kochenderfer, J.D.; Gottschalk, K.W. 2014. Ten-year response of competing vegetation after oak shelterwood treatments in West Virginia. In: Groninger, J.W.; Holzmueller, E.J.; Nielsen, C.K.; Dey, D.C., eds. Proceedings of the 19th central hardwood forest conference. Gen. Tech. Rep. NRS-P-142. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 156-171.

Miller, J.H. 1990. Streamline basal application of herbicide for small-stem hardwood control. Southern Journal of Applied Forestry. 14(4): 161-165.

Monk, C.D.; McGinty, D.T.; Day, F.P. 1985. The ecological importance of *Kalmia latifolia* and *Rhododendron maximum* in the deciduous forest of the southern Appalachians. Bulletin of the Torrey Botanical Club. 112(2): 187-193.

Moser, W.K.; Ducey, M.J.; Ashton, P.M.S. 1996. Effects of fire intensity on competitive dynamics between red and black oaks and mountain laurel. Northern Journal of Applied Forestry. 13: 119-123.

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the eastern United States. BioScience. 58(2): 123-138.

Nyland, R.D.; Bashant, A.L; Bohn, K.K.; Verostek, J.M. 2006. Interference to hardwood regeneration in northeastern North America: controlling effects of American beech, striped maple, and hobblebush. Southern Journal of Applied Forestry. 23: 122-132.

Patten, K. 2003. Persistence and non-target impact of imazapyr associated with smooth cordgrass control in an estuary. Journal of Aquatic Plant Management. 41: 1-6.

Rathcke, B.J. 2003. Floral longevity and reproductive assurance: seasonal patterns and an experimental test with *Kalmia latifolia* (Ericacaceae). American Journal of Botany. 90(9): 1328-1332.

Rebbeck, J.; Gottschalk, K.; Scherzer, A. 2011. Do chestnut, northern red, and white oak germinant seedlings respond similarly to light treatments? growth and biomass. Canadian Journal of Forest Research. 41: 2219-2230.

Richardson, A.D.; O'Keefe, J. 2009. Phenological differences between understory and overstory: a case study using the long-term Harvard Forest records. In: Noormets, A., ed. Phenology of ecosystem processes. New York: Springer: 87-117.

Rooney, T.P. 2009. High white-tailed deer densities benefit graminoids and contribute to biotic homogenization of forest ground-layer vegetation. Plant Ecology. 202(1): 103-111.

SAS Institute Inc. 2011. SAS/STAT® 9.3 User's Guide. Cary, NC: SAS Institute Inc.

Schuler, T.M. 2004. Fifty years of partial harvesting in a mixed mesophytic forest: composition and productivity. Canadian Journal of Forest Research. 34(5): 985-997.

Schweitzer, C.J.; Dey, D.C. 2015. The conundrum of creating understory light conditions conducive to promoting oak reproduction: midstory herbicide and prescribed fire treatments. In: Holley, A.G.; Conner, K.F.; Haywood, J.D., eds. Proceedings of the 17th biennial southern silvicultural research conference. e–Gen. Tech. Rep. SRS-203. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 45-56.

Stone, K.M.; Matthews, E.D. 1974. Soil survey of Garrett County, Maryland. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 83 p.

Tatum, V.L. 2004. Toxicity, transport, and fate of forest herbicides. Wildlife Society Bulletin. 32(4): 1042-1048.

Tu, M.; Hurd, C.; Randall, J.M. 2001.Weed control methods handbook: tools and techniques for use in natural areas. The Nature Conservancy. http://www.invasive.org/gist/handbook.html. [Date accessed: March 25, 2015].

Zedaker, S.M. 1986. Herbicides and application techniques for managing immature hardwoods. In: Smith, H.C.; Eye, M.C., eds. Proceedings, Guidelines for managing immature Appalachian hardwood stands. SAF Publ. 86-02. Bethesda, MD: Society of American Foresters: 240-250.

DEVELOPMENT OF AN UPLAND HARDWOOD DEMONSTRATION FOREST ON THE MARY OLIVE THOMAS DEMONSTRATION FOREST.

Seth D. Hunt, John S. Kush, Rebecca J. Barlow[1]

Abstract—Landowners have experienced a dizzying array of timber prices over the past several years. At one time, hardwood pulpwood brought very little per ton and today it brings as much or more than pine pulpwood. In some markets in the Southeast today, oak sawtimber is bringing more than pine poles. Many landowners, who previously said they wanted their hardwood stands left alone because of the perceived wildlife value, are expressing an interest in upland hardwood management and regenerating their stands. Very few demonstrations are available to show landowners how they might work towards this goal. An effort is underway to establish an upland hardwood management demonstration on the Mary Olive Thomas Demonstration Forest (MOT) near Auburn, AL. The MOT has been managed by Auburn University's School of Forestry & Wildlife Sciences since it was given as a gift to the School in 1984. As part of the effort, a 9.9-acre upland hardwood forest dominated by oak and yellow-poplar will be used. All trees greater than 4.5 inches DBH have been stem-mapped; trees less than that have been sub-sampled and data have been collected on the litter layer/fuel load. Overstory trees have been cored to determine ring count. A preliminary examination of the data indicates there are few species in the midstory or understory that are of commercial value. This presentation will report on the initial stand conditions and discuss the plans for hardwood management.

INTRODUCTION

In today's markets in Alabama, hardwood pulpwood is bringing as much money as pine pulpwood and both oak and mixed sawtimber are going for more than pine sawtimber. This price phenomena is a marked change from historical stumpage trends. Many landowners have refrained from hardwood management for reasons ranging from unfavorable markets to the perception that their hardwood stands should be left untouched for a perceived wildlife value. While this change in markets has brought new opportunities to landowners with hardwood forests, it begs the question of whether landowners are thinking about the regeneration process of these hardwood stands, particularly oak stands.

Successful oak regeneration and management has two main requirements: the presence of competitive oak regeneration and timely release of this oak regeneration (Loftis 2004). In xeric forests, regeneration of oak is dependent on the accumulation of advance reproduction and the creation and maintenance of the conditions that favor such accumulation (Smith 1993). An issue is also falls the poor retention and growth of oak seedlings already present (Lorimer and others 1994). Many landowners are not aware of the need for proper conditions to facilitate oak regeneration through natural means. A demonstration forest could help provide an example to get landowners to think about hardwood management and the subsequent regeneration process.

Currently, the Mary Olive Thomas Demonstration Forest (MOT) is lacking an upland hardwood management demonstration forest. The MOT consists of a 400 acre +/- tract of land five miles outside of the Auburn University main campus. It was given in 1983 to the Alabama Cooperative Extension System for the purpose of demonstration of forest management. Management was delegated to the School of Forestry and Wildlife Sciences. As such, the purpose of this forest is to demonstrate the possibilities of upland hardwood management to landowners as well as to produce opportunities for hardwood management education within the school. The management goals of this forest are to provide a demonstration to help landowners begin thinking about the management and regeneration process for hardwood forests.

METHODS

Sites

The proposed stand is 9.9 acres +/-. Pacolet sandy loam covers most of the stand while Toccoa sandy loam is found along the southern boundary. Topography on the stand is defined by the ridge running through the middle of the stand. Elevation of the stand ranges from 600 feet to 650 feet. The stand was damaged in 1995 by

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Hurricane Opal though the total impact is unknown. No harvest has been conducted since ownership changed hands in 1983 other than a firewood salvage after Hurricane Opal.

Procedures

All trees greater than 4.5 inches DBH were stem-mapped. For each tree 4.5 inches or greater, species and diameter at breast height to the nearest tenth of an inch were recorded. All stems under 4.5 inches were systematical sampled using 10 1/10th acre plots. On each 1/10th acre plot species abundance was tallied. The leaf litter was randomly sampled using 10 1 meter squared plots. Leaf litter depth was measured to the nearest tenth of an inch to bare soil. Litter was then oven dried and weighed. A dominant or co-dominant oak in each 2 inch DBH class was cored to determine ring count at breast height.

RESULTS AND DISCUSSION

The stand averaged 138 trees and 96 square feet of basal area per acre. Basal area per species group were 22.3, 17.5, 17.3, 21.2 square feet for white oaks (*Quercus spp.)*, red oaks (*Quercus spp.)*, yellow-poplars (*Liriodendron tulipifera*), and hickories (*Carya spp.*), respectively. Notable species found in the overstory consist of white oak (*Quercus alba*), northern red oak (*Quercus rubra)*, yellow-poplar, pignut hickory (*Carya glabra*) and American beech (*Fagus grandifolia)*. The midstory is comprised mostly of American beech of both small and large diameters. American beech accounted for 8 square feet of basal area per acre. The understory is relatively open, lacking much in the way of advance regeneration of desirable species and has a developed leaf litter layer. The litter layer averaged around 2.9 +/- .8 inches with an average dry weight of 5.1 +/- .9 tons per acre. Tree cores from oaks indicated that the oaks were generally over 60 years of age at breast height.

The stand offers several options in terms of upland hardwood management. Depending on the objectives, management and the subsequent regeneration process of the stand can focus on either oaks or yellow-poplar. Given the presence of white oak and northern red oak, the stand offers a unique opportunity to demonstrate oak regeneration techniques. However, widespread difficulty has been encountered when trying to regenerate oaks on average or productive sites due to erratic seed production, acorn consumption by animals, defoliation and browsing of oak, decreased fire frequency, and climatic change (Lorimer and others 1994).

Oaks are more competitive on drier mesic or xeric sites compared to mesic sites (Hodges and Gardiner 1993). Where site indexes for oaks are 60 feet or below (at 50 years), they are likely to support stable oak forests, however, where site indexes are greater than 60 feet, they are prone to much more competition (Lorimer 1993). The shelterwood method is often recommended to promote the establishment of oak regeneration but competition from yellow-poplar, red maple (*Acer rubrum)*, and sweetgum (*Liquidambar styraciflua)* developing in response to overstory removal and understory disturbance have caused mixed results (Brose and others 1999).

For northern red oak and white oak, the initial growth strategy is root development in lieu of shoot development (Lorimer 1993). White oak is moderately tolerant of shade while northern red oak is intermediately to moderately tolerant and both are capable of developing under moderate shade with good survival rates (Clark and Watt 1971, Smith 1993). However, low understory light levels may be the most limiting factor for the establishment and growth of oak regeneration (Hodges and Gardiner 1993). Therefore, it is suggested to reduce competition, make oaks more competitive, or do both (Loftis 2004). Removal of overstory trees can create increased light conditions. However, even if an opening in the overstory is made, a midstory of shade-tolerant species will not provide enough light for seedlings oaks (Johnson 1979).

The high occurrence of American beech in the stand could pose a problem for oak regeneration. The presence of American beech in this stand is a product of fire exclusion and high basal area (Abrams 2003). Mechanical, chemical, or fire treatments can reduce midstory stems and create increased light levels (Cunningham 2012). In general, American beech is capable producing root suckers and will sprout, however, these sprouts have a high rate of mortality (Wagner 2010). Fire, which could possibly reduce midstory stem densities, could also serve a dual purpose in reducing the leaf litter layer. Development of a large litter layer will inhibit oak regeneration from establishing. Reducing the litter and duff layers below 2 inches before the seed fall and leaf fall with prescribed fire can favor oaks seedlings (Dey and Fan 2009).

Historically, fire played an important role for oak development (Abrams 2003, Lorimer 1993). Oaks are capable of surviving higher intensity fires and are more fire resistant than their competitors (Brose and others 1999, Van Lear 2004). Burning is beneficial for oak by reducing competition and top-killing poorly formed oak advance reproduction (Loftis 2004). Single burn research has shown mixed results, however research with multiple burns has shown more positive results (Brose and others 2014). Thinning with multiple burns was shown to successfully increase the relative density of large oak and hickory seedlings (Iverson and others 2008). Timing of burn is crucial. Multiple winter burns have been shown to reduce competition (Barnes and

Van Lear 1998). Growing season burns are better for competition control than winter burns (Brose and others 2014). With growing season fires, oak is likely to survive if advance regeneration is greater than .5 inches root collar diameter and is capable of sending up new sprouts. (Van Lear 2004). Winter fires, where root reserves are at their highest, have the greatest ability to cause hardwoods to sprout following being top killed. Spring and summer fires can cause mortality in small diameter oak stems but can benefit the larger ones (Van Lear 2004). A study by Adams and Rieske (2001) showed that for white oak, seedling growth (height, shoot elongation, diameter, and specific leaf mass) was greatest on once burned sites, intermediate on twice-burned sites, and least on unburned control sites. While loss of growth does occur, multiple burns can greatly reduce competitors to oak.

CONCLUSION

To aid landowners with upland hardwood management, a demonstration forest stand was proposed for the MOT demonstration forest. The inventory of the stand showed a high proportion of oaks in the overstory, a beech dominated midstory, and absent understory. Reduction of the high amount of leaf litter and beech in the understory are top priority for facilitation of oak regeneration. Future management decisions largely hinge on reductions of the leaf litter layer and changes in basal area and light conditions created with the removal of American beech and sweetgum.

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LITERATURE CITATIONED

Adams, A.S.; Rieske, L.K. 2001. Herbivory and fire influence white oak (*Quercus alba* L.) seedling vigor. Forest Science. 47(3): 331-337.

Abrams, M.D. 2003. Where has all the white oak gone? BioScience. 53(10): 927-939.

Barnes, T.A.; Van Lear, D.H. 1998. Prescribed fire effects on advanced regeneration in mixed hardwood stands. Southern Journal of Applied Forestry. 22(3): 138-142.

Brose, P.H.; Dey, D.C.; Waldrop, T.A. 2014. The fire—oak literature of eastern North America: synthesis and guidelines. Gen. Tech. Rep. NRS-135. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 98 p.

Brose, P.H. Van Lear, D.H., Keyser, P.D. 1999. A shelterwood-burn technique for regenerating productive upland oak sites in the Piedmont Region. Southern Journal of Applied Forestry. 16(3): 158-163.

Clark, F.B.; Watt, R.F. 1971. Silvicultural methods for regenerating oaks. In: Oak symposium proceedings. 1971 August 16-20. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station: 37-43.

Cunningham, K.K. 2012. First year response of oak natural regeneration to a shelterwood harvest and midstory competition control in the Arkansas Ozarks. In: Butnor, J. R., ed. 2012. Proceedings of the 16th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 76-81.

Dey, D.C.; Fan, Z., 2009. A review of fire and oak regeneration and overstory recruitment. In: Hutchinson, T.F., ed. Proceedings of the 3rd fire in eastern oak forests conference; Carbondale, IL. Gen. Tech. Rep. NRS-P-46. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 2-20.

Hodges, J.D.; Gardiner, E.S. 1993. Ecology and physiology of oak regeneration. In: Loftis, D.L.; McGee, C.E., eds. Oak regeneration: Serious problems, practical recommendations (symposium proceedings). Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 54-65.

Iverson, L.R.; Hutchinson, T.F.; Prasad, A.M.; Peters, M.P. 2008. Thinning, fire, and oak regeneration across a heterogeneous landscape in the eastern U.S.: 7-year results. Forest Ecology and Management. 255: 3035-3050.

Johnson, R.L. 1979. Adequate oak regeneration – a problem without a solution? In: Management and utilization of oak, Proceedings 7th annual hardwood symposium of the hardwood research council. Cashiers, NC: Hardwood Research Council: 59-65.

Loftis, D.L. 2004. Upland oak regeneration and management. In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 163-167.

Lorimer, C.G. 1993. Causes of the oak regeneration problem. In: Loftis, D.L., McGee, C.E., eds. Oak regeneration: serious problems, practical recommendations (symposium proceedings). TN. Gen. Tech. Rep. SE-84. U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 14-39.

Lorimer, C.G., Chapman, J.W., Lambert, W.D. 1994. Tall understory vegetation as a factor in the poor development of oak seedlings beneath mature stands. Journal of Ecology. 82(2): 227-237.

Smith, D.W. 1993. Oak regeneration: the scope of the problem. In: Loftis, D.L., Mcgee, C.E., eds. Oak regeneration: serious problem, practical recommendations (symposium proceedings). Gen. Tech. Rep. SE-84. U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 40-52.

Wagner, S.; Collet, C.; Madsen, P.; [and others]. 2010. Beech regeneration research: from ecological to silvicultural aspects. Forest Ecology and Management. 259: 2172-2182.

Van Lear, D.H. 2004. Upland Oak Ecology and Management. In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 123-127.

EPICORMIC DEVELOPMENT IN POLE-SIZE WHITE OAK (*QUERCUS ALBA* L.) PROGENY TESTS THREE YEARS FOLLOWING CROWN RELEASE

Andrew Meier and Mike R. Saunders[1]

Abstract—Epicormic development has been traditionally thought to be a response to increased light and to varying vigor and genetics among individuals within a species, but epicormic ontogeny has increasingly been shown to be a more fundamental factor. Previous studies have quantified the impact of tree vigor on epicormic sprouting and epicormic branch size following silvicultural treatments, but few assessed the genetic component of development. In this study, data from a crop tree release treatment in a 25 year old white oak (*Quercus alba* L.) progeny test in Indiana was used to isolate the relative influences of tree vigor and progeny-level genetic variation on epicormic development. We found no significant difference in the total number of epicormic branches three years following crop tree release ($p = 0.940$), but a significant increase in epicormic branch size ($p < 0.001$). Variation explained by family effects was stronger for branch size than for branch numbers, but provenance did not improve fits for any model. Tree vigor-related variables were significant predictors in all models, but the epicormic response was driven most strongly by pre-treatment epicormic composition. Therefore, a major objective in the management of young oak stands should be to maintain tree vigor to avoid the initial sprouting of epicormics and to remove individuals with many epicormics early in the rotation.

INTRODUCTION

Epicormic branches in many trees sprout from buds that have either been dormant since initial shoot elongation or have descended from primary buds initiated on an annual shoot (Del Tredici 2001, Meier and others 2012). In the oaks (*Quercus* spp.), epicormic buds can be traced to an original primary bud on an expanding annual shoot (Fontaine and others 1999, Yokoi and Yamaguchi 1996, Meier and Saunders 2013). As shoots become older, single buds develop into complex epicormic structures consisting of numerous buds and sprouts. Diameter expansion of these structures over time can produce knots in the wood that reduce log value (Stubbs 1986, Meadows and Burkhardt 2001).

Environmental and genetic factors interact to influence the development of epicormics on individual trees (Meadows 1995, Colin and others 2010, Meier and others 2012). Genetic effects on the development of complex epicormic structures have not been widely studied (Meier and others 2012), though some level of genetic control over initial epicormic sprouting has been demonstrated (Bowersox and Ward 1968, Jensen and others 1997). In oaks, the strongest genetic variation in epicormics occurs at the species level (Meadows 1995), with the white oak group (section Quercus) more susceptible to sprouting than the red oak group (section Lobatae) (Meier and others 2012). Provenance level genetic effects are generally weak (Jensen and others 1997, Colin and others 2010), but progeny level heritability has been reported to be high for some species (Jensen and others 1997).

Silvicultural treatments manipulate environmental conditions, thereby potentially impacting epicormic trajectories. Treatments that increase individual tree vigor are sometimes associated with lower levels of epicormic branching (Miller 1996, Meadows and Goelz 2002, Lockhart and others 2006) compared to low vigor, overstocked stands (Evans 1987, Nicolini and others 2001). However, by increasing light levels below the main canopy, these treatments can also create an environment conducive to epicormic growth (Spiecker 1991, Meadows 1995, Yokoi and Yamaguchi 1996) and thinning stands with stressed or low vigor trees tends to exacerbate incipient epicormic problems (Devine and Harrington 2006, Dimov and others 2006).

This study compares the relative effects of tree vigor, progeny level genetic variation and epicormic ontogeny on the epicormic response to complete crown release in polesize white oak (*Q. alba* L.) plantations. We compared the epicormic response in both high and low

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vigor trees from families with low, medium and high epicormic tendencies. To simulate vigor reduction, a subset of these trees was subject to partial crown removal. We hypothesized that: 1) branch size and persistence would increase in response to crown release, but the total number of branches would show only a minimal response; 2) crown removal through pruning would stimulate significant increases in branch numbers while significantly reducing tree diameter growth; 3) the overall number of epicormic branches on an individual tree would vary among white oak families, but the size and persistence of branches would be influenced more by tree vigor components; and 4) epicormic characteristics three years following treatment would be strongly influenced by pre-treatment characteristics.

METHODS

Sites

This study was implemented in two plantations that are part of a half-sib white oak progeny test in Indiana. They are located at the Jasper-Pulaski Fish and Wildlife Area (JP) (41° 09' N, 86° 54' W, 1.1 ha) and the Harrison-Crawford State Forest (HC) (38° 15' N, 86° 15' W, 1.5 ha). Acorns were collected in 1982 from stands in Indiana, Illinois and Missouri and grown for one year in Indiana's Vallonia State Tree Nursery. Seedlings were outplanted in the spring of 1984 in row plots of 4 trees per family; there were 6 replicates of each family at JP and 8 replicates at HC (Coggeshall 1993). Height data has been collected periodically, with the most recent comprehensive measurement of all plantations completed in the fall and winter of 2005-06. This measurement cycle also included diameter at breast height (dbh, at 1.37 m) and qualitative ratings for tree form and epicormic branching (S. Rogers, unpublished data). Immediately prior to the current study, dbh (cm) at JP and HC averaged 16.9 (sd [standard deviation] = 3.8) and 14.3 (sd = 3.3), respectively. Basal area (m^2 ha^{-1}) was slightly lower at JP (= 21.8, sd = 4.3) than at HC (= 22.2, sd = 2.9). Soil at HC is a Haymond silt loam and at JP a Maumee sandy loam (O'Connor and Coggeshall 2011); HC has a slightly higher site quality with a site index (height at age 50 years) of 21 m compared to 20 m at JP.

Treatments

Based on family mean epicormic ratings from 2005-06, we randomly selected six individual families with high (in top quartile), medium (in mid two quartiles), and low (in bottom quartile) epicormic tendencies. Within each resulting epicormic class (epiC) individual trees were stratified into upper (dominant and codominant) and lower (intermediate and suppressed) pre-treatment crown classes (cc) to capture within family variation in tree vigor. An approximately equal number of trees were then randomly selected as study trees from within each combination of epiC and cc at each site. Full details of the study tree selection protocols, as well as a list of study families, can be found in Meier (2012).

Study trees were given a full crown-touching release (Miller and others 2007) requiring the removal of 1 to 8 trees in the immediately adjacent planting locations. Trees were randomly assigned to one of three pruning treatments: no crown removal, one-third (= low) or two-thirds (=high) live crown removal. Crowns were pruned from the bottom of the crown upward. Because of the variation in crown structure for individual trees, we reclassified trees into quantitative pruning classes following treatment (prC, table 1); these classes were similar to the initial pruning treatment assignments though there was some variation (data not shown).

Data Collection

Initial data collection occurred prior to the installation of treatments from January to March 2010 and was repeated in both late spring and early fall of 2010-12, a total of seven measurement cycles. Variables measured are shown in Table 1. Growth related variables were measured prior to treatment and only during fall measurement cycles; epicormic branch counts were taken at every cycle. Branches were counted on the trunk section from 0.15-3.81 m in height, allowing for a 0.15 m stump height. This corresponds to approximately a 3.66 m butt log, which is a common merchantable log length in Indiana. Epicormic branches were distinguished from sequential branches by their generally large branch angles, weak trunk attachment, small size and highly variable branch structure (Spiecker 1991), though only rarely were live sequential branches encountered in this section of any bole. Epicormic branches were considered dead if they were brittle or easily broken off the tree. New branches were tallied separately for each of the post-treatment measurement cycles. Only branches with a single green growth unit were considered to be new branches; we had no way of distinguishing between growth units resulting from multiple flushes within a single growing season and growth units developing over multiple years (Spiecker 1991). Epicormic branches were tallied by branch diameter classes (brDC), as measured at the base of the branch just above the basal swell. These classes were: brDC1: <0.64 cm, brDC2: 0.64-1.27 cm, brDC3: 1.27-2.54 cm, brDC4: 2.54-3.81 cm and brDC5 >3.81 cm. The threshold between brDC1 and brDC2 corresponds to the size at which an epicormic must in all cases be considered a defect in US Forest Service log grading (Rast and others 1973). When multiple epicormic branches emanated from the same underlying epicormic structure, they were counted as a single branch (Spiecker 1991) and the diameter of the cluster was recorded. The number of clusters was not tallied.

Table 1—Descriptions and mean values for predictor variables and factors that were included in the full models. Variables that were significant in either the br12 or the brBA12 models are listed in bold; variables that were removed from br12 or brBA12 because of high collinearity are shown in italics. Note that response variables are for the total log length, not the per meter average

Variable	Description	Mean value (±SD) or distribution of factors
Response		
br10t[†]	Total number of epicormic branches in 2010 to 3.81 m	53.7 (±25.7)
brBA10t[†]	Total epicormic branch basal area in 2010 (cm^2)	8.6 (±5.0)
br12t	Total number of epicormic branches in 2012 to 3.81 m	50.6 (±26.3)
brBA12t	Total epicormic branch basal area in 2012 (cm^2)	20.1 (±9.9)
Factors		
epiC	Epicormic class (L = low, M = medium, H = high)	L (26), M (27), H (28)
prC	Crown pruning classes (N = none, L = low, H = high)	N (27), L (27), H (27)
cc	Crown class (L = intermediate and suppressed, U = dominant and codominant)	L (39), U (42)
ftg12	The free to grow rating of the tree in 2012, ranging from 0 (no lighted faces) to 4 (all faces receiving sunlight)	0 (2), 1(5), 2(12), 3(31), 4(31)
site	One of two plantation locations, either JP or HC	HC (50), JP (31)
Covariates		
brNewM	Mean number of new branches for all measurement cycles (2010-2012)	12.7 (±8.8)
dbh10	Tree diameter (cm) prior to the 2010 growing season	14.8 (±3.9)
dbh06	Tree diameter (cm) prior to the 2006 growing season	13.3 (±3.2)
dbhincP	Annualized pre-treatment dbh increment (cm) for 2006-2009 growing seasons	0.4 (±0.3)
dbhinc	Mean annual dbh increment (cm) for 2010-2012 growing seasons	0.5 (±0.2)
hgt94	Total height (m) in 1994	3.0 (±0.9)
hgtincP	Annualized pre-treatment height increment (m) for 2006-2009 growing seasons	0.4 (±0.2)
hgtinc	Mean annualized height increment for 2010-2012 growing seasons	0.4 (±0.6)
lcrinc	Mean live crown radius for 2010-2012 growing seasons	1.9 (±0.7)
lcr	Average pre-treatment live crown radius (m)	1.9 (±0.6)
pctlc	Pre-treatment percent live crown	50.5 (±12.5)
ci	Pre-treatment competition index, calculated $\sum_{j=1}^{n} D_j/D_i$ where n is the number of competitors, D_j is dbh of an individual competitor and D_i is dbh of the study tree (from Lorimer 1983)	5.9 (±2.6)

[†]Also a covariate in br12t and brBA12t models.

Data Preparation and Analysis

We modeled the pre-treatment epicormics and post-treatment epicormic response based on two epicormic branch variables: the total number of epicormic branches (br) and the total basal area of epicormic branches (brBA, cm^2) as calculated using the midpoint diameter of each brDC. We chose to use absolute measures for both because relative values in regards to initial conditions would have the potential to disproportionally amplify effects in individuals with small initial values. We elected to analyze responses using only the spring 2010 and fall 2012 data and avoided a complex time series analysis; Meier (2012) found little variation in epicormic characteristics in the intervening measurement cycles. We were also interested in the magnitude of new epicormic sprouting across all years rather than only the number of new epicormic branches present in fall 2012, so we calculated the mean number of new epicormic branches per year (brNewM) and included this in the br and brBA models. Though our branch variables were based on count data, there were no trees with counts of zero for br or brBA. Therefore, these variables were transformed to approximately normal distribution using square root transformations.

We utilized linear mixed effects models to analyze our data and assessed family (fam) variation as a random effect. Site was included as a fixed effect in all models because we did not have a sufficient number of sites to calculate meaningful random effect parameters. Full models for br and brBA included growth, stocking and prior epicormic characteristics (table 1). In all models, when pairs of variables showed high Pearson's correlations ($|r| > 0.7$) one of the pair was removed (table 1, italics) prior to stepwise model reduction. Generally, variables retained had the strongest pairwise relationships with the response variable and were correlated with the fewest other predictors.

Linear mixed effects models of the transformed response variables were fit using restricted maximum likelihood estimation. The significance of variables ($\alpha = 0.05$) in the full models was tested with marginal F-tests and the least significant variable was removed in a stepwise manner (Pinheiro and Bates 2000). Relative AIC was calculated subsequent to each stepwise model reduction and variables were retained if their removal caused a large increase (>5) in the AIC. In some instances, non-significant terms remained in the model when AIC was minimized; these were also removed in the interest of model parsimony. In the reduced models, we used likelihood ratio tests to test the significance of the family random effect by comparing models with identical fixed effects, but with and without family random effects (Pinheiro and Bates 2000). The magnitude of the family effect was assessed using the estimated standard deviation of the random intercept ($\hat{d}$) (Zuur and others 2009). Diagnostic plots of the final models were used to identify variation from the normality assumption and heteroscedasticity of the models. Model fit was further evaluated by calculating lme R^2 (Maj 2011).

In addition to the models described above, absolute differences between transformed br and brBA in spring 2010 and fall 2012 values were assessed with paired t-tests. A Mann-Whitney test was used to test whether there was a difference in the proportion of branches that were new in 2010 to the proportion that were new in 2012. All analyses were performed using the R statistical computing program, version 2.15.0 (R Development Core Team 2012), using the packages stats, nlme (Pinheiro and others 2012), bbmle (Bolker 2012), and lmmfit (Maj 2011).

RESULTS

Diameter Growth

Diameter growth (dbhinc, cm yr^{-1}) across all classes was greater following treatment (= 0.47, sd = 0.21) than prior to treatment (= 0.39, sd = 0.28), and it did not differ significantly by site. Diameter growth varied little by epiC (low: = 0.47, sd = 0.22; medium: = 0.46, sd = 0.22; high: = 0.46, sd = 0.20) and decreased slightly with prC (none: = 0.50, sd = 0.24; low: = 0.46, sd = 0.22; high: = 0.44, sd = 0.16). There was a significant influence of crown class (cc) on diameter increment ($p < 0.001$), with upper crown classes (= 0.58, sd = 0.19) growing faster than lower (= 0.34, sd = 0.17). Trees with initially larger crowns had higher growth rates; unpruned trees in the upper crown class (fig. 1, white circles) had the highest average growth rates, while the most heavily pruned lower crown class trees (fig. 1, gray triangles) had the lowest.

Pre-treatment Epicormic Dynamics

The mean number of epicormic branches prior to treatment (br10) was 14.7 m^{-1} (sd = 7.0). There were no distinct pretreatment differences between the study sites (fig. 2a). Strong pretreatment differences were also not evident between epicormic classes or pruning classes, though there was some separation between crown classes (fig. 3a-c). A large proportion (= 0.37, sd = 0.24) of pre-treatment epicormic branches were newly formed in the prior year. The best model for br10 included only cc as a significant effect; variation attributable to different families was minimal. However, the model had a very low predictive value ($R^2 = 0.08$, table 2).

Means for total branch basal area per meter prior to treatment (brBA10, cm^2) were similar, but significantly different ($p = 0.005$) between HC and JP (fig. 2b). For all other grouping variables, differences were small, variability was high (fig. 3d-f), and none were significant

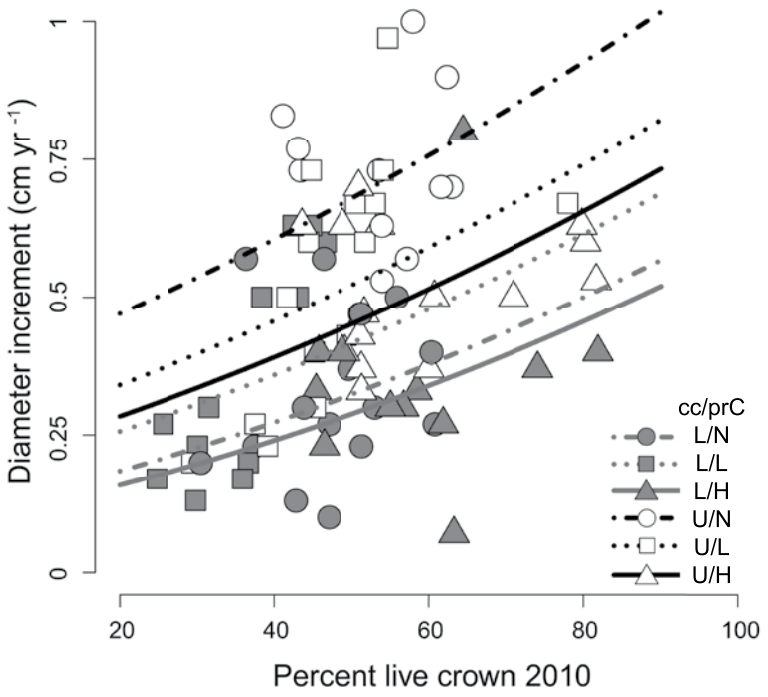


Figure 1—Annual diameter increment following treatment by pre-treatment percent live crown for individual combinations of crown class (cc) and pruning class (prC). Grouping codes are defined in Table 1.

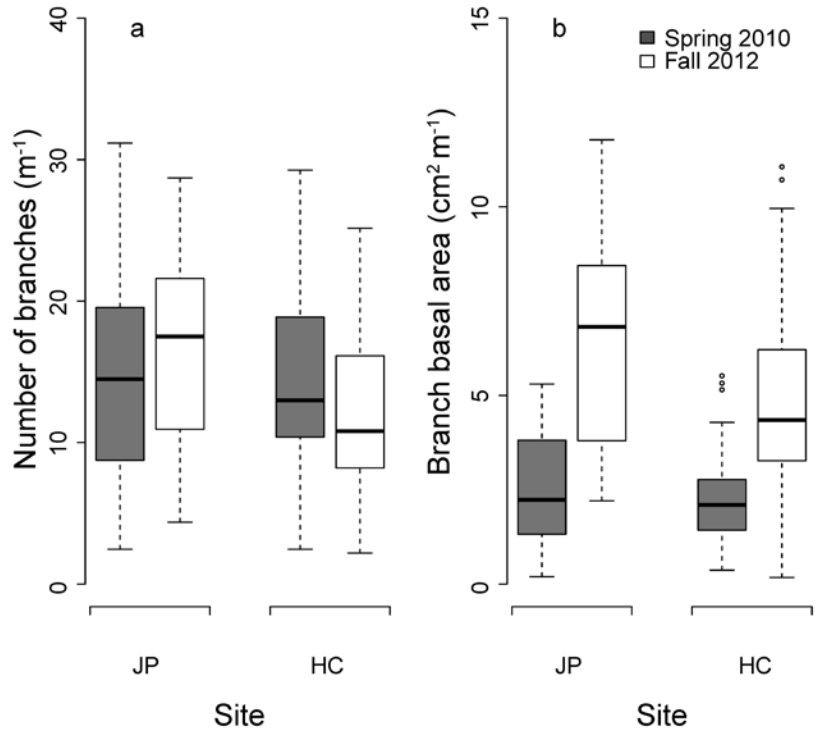


Figure 2—Boxplots indicating the variation in number of epicormic branches per meter (br; a) and branch basal area per meter (brBA; b) by site prior to the first growing season following treatment (br10 or brBA10; gray boxplots) and after three full growing seasons (br12 or brBA12; white boxplots).

in the final model (table 2). Instead, dbh increment prior to treatment (dbhincP) was negatively correlated and tree height in 1986 (hgt86) was weakly positively correlated with brBA10 in the final model (table 2). Family random effects were greater in this model than for br10, but still non-significant (p = 0.578), and model fit was somewhat better (R^2 = 0.20).

Post-treatment Epicormic Dynamics

The number of epicormic branches per meter in fall of 2012 (br12) declined slightly (= 13.9, sd = 7.2) from br10, but was not significantly different (V = 1203, p = 0.968). Branch numbers at JP were higher than at HC (fig. 2a). The proportion of branches that were new was significantly lower three years after treatment (= 0.15, sd = 0.12; V = 301.5, p <0.001). For the various grouping classes, br12 was similar and not significantly different (Fig. 3a-c). Average branch size generally increased for all families (data not shown; see Meier 2012). Four covariates were significant in the final br12 model (table 2); these included pretreatment epicormic composition (br10), competition (ci), pre-treatment heigh increment (hgtIncP), and the average annual production of new shoots (brNewM). Family effects were minimal, and this model explained overall variation well (R^2 = 0.71) (table 2).

In contrast to the total number of branches, there was a significant increase in total branch basal area per meter (V = 3294, p < 0.001) three years following treatment (brBA12, = 5.5, sd = 2.7); this increase was evident across all groups and at both sites (fig. 2, fig. 3). Of the 81 study trees, only 3 had lower brBA in 2012 than 2010; the average increase in brBA of the 3.66 m butt log was 11.1 cm^2 (sd = 8.4), equivalent to about 15 branches transitioning from brDC1 to a defect-causing brDC2. The brBA12 per meter was higher at JP than at HC (fig. 2b) and also differed by epiC (fig. 3d). There was little variation between cc (fig. 3e), but pruning stimulated an increase in brBA in respect to initial levels (fig. 4). The final brBA12 model included brBA10, site and prC table 2). Family effects were stronger than for other models, but still insignificant; variation explained was moderate (R^2 = 0.40).

DISCUSSION

Growth and Vigor

In this study, we considered higher than average diameter growth rates to be indicative of high vigor. Overall, we observed a significant increase in growth rates in this study following crop tree release compared to pre-treatment growth. The growth rate was similar to that for other studies of pole-sized oaks following release (Minckler 1967, Schlesinger 1978). Pre-treatment crown characteristics significantly influenced growth rate and crown pruning caused variation in growth both among and within crown classes (fig. 1). The heavily

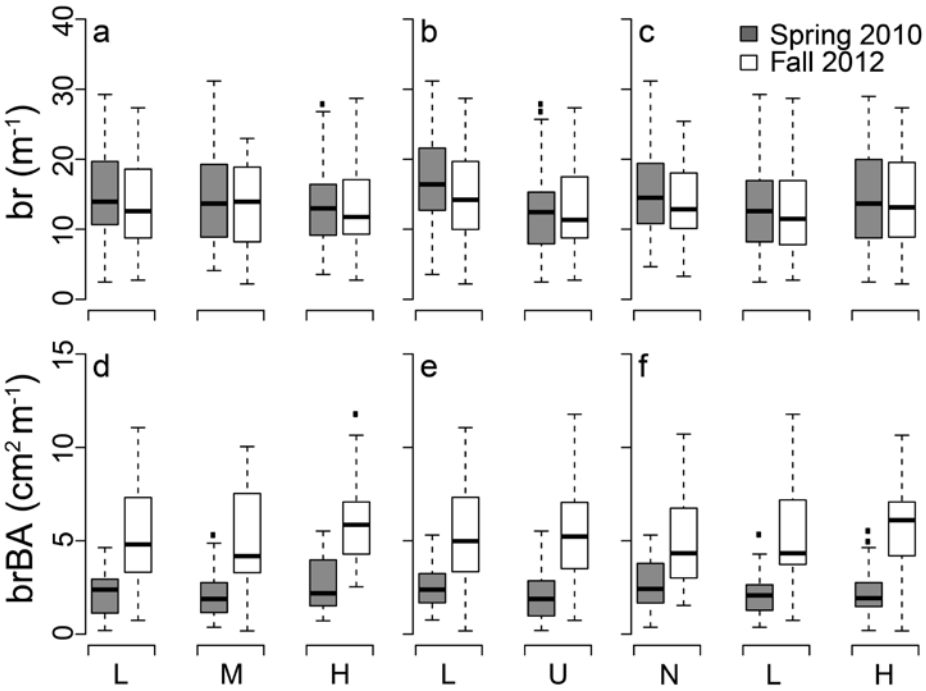


Figure 3—Boxplots for number of epicormic branches per meter (a-c) and branch basal area per meter (d-f) for three grouping factors, epicormic class (epiC: a, d), crown class (cc: b, e) and pruning class (prC: c, f), prior to the first growing season following treatment (gray bars) and after three full growing seasons (white bars). Grouping codes are defined in Table 1. Thick black bars represent sample medians.

Table 2—Parameter estimates for linear mixed effects models of the various response variables. The family effect was calculated with a likelihood ratio test of models with and without the family random effect included; the associated *P* value indicates whether there was significant variation between models. Significance in all cases was assessed at α = 0.05

Response variable	Family random effect		lmm $R^{2\,\dagger}$	Parameters			
	$\hat{d}$	*P* value		Fixed effects	Estimate	SE	*P* value
sqrt(br10t)	<0.001	0.999	0.08	intercept	7.631	<0.001	<0.001
				ccU	-1.015	0.387	0.011
sqrt(brBA10t)	0.200	0.578	0.20	intercept	2.634	0.304	<0.001
				siteJP	0.583	0.198	0.005
				dbhincP	-1.605	0.364	<0.001
				hgt86	1.587	0.726	0.033
sqrt(br12t)	<0.001	0.999	0.71	intercept	0.521	0.703	0.462
				ccU	0.658	0.271	0.018
				siteJP	1.342	0.293	<0.001
				sqrt(br10)	0.538	0.085	<0.001
				brNewM	0.078	0.018	<0.001
				hgtincP	-1.347	0.610	0.031
				ci	0.196	0.055	<0.001
sqrt(brBA12t)	0.228	0.440	0.40	intercept	1.454	0.401	0.001
				prC1	0.730	0.250	0.005
				prC2	0.555	0.240	0.024
				siteJP	0.705	0.213	0.002
				sqrt(brBA10)	0.781	0.117	<0.001

$^{\dagger}R^2$ values were calculated as adjusted R^2 when multiple fixed effects remained in the final model.

pruned upper canopy trees had growth rates that were close to the growth rates of the most vigorous lower canopy trees, fulfilling our objective of simulating the development of low vigor trees in the upper crown classes.

Branch Number and Size

We confirmed our first hypothesis that the increase in size of epicormic branches is a much more important component of the response to crop tree release than increases in the overall number of branches. We observed no changes in br three growing seasons after thinning (fig. 3a-c), but did observe significant increases in brBA (fig. 3d-f). These brBA increases were driven by increased diameters of extant branches. Transitions from the smallest, more ephemeral branches in brDC1 to more persistent, larger branches in brDC2 accounted for most of the increase in brBA. Trees with the highest number of large branches prior to treatment were those with the highest brBA subsequent to treatment (fig. 4).

Our observed br and brBA dynamics have been reported in other oak thinning studies (Minckler 1967, Yokoi and Yamaguchi 1996), though some authors have reported significant increases in branch numbers (Stubbs 1986, McDonald and Ritchie 1994). The lack of a response in branch numbers is often attributed to vigor increases in released trees (Lockhart and others 2006) or the removal of low vigor trees (Meadows and Goelz 2002), while branch stimulation often occurs following the release of very low vigor trees (Devine and Harrington 2006, Dimov and others 2006). Though we did observe increased diameter growth at the tree level, and therefore increased vigor in this study, vigor following release was not a significant covariate in the br model. Rather, the epicormic impact we observed may have occurred as a result of release at the level of individual branches. Since epicormic branches were numerous on most trees in this study prior to treatment, we suggest that increased light levels following treatment promoted increased vigor of the

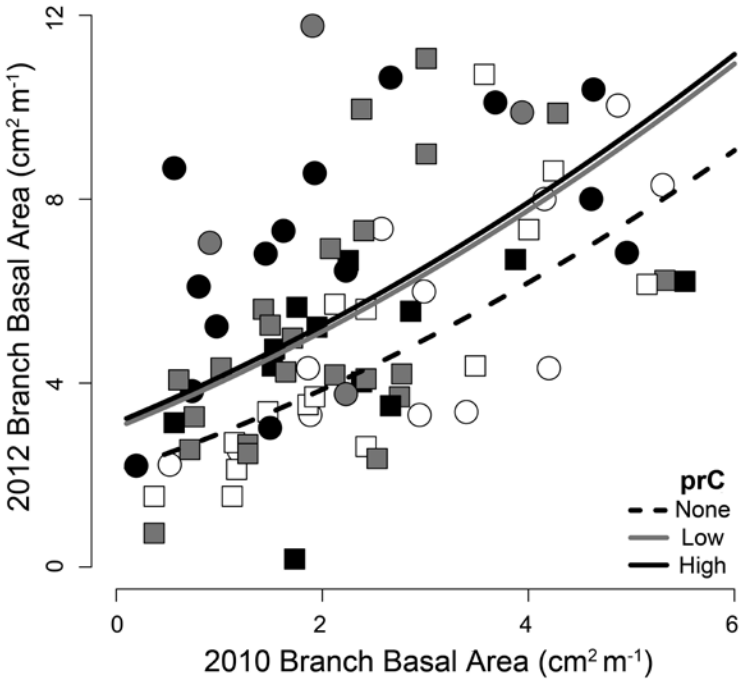


Figure 4—Relationship between branch basal area per meter in 2010 and 2012 for sites JP (circles) and HC (squares). Regression lines and symbol colors represent pruning classes (prC) at each site. Model parameters for the full mixed model are shown in Table 2.

dominant meristems of these extant branches, thus increasing the physiological control by these meristems over dormant buds in the subtending bud cluster (Chen and others 1996, Meier and others 2012). Our finding that the proportion of branches that were less than one year old was halved between the initial 2010 measurement and the final 2012 measurement further indicates CTR actually limited new sprouting by favoring existing sprouts. Apical dominance is a well-established concept in the literature (Brown and others 1967, Kormanik and Brown 1969), and there is no reason to assume that this concept should not apply in the same way at the level of an individual epicormic structure as it does at the level of the entire tree. In trees with few epicormic sprouts prior to canopy release, prolific sprouting may occur initially, as observed in other studies, followed by a period of branch development similar to our study.

Crown Pruning

Crown pruning and the associated reduction in vigor had surprisingly no significant impact on branch sprouting (fig. 2, 3c), which was contrary to our second hypothesis. In a number of species, branch pruning has been shown to stimulate epicormic sprouting (Evans 1987, Springmann and others 2011), but it commonly occurs from the buds nearest to the point of damage (Kormanik and Brown 1969, Wignall and Browning 1987). Therefore, it is possible that crown pruning did indeed stimulate epicormic sprouting in this study, but since crowns in most cases were pruned well above our 3.81 m counting limit, the resulting new branches would not have been tallied.

Although crown pruning did not stimulate the sprouting on the lower bole, it did influence development of extant branches (fig. 4). The brBA response is probably the result of two factors. The first relates to reduction in self-shading on the bole after the partial crown removal. This would allow for higher photosynthetic rates in epicormic branches lower on the tree, which in turn, would lead to increased branch size. The second factor could be related to nutrient sink dynamics. With the removal of branches in the live crown during the dormant season, the amount of potential leaf area in the crown would have been substantially less at the time of bud flush than the nutrient base that had supported leaf area in the previous growing season. Therefore, previously low vigor epicormic branches on the bole could have become stronger nutrient sinks and benefited from an influx of nutrients that would otherwise not have been available (Ishii and others 2007, Meier and others 2012).

Genetics and Ontogeny

In general, family effects on epicormic development after either pruning or crop tree release were weak or even nonexistent (table 2). Somewhat contrary to our

third hypothesis, progeny level genetic variation was higher for branch size than branch numbers, though the effect was still negligible. In contrast to genetic effects, epicormic characteristics prior to treatment had a significant effect on epicormic characteristics three years following treatment, confirming our final hypothesis. Many studies have alluded to ontogenic effects on epicormic development (e.g. Ward 1966, Minckler 1967, Miller 1996), but only recently has ontogeny become an important line of study (Fontaine and others 1999, Colin and others 2010).

More specifically, our results suggest that the numbers of epicormics are more strongly controlled by intrinsic developmental processes and are less responsive to environmental and genetic factors in pole-sized white oaks (table 2). The importance of these ontogenic factors is demonstrated by the substantially stronger fit of the br12 model (lme $R^2 = 0.77$) than the br10 model (lme $R^2 = 0.08$); in the br12 model we were able to include br10 and brNewM as predictors while there was no quantitative data available to include in the br10 model describing epicormic ontogeny. In another study at these sites, it was found that the number of epicormic structures on a tree is heavily influenced by the per meter density of non-epicormic branches, a factor that is set at a very young age (Meier and Saunders 2013). Therefore, selection of individuals that show greater apical dominance at an early age could lead to a decrease epicormic numbers later.

On the other hand, variations in brBA among groups became more accentuated following CTR and there was a significant brBA increase in response to release. The stronger family random effect suggests that there may be more potential for selection of superior genotypes by considering the cumulative size of branches rather than the absolute number of branches (sensu Meier 2012). Further research seems merited, both for thinned and unthinned stands. Additionally, with half-sib plantations, some of the family variation is introduced by the genetics of the paternal tree; a stronger understanding of the genetic effect could be gained from studying either controlled crosses or clones, or by conducting parentage analysis on individual half-sibs to identify which share the same parent.

CONCLUSIONS

Colin and others (2010) defined a research framework in which the relative impacts of genetics and environment should be studied in terms of their effect on epicormic ontogeny. In this study, we focused on the contribution of family history and tree vigor to total branch numbers and branch size following crop tree release. The family effect on total branch numbers was negligible, but the family effect on branch size was slightly more important. Vigor was an important influence on the number and size of branches present prior to treatment, with higher vigor trees having fewer, smaller branches than less vigorous trees. However, epicormic response to crop tree release was most strongly influenced by the epicormic composition prior to treatment. This suggests that a better understanding is needed of the ontogenic factors that influence epicormic development early in a tree's life.

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LITERATURE CITED

Bolker, B. 2012. Package 'bbmle': tools for general maximum likelihood estimation. R package version 1.0.4.1. http://cran.r-project.org/web/packages/bbmle/index.html [Date accessed: May 10, 2015].

Bowersox, T.W.; Ward, W.W. 1968. Auxin inhibition of epicormic shoots in white oak. Forest Science. 14: 192-196.

Brown, C.L.; McAlpine, R.G.; Kormanik, P.P. 1967. Apical dominance and form in woody plants: a reappraisal. American Journal of Botany. 54: 153-162.

Chen, H.-J.; Bollmark, M.; Eliasson, L. 1996. Evidence that cytokinin controls bud size and branch form in Norway spruce. Physiologia Plantarum. 98: 612-618.

Coggeshall, M.V. 1993. Oak tree improvement in Indiana. Annales des Science Forestières. 50: 416s-419s.

Colin, F.; Ducousso, A.; Fontaine, F. 2010. Epicormics in 13-year-old *Quercus petraea*: small effect of provenance and large influence of branches and growth unit limits. Annals of Forest Science. 67: 312.

Del Tredici, P. 2001. Sprouting in temperate trees: a morphological and ecological review. Botantical Reviews. 67: 121-140.

Devine, W.D.; Harrington, C.A. 2006. Changes in Oregon white oak (*Quercus garryana* Dougl. ex Hook.) following release from overtopping conifers. Trees–Structure and Function. 20: 747-756.

Dimov, L.D.; Stelzer, E., Wharton, K. [and others]. 2006. Effects of thinning intensity and crown class on cherrybark oak epicormic branching five years after treatment. In: Conner, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 606-610.

Evans, J. 1987. The control of epicormic branches. In: Patch, D., ed. Advances in practical arboriculture. Bull. 65. London, England: Forestry Commission: 115-120.

Fontaine, F.; Kiefer, E.; Clement, C. [and others]. 1999. Ontogeny of the proventitious epicormic buds in *Quercus petraea*. II. From 6 to 40 years of the tree's life. Trees–Structure and Function. 14:83-90.

Ishii, H.T.; Ford, E.D.; Kennedy, M.C. 2007. Physiological and ecological implications of adaptive reiteration as a mechanism for crown maintenance and longevity. Tree Physiology. 27: 455-462.

Jensen, J.S.; Wellendorf, H.; Jager, K. [and othrs]. 1997. Analysis of a 17-year old Dutch open-pollinated progeny trial with *Quercus robur* (L.). Forest Genetics. 4: 139-147.

Kormanik, P.P.; Brown, C.L. 1969. Origin and development of epicormic branches in sweetgum. Res. Pap. SE-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 17 p.

Lockhart, B.R.; Michalek, A.J.; Lowe, M.W. 2006. Epicormic branching in red oak crop trees five years after thinning and fertilizer application in a bottomland hardwood stand. In: Conner, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 595-598.

Lorimer, C.G. 1983. Tests of age-independent competition indices for individual trees in natural hardwood stands. Forest Ecology and Management. 6: 343-360.

Maj, A. 2011. Package 'lmmFIT': goodness-of-fit measures for linear mixed models with one-level-grouping. R package version 1.0. http://cran.r-project.org/web/packages/lmmfit/index.html [Date accessed: May 10, 2015].

McDonald, P.M.; Ritchie, M.W. 1994. Epicormic branching of California black oak: effect of stand and tree characteristics. Northwest Science. 68: 6-10.

Meadows, J.S. 1995. Epicormic branches and lumber grade of bottomland oak. In: Lowery, G.; Meyer, D., eds. Proceedings of the 23rd annual hardwood symposium: advances in hardwood utilization: following profitability from the woods through rough dimension. Memphis, TN: National Hardwood Lumberman's Association: 19-25.

Meadows, J.S.; Burkhardt, E.C. 2001. Epicormic branches affect lumber grade and value in willow oak. Southern Journal Applied Forestry. 25: 136-141.

Meadows, J.S. and J.C.G. Goelz. 2002. Fourth-year effects of thinning on growth and epicormic branching in a red oak-sweetgum stand on a minor stream bottom site in west-central Alabama. In: Outcalt, K.W., ed. Proceedings of the 11th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-48. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 201-208.

Meier, A.R. 2012. Aspects of epicormic development in *Quercus alba* (L.) and other eastern North American oak species in relation to genetics, tree vigor and silvicultural treatments. West Lafayette, IN: Purdue University. 244 p. M.S. thesis.

Meier, A.R.; Saunders, M.R.; Michler, C.H. 2012. Epicormic buds in trees: a review of bud establishment, development and dormancy release. Tree Physiology. 32: 565-584.

Meier, A.R.; Saunders, M.R. 2013. Assessing internal epicormic dynamics in *Quercus alba* L. using CT scanning: the strong effects of shoot development and tree growth relative to progeny level genetic variation. Trees–Structure and Function. 27: 865-877.

Miller, G.W. 1996. Epicormic branching on central Appalachian hardwoods 10 years after deferment cutting. Res. Pap. NE-702. Radnor, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 9 p.

Miller, G.W.; Stringer, J.W.; Mercker, D.C. 2007. Technical guide to crop tree release in hardwood forests. Pub. PB1774. Knoxville, TN: University of Tennessee Extension. 23 p.

Minckler, L.S. 1967. Release and pruning can improve growth and quality of white oak. Journal of Forestry. 65: 654-655.

Nicolini, E.; Chanson, B.; Bonne, F. 2001. Stem growth and epicormic branch formation in understorey beech trees *(Fagus sylvatica* L.). Annals of Botany. 87: 737-750.

O'Connor, P.A; Coggeshall, M.V. 2011. White oak seed source performance across multiple sites in Indiana through age 16. In: Fei, S.; Lhotka, J.M.; Stringer, J.W. [and others], eds. Proceedings of the 17th central hardwood forest conference. Gen. Tech. Rep. P-78. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 358-363.

Pinheiro, J.C.; Bates, D.M. 2000. Mixed-effects models in S and S-plus. New York: Springer-Verlag. 528 p.

Pinheiro, J.; Bates, D.; DebRoy, S.; Sarkar, D.; R Core Team. 2012. Package 'nlme': linear and nonlinear mixed effects models. R package version 3.1-104. http://cran.r-project.org/web/packages/nlme/index.html [Date accessed May 10, 2015].

R Development Core Team. 2012. R: A language and environment for statistical computing, version 2.15.0. Vienna, Austria: R Foundation for Statistical Computing.

Rast, E.D.; Sonderman, D.L.; Gammon, G.L. 1973. A guide to hardwood log grading (revised). Gen. Tech. Rep. NE-GTR-1. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 32 p.

Schlesinger, R.C. 1978. Increased growth of released white oak poles continues through two decades. Journal of Forestry. 76: 726-727.

Spiecker, H. 1991. Controlling the diameter growth and the natural pruning of sessile and pedunculate oaks. Schriftenreihe der Landesforstverwaltung. Stuttgart, Germany: Baden-Württemberg Band 72. 135 p.

Springmann, S.; Rogers, R.; Spiecker, H. 2011. Impact of artificial pruning on growth and secondary shoot development of wild cherry (*Prunus avium* L.). Forest Ecology and Management. 261: 764-769.

Stubbs, J. 1986. Hardwood epicormic branching – small knots but large losses. Southern Journal of Applied Forestry. 10: 217-220.

Ward, W.W. 1966. Epicormic branching of black and white oaks. Forest Science. 12: 290-296.

Yokoi, S.; Yamaguchi, K. 1996. Origin of epicormic branches and effect of thinning on their development in *Quercus mongolic* var. *grosserrata*. Journal of the Japanese Forest Society. 78: 169-174.

Wignall, T.A.; Browning, G.; Mackenzie, K.A.D. 1987. The physiology of epicormic bud emergence in pedunculate oak (*Quercus robur* L.). Responses to partial notch girdling in thinned and unthinned stands. Forestry. 60: 45-56.

Zuur, A.; Ieno, E.; Walker, N.; Saveliev, A.; Smith, G.M. 2009. Mixed effects models and extensions in ecology with R. New York: Springer. 574 p.

THE ALLELOPATHIC INFLUENCE OF POST OAK (*QUERCUS STELLATA*) ON PLANT SPECIES IN SOUTHERN U.S. FORESTS

Nicollette A. Baldwin and Michael K. Crosby[1]

Abstract—Post oak (*Quercus stellata*) is a commonly occurring tree in the southeastern United States, offering forage and shelter for a variety of wildlife as well as having commercial uses. This species is often planted in parks and urban green-spaces for the shade it offers. Previous studies have found that parts of the plant can be toxic to livestock and that it can inhibit the germination and/or growth of plant species in its vicinity. This study focuses on the allelopathic potential of post oak in an urban, old growth forest. Post oak was selected subsequent to an understory inventory of plant species on a plot established in Marshall Forest in Rome, GA. White oak (*Quercus alba*) and chestnut oak (*Quercus montana*) seeds, and muscadine (*Vitis rotundifolia)* were used to determine if leachates prepared from leaves collected from Post oak inhibit germination and/or growth. Radish (*Raphanus sativus)* was also used, as it was previously found to be impacted by post oak. Two different concentrations of leachate were prepared and tested on the selected species. The results revealed significant differences in both germination rates and mean sprout lengths between the control (distilled water) and both concentration groups. No significant difference was found between the two concentrations of leaf leachate during the experiment. These results suggest that post oak inhibits the germination rate and sprout length of the tested species. It is important that resource managers understand these relationships in managed landscapes (e.g., parks). In the future, it may be possible to utilize allelochemicals from post oak as a biocontrol agent.

INTRODUCTION

Allelopathy is the process by which an organism produces biochemicals that have a potentially detrimental impact on the survival, growth, and/or reproduction of another organism (Pérez and others 2013). While the mechanism of allelopathy is not entirely known, it has been found that some biochemicals interfere with plant physiological processes through disrupting the cell membrane, target energy producing, and energy consuming steps (Rietveld 1983). Interfering with these processes in plants will result in impacts on growth and/or survivability of seedlings. This phenomenon has not been thoroughly explored and the effects that certain plants (e.g., trees) have on their surrounding environment (i.e., the understory) warrants further investigation.

Allelochemicals can be introduced into the environment by two methods. The first is via living plant matter where rainfall washes allelochemicals off the leaves and into the soil. The second method is leaf litter, which occurs when leaves senesce, fall to the ground, and decompose on the forest floor. The decomposition of leaves allows the allelochemicals to leach into the soil. Determining the strength of the allelopathic chemicals present in plants is important for determining the impacts to surrounding plant species and may be useful for producing natural, targeted, herbicides allowing for more efficient crop production by killing specific weeds and not harming crop plants. Understanding allelopathic processes is also important when dealing with invasive species, as many are known to produce biochemicals that inhibit surrounding plants, allowing the invasive plant to outcompete them for resources (Pérez and others 2013).

Post oak (*Quercus stellata)* has not been widely explored as a potential allelopathic species and has been classified as a moderate allelopathic species (Coder 1999). This species occupies rocky/sandy ridges and dry woodlands, is native to the eastern U.S., and is considered drought resistant (Starskey 1990). Post Oak is also commonly planted in parks for use as shade trees because of the large, lobed leaves. Buehler (2010) used post oak as a control to compare a known allelopathic species *Casuarinas* on radish and beans because Post Oak was not previously documented as an allelopathic species. The results demonstrated that post oak significantly inhibited germination of radish and beans.

The objective of this study was to determine the allelopathic effects of post oak leachates on reduction

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of germination and sprout lengths in white oak (*Quercus alba*), chestnut oak (*Quercus montana*), muscadine (*Vitis rotundifolia*), and radish (*Raphanus sativus*). The selected species were chosen because of an initial understory survey. Muscadine was the most prevalent species found under post oak. The second objective was to determine if there was a significant difference between two concentrations of the allelopathic leachates.

MATERIALS AND METHODS

Plant Collection and Leachate Preparation

A half-acre plot was established in Marshall Forest, an urban, old-growth forest in Rome, GA, in September 2014. An inventory of understory species was recorded for post oaks as well as several associated species from study plots. Species diversity under post oak, chestnut oak (*Quercus montana)*, white oak (*Quercus alba)*, loblolly pine (*Pinus taeda)*, and pignut hickory (*Carya glabra)* were assessed, and post oak was found to have a lower species diversity (table 1). Fresh leaves were then collected from post oak trees and were immediately transported to a laboratory at Shorter University where they were weighed to make two different concentrations of leachate. Two hundred fifty-six grams of fresh green leaves were weighed for concentration one and placed in a large basin to prepare for leachate extraction. Five hundred twelve grams of fresh green leaves were weighed for concentration two and placed in another large basin for preparation of leachates. The leaves were covered with 1L of distilled water and allowed to soak for seven days to extract allelochemicals. Once the allelochemicals had been extracted, leachate was vacuum filtered through Whatman #1 filter paper and stored at 4 degrees Celsius. The pH was measured and recorded as 5.7.

Preparation of Selected Species and Recording Data

The study design consisted of four species (radish, chestnut oak, white oak, and muscadine), a post oak leachate control and two leachate concentrations (C1 and C2), and three study replicates (R1, R2, and R3) of each concentration and species. Laboratory temperature and humidity conditions were kept at approximately 25 degrees Celsius and 60 percent relative humidity. A grow lamp was suspended approximately 3 feet above Petri dishes used for seed germination to ensure equal distribution of light. Light was turned on for 12 hours and off for 12 hours for one month to allow the selected species to sprout and grow. Controls were moistened with 3mL of distilled water, and replicates were moistened with 3 mL of the leachates. Radish seed were germinated by placing Whatman #1 filter paper in the bottom of Petri dishes and placing 20 radish seeds in each plate, moistened with either distilled water (controls) or leachates (replicates), and covered with Parafilm. Thirty chestnut oak and white oak acorns were placed in each bin (plastic container) on top of sterilized sand and moistened with the distilled water for controls and leachates for the replicates. Ten Muscadine seeds were placed in Petri dishes on Whatman #1 filter paper and moistened with distilled water (controls) and leachates (replicates). The germination rates were counted and sprout lengths measured using a micrometer and recorded at regular intervals (time periods T1, T2, T3, and T4). The effects of the leachates on percent germination and sprout length were analyzed using ANOVA, and the Tukey post-hoc analysis with a confidence level of 95 percent was used to test differences between individual time periods.

RESULTS AND DISCUSSION

Radish Sprouting Percentage and Mean Sprout Length

Radish germination varied between controls and leachates (fig. 1a) with a 95 percent germination rate (control) at time period T1 which slightly increased throughout the measurement period. Germination rates for C1 start at 75 percent at time period T1 measurement and increase to about 81 percent through time period T3 measurement. C2 starts 68 percent sprouting percentage at T1 and increases to about 71 percent through T3. These decreases demonstrate that radish sprouting percentage was inhibited by both leachate concentrations and that C2 inhibited the radish sprouting percentage more than C1.

Radish sprout length (fig. 1b) was measured through three time periods. The control was measured at 8mm in T1 and increased significantly to 35mm through T3. C1 started near 5mm at T1 and increased slowly to 20mm through T3. C2 also started at 5mm at T1 and increased to 19mm through T3. This indicates that C1 and C2 leachates inhibited radish sprout length as mean sprout length for the control was higher and increased more rapidly than the C1 and C2 radish seeds. If sprout lengths are inhibited in natural conditions, ability to compete and survive may be diminished. C2 inhibited the mean sprout length of radish the greatest.

ANOVA results were used to determine if a significant difference between the control and concentrations C1 and C2 during the same period (table 2). For control versus C1 at time period T2, the p-value was less than 0.05, indicating that at time period T2, a significant difference exists between the mean sprout lengths of the control and C1. For control versus C1 at time period T3, there was a significant difference between the mean sprout lengths of the control and C1. For control versus C2 at time periods T2 and T3 a significant difference occurs between mean sprout lengths between the control and C2 for time periods T2 and T3. There was no significant difference between C1 and C2 in sprouting percentage or mean sprout length growth.

Table 1—Results of understory species inventory for species on study plot in Marshall Forest

Species	Muscadine	%	Red Maple	%	Chestnut Oak	%	Pignut Hickory	%	Spotted Wintergreen	%	White Oak	%
Post Oak	30	67	3	7	1	2	6	13	5	11	0	0
Loblolly Pine	12	40	11	37	0	0	6	20	1	3	0	0
Chestnut Oak	12	24	6	12	8	16	3	6	21	42	0	0
White Oak	7	16	9	20	1	2	5	11	17	39	5	11

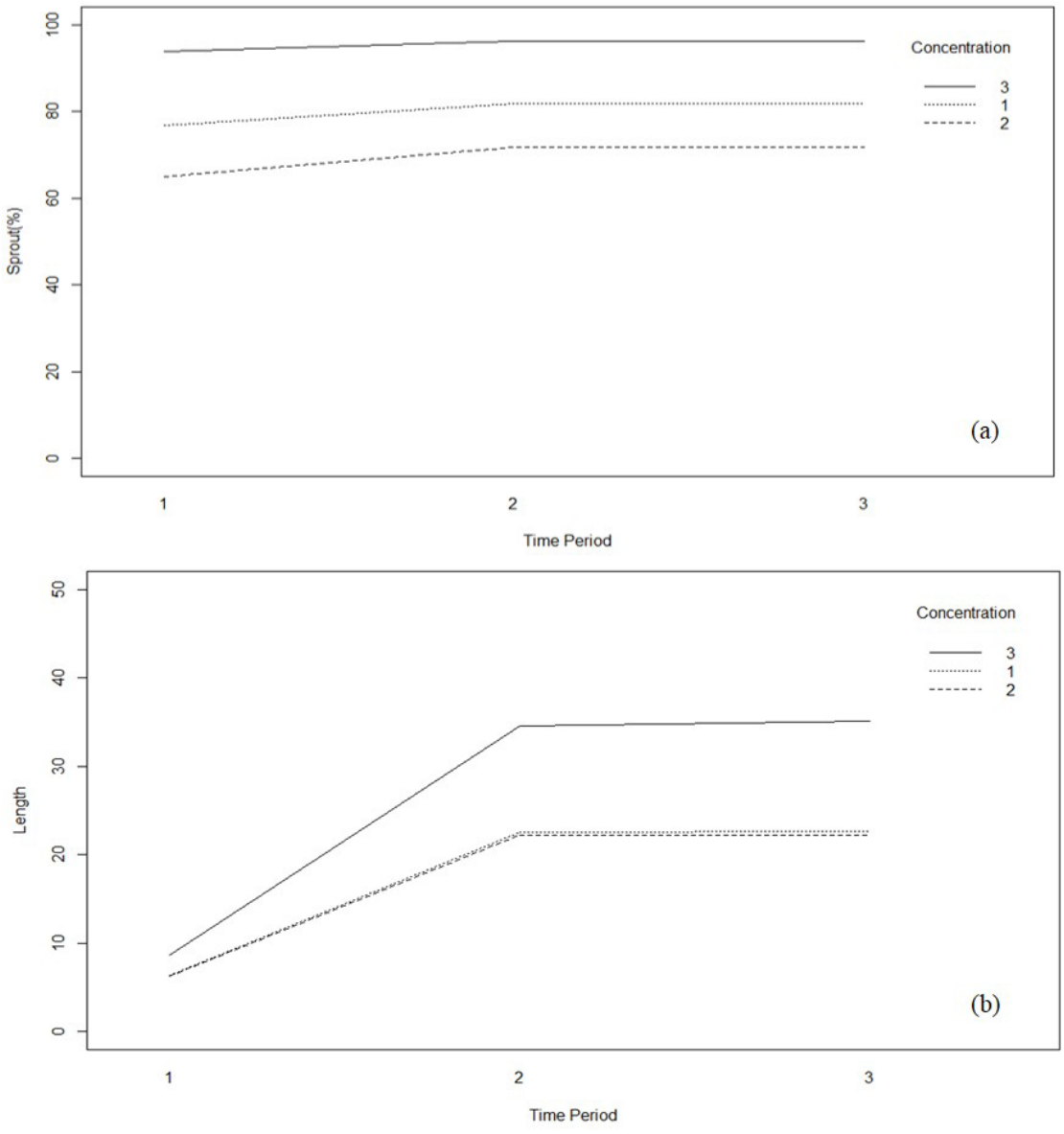


Figure 1—Radish germination (a) and sprout length (b). The start date was September 17, 2014; time period 1(T1) = September 19, 2014; time period 2 (T2) = September 24, 2014; time period 3 (T3) = September 30, 2014.

Table 2—ANOVA results for radish

Concentration	Time Period	P-value
Control	1 vs. 2	<0.05
Control	1 vs. 3	<0.05
C2	1 vs. 2	<0.05
C2	1 vs. 3	<0.05
Control vs. C1	2	<0.05
Control vs. C2	2	<0.05
Control vs. C1	3	<0.05
Control vs. C2	3	<0.05

Both germination and sprout length were inhibited by post oak leachate. This is not surprising, as germination inhibition was noted by Buehler (2010). Given that post oak is commonly planted for shade, these results may be of interest to those with vegetable gardens near post oak trees. Additional tests with other vegetable species (e.g., corn, beans, etc.) would be required.

Chestnut Oak Sprouting Percentage and Mean Sprout Lengths

The chestnut oak control was measured at 37 percent sprouting percentage at T1 and increased to 88 percent through T4. C1 was measured at 25 percent at time period T1 and increased slowly to 47 percent through T4. C2 was measured at 25 percent at time period T1 and increased to 61 percent through T4. These results indicate that C1 and C2 inhibited the sprouting percentage of chestnut oak, but C1 inhibited the sprouting percentage the greatest (fig. 2a).

The mean sprout length of chestnut oak was measured through four time periods (fig. 2b). The control measured 9mm at T1 and increased significantly 50mm through T4. C1 started at 11mm and increased to 31mm through T4. C2 started at 5mm and increased to 36mm through T4. This indicates that C1 inhibited the mean sprout length of chestnut oak greater than C2.

White Oak Sprouting Percentage and Mean Sprout Lengths

White oak controls started at 55 percent sprouting percentage in T1 and increased to 95 percent through T4 (fig. 3a). C1 started at 36 percent and increased to 55 percent throughout the measuring periods to T4. C2 started at 44 percent and increased to 77 percent through T4. These results indicate that C1 inhibited the sprouting percentage greater than C2. These results can also suggest that there is possibly a threshold of how much post oak allelochemicals can inhibit the selected species because doubling the concentration (C2) seems to not inhibit the sprouting percentage or mean sprout lengths more than C1.

The mean sprout lengths for the control started at 5mm at T1 and increased to 33mm through T4. C1 started at 10mm and increased to 21mm though T4. C2 started at 6mm and increased to 15mm through T4. This indicates that C2 inhibited the mean sprout lengths greater than C1 (fig. 3b).

For the ANOVA test, chestnut oak and white oak numbers were combined for mean sprout lengths (table 3). These results showed that there was a significant difference between control and C1 at T3 (p-value 0.0163) and T4 (p-value 0.0172). There was also a significant difference between the control and C2 at time periods T3 (p-value 0.0149) and T4 (p-value 0.0016). There was no significant difference between C1 and C2. It is worth noting that the differences in sprout length are not immediate. Figures 2b and 3b show that it takes approximately two weeks for significant differences between leachates to occur. Where these species (i.e., post oak and white oak) occur together, post oak allelopathic inhibition of germination or sprout length, could lead to a reduction of these species in forests.

Muscadine Sprouting Percentage

Muscadine germination rates for control seeds begin at 35 percent in T1 and slightly increased to 36 percent over the two measuring periods, T1 and T2 (fig. 4). C1 started at 35 percent in T1 and increased to 39 percent at T2. C2 started at 10 percent at T1 and increased to 18 percent at T2. These results are not consistent with the results of other species tested (white oak and chestnut oak), but could possibly be a result of the short measuring period. Mean sprout length could not be graphically displayed due to the small number of germinated seeds.

Overall, species (other than muscadine) exhibit a reduction in germination rates and sprout lengths when leachates were applied. It is also worth noting that in a cursory survey of Marshall Forest, where post oak occurs, there are several trees in relatively close proximity. Other associated species (e.g., hickories and red maple (*Acer rubrum*)) should be tested to see if this relationship holds for such species. The understory plant inventory indicates that red maple may be susceptible to post oak leachate. This would be useful as red maple is increasing in abundance throughout the forest, which is altering the forest composition on Marshall Forest as fire is suppressed.

CONCLUSION

A significant difference was observed for germination and sprout length between the post oak leachate

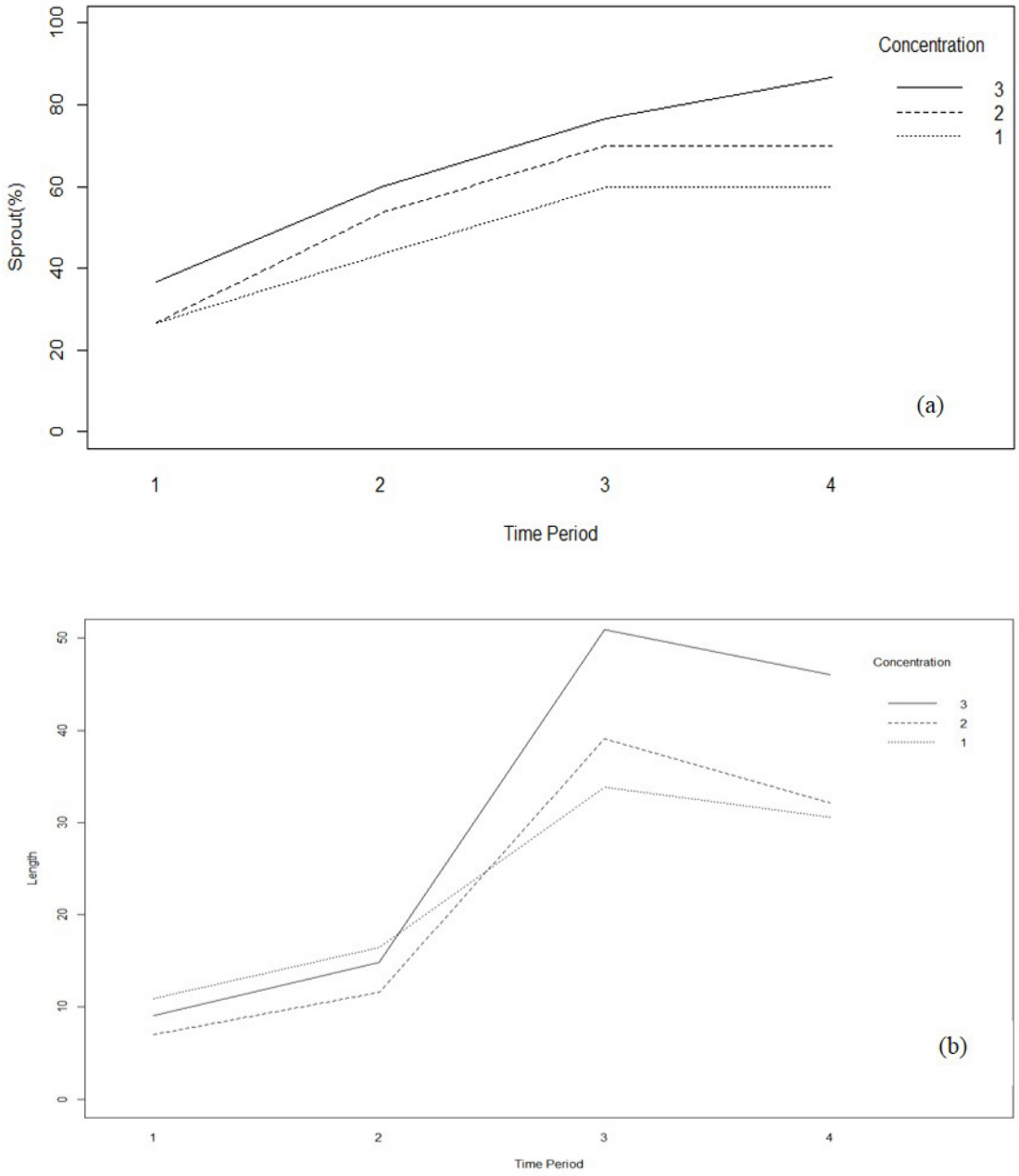


Figure 2—Chestnut oak germination (a) and sprout length (b). The start date was October 31, 2014; time period 1(T1) = November 3, 2014; time period 2 (T2) = November 5, 2014; time period 3(T3) = November 14, 2014; time period 4 (T4) = November 20, 2014.

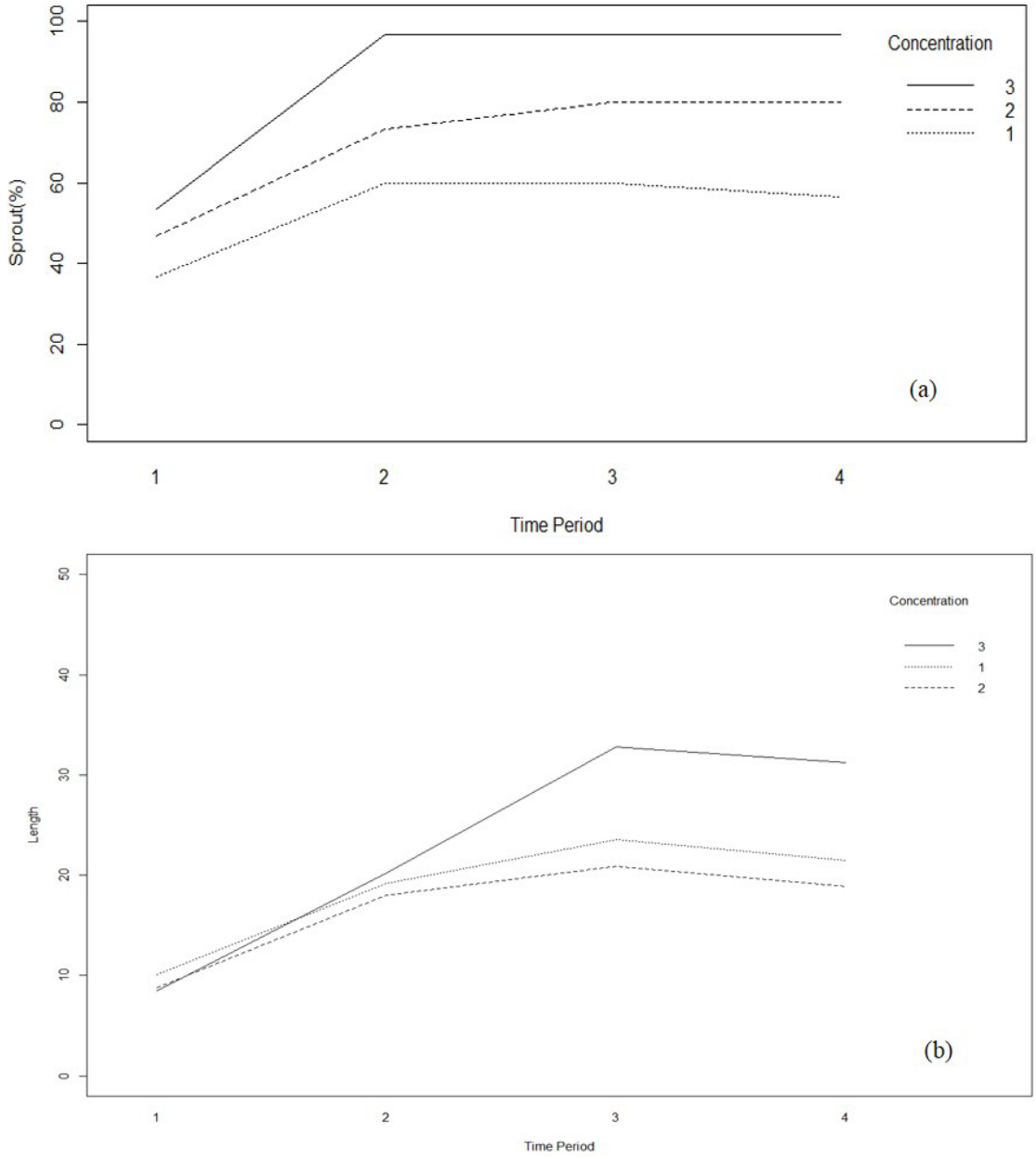


Figure 3—White oak germination (a) and sprout length (b). The start date was October 31, 2014; time period 1 (T1) = November 3, 2014; time period 2 (T2) = November 5, 2014; time period 3 (T3) = November 14, 2014; time period 4 (T4) = November 20, 2014.

Table 3—ANOVA results for oak species (Chestnut oak and white oak combined)

Concentration	Time Period	P-value
Control	1 vs. 3	<0.05
Control	1 vs. 4	<0.05
Control	2 vs. 3	<0.05
Control	2 vs. 4	<0.05
C1	1 vs. 3	<0.05
C1	1 vs. 4	<0.05
C2	1 vs. 3	<0.05
C2	1 vs. 4	<0.05
Control vs. C1	3	<0.05
Control vs. C2	3	<0.05
Control vs. C1	4	<0.05
Control vs. C2	4	<0.05

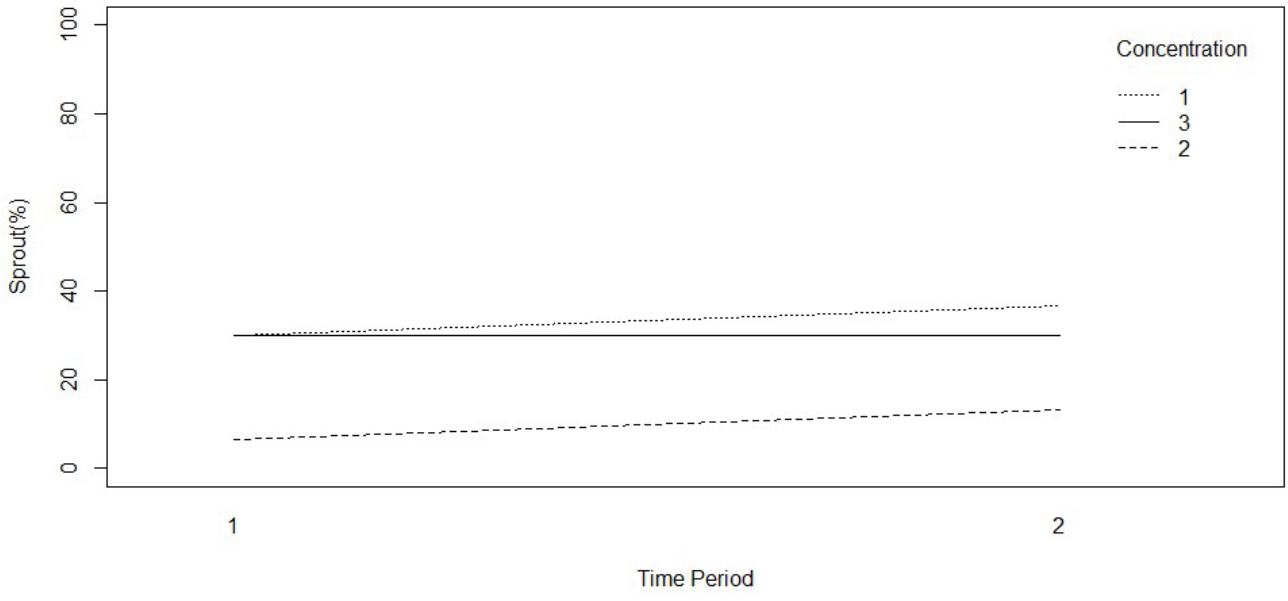


Figure 4—Muscadine germination. The start date was January 21, 2015; time period 1 (T1) = January 29, 2015; time period 2 (T2) = February 5, 2015.

control and different concentration levels (C1 and C2). There was no significant difference between leachate C1 and C2; that is, doubling the concentration of leaves in the leachate had no significant effect. Germination rates for C2 were only slightly higher than for C1. This suggests a fine balance where plants are sensitive to a threshold level of allelochemicals from leaves and increased levels have no additional effect. Marshall Forest is an old-growth forest and serves as a model for natural forest ecosystems in the southeastern United States. Additional species warrant study and may be useful for determining the influence of allelochemicals on forest compositional changes. These results demonstrate that post oak does exhibit allelopathic effects on the selected species by reducing germination rates and inhibiting mean sprout growth. Additional study will elucidate the allelochemical involved and its potential pathway for inhibiting plant germination and sprout growth.

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LITERATURE CITED

Buehler, C. 2010. Soil modification and potential allelopathy: an investigation into how the invasive *Casuarina Equisetifolia* L (Australian pine) modify their environment. Starkville, MS: Mississippi State University. 103 p. M.S. thesis.

Coder, K. 1999. Allelopathy in trees. FOR99-004. Athens, GA: University of Georgia Daniel B. Warnell School of Forest Resources Extension Publication. [Not paged].

Pérez-Corona M.E.; de las Heras, P.; Vázquez de Aldana, B.R. 2013. Allelopathic potential of invasive *Ulmus pumila* on understory plant species. Allelopathy Journal. 32: 101-112.

Rietveld, W.J. 1983. Allelopathic effect of juglone on germination and growth of several herbaceous and woody species. Journal of Chemical Ecology. 9: 295-309.

Starskey, J. 1990. *Quercus stellata.* In: Burns, R.M.; Honkala, B.H., tech. cords. Silvics of North America. Vol. 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 877.

Shortleaf

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UNDERPLANTED SHORTLEAF PINE SEEDLING SURVIVAL AND GROWTH IN THE NORTH CAROLINA PIEDMONT

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Abstract—A study was established in North Carolina to evaluate the viability of underplanting shortleaf pine (*Pinus echinata* Mill.) seedlings beneath a residual hardwood overstory as a method of reestablishing the shortleaf pine component to Central Appalachian Piedmont sites. Twenty-eight treatment plots were harvested to retain one of four residual overstory basal areas (RBA): 0, 15, 30, or 45 square feet per acre. Three shortleaf pine stock types were established within the RBA treatment plots; bareroot stock (BR), and containerized stock with small plugs (SP), and large plugs (LP). Overstory basal area affected survival only in the RBA0 plots which had the poorest survival for all three stock types over the first growing season. Seedling growth declined with increasing overstory basal area for all three stock types over the second growing season. Significant differences in percent survival were also noticed between the three stock types. The LP seedlings had the highest survival and the BR the lowest. Containerized seedlings achieved superior height and groundline diameter growth across all treatments but the differences were greatest between the LP and BR seedlings. Comparatively low survival in the RBA0 plots and the inverse relationship between overstory basal area and growth are attributed to gradients in overstory and understory competition levels and site harshness across the four RBA levels. The superior growth and survival of containerized seedlings is attributed to more intact root systems with higher root mass although we cannot rule out seed source differences. The results of this study suggest that underplanting may be a suitable regeneration option for the initial establishment of shortleaf pine on marginal Central Appalachian Piedmont sites. Further improvements in seedling survival and growth may be realized by planting containerized seedlings.

INTRODUCTION

Previous research suggests that underplanting shortleaf pine (*Pinus echinata* Mill.) seedlings beneath a residual hardwood overstory may be a viable regeneration option for some forest landowners. Many landowners consider the retention of some residual overstory basal area (RBA) to be less visually offensive than clearcut harvesting. Additionally, retained overstory can partially control woody and herbaceous competition and reduce the need for release herbicides (Jensen and others 2007). This regeneration approach may therefore be suitable for the rapidly developing Central Appalachian Piedmont Region where public opinion and changing landowner values can hinder the use of many traditional southern pine regeneration methods that include clearcut harvesting and herbicide applications. Unfortunately, research pertaining to this approach for shortleaf pine is sparse, and almost exclusively limited to the western portion of shortleaf pine's native range (Guldin and Heath 2001, Jensen and Gwaze 2007, Jensen and others 2007, Kabrick and others 2011).

Guldin and Heath (2001) found that bareroot shortleaf pine seedling survival was not significantly affected by RBA after three, five and seven growing seasons, but increasing RBA resulted in decreased height and groundline diameter (GLD) growth of seedlings in the Ouachita Mountains of Arkansas. Jensen and others (2007), Jensen and Gwaze (2007), and Kabrick and others (2011) found inverse relationships between residual overstory stocking and bareroot seedling growth on sites in Missouri. Kabrick and others (2011) also found that increasing overstory stocking marginally increased seedling survival.

In 2012, the North Carolina Department of Agriculture and Consumer Services Research Stations Division (NCDA&CS-RSD) and Mississippi State University applied several components of the underplanting studies from Arkansas and Missouri to a site on the North Carolina Piedmont. The goal of this study was to evaluate the effectiveness of underplanting to establish a shortleaf pine component in pine-hardwood stands in the Central Appalachian Piedmont. The specific objectives were to (1) evaluate the impact of RBA on survival and growth of underplanted seedlings and (2) evaluate differences in survival and growth between containerized and bareroot shortleaf pine planting stock.

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METHODS

Site

The study site is located on the NCDA&CS-RSD Umstead Research Farm in Durham County, NC (36° 9'25.75"N, 78°48'54.32"W). Elevations range from 434 to 486 feet along a ridge with less than 10 percent slope and well-defined east and west aspects. The site received an annual average of 47.8 inches of precipitation and has an average growing season of 194 +/- 14 days (Perry 1996, State Climate Office of North Carolina). The study site is in the Charlotte Slate Belt subsection of the Central Appalachian Piedmont Geological Province (North Carolina Geological Survey 1985, Bailey 1995). Lignum silt loam dominates the upper portions of the ridge and Helena sandy loam may be found on the lower hillslopes (Kirby 1976). Both soil types are moderately well-drained and rocky with a low percentage of organic material.

The forest cover of the study site prior to harvest consisted of a naturally-regenerated mixed upland hardwood-pine stand that developed following agricultural abandonment in the 1940s. The overstory was dominated by oak and hickory species including white oak (*Quercus alba* L.), northern red oak (*Q. rubra* L.), southern red oak (*Q. falcata* Michx.), post oak (*Q. stallata* Wangenh), black oak (*Q. velutina* Lam.), willow oak (*Q. phellos* L.), mockernut hickory (*Carya tomentosa* (Poir.) Nutt.), pignut hickory (*C. glabra* (Mill.) sweet), and red hickory (*C. ovalis* (Wangenh.) Sarg.). A number of other species occupied dominant or co-dominant overstory positions in limited portions of the stand, including yellow poplar (*Liriodendron* tulipifera L.), winged elm (*Ulmus alata* Michx.) red maple (*Acer rubrum* L.), sweetgum (*Liquidambar styraciflua* L.), Virginia pine (*Pinus virginiana* Mill.), loblolly pine (*P. taeda* L.), shortleaf pine and Eastern redcedar (*Juniperous virginiana* L.).

The midstory was dominated by winged elm, hickory, American hornbean (*Carpinus caroliniana* Walter), hop hornbeam (*Ostrya virginiana* (Mill.) K.), American holly (*Ilex opaca* Aiton), blackjack oak (*Q. marilandica* Muenchh.), and Eastern redcedar, and contained a minor component of American beech (*Fagus grandifolia* Ehrh.). Advanced regeneration within the stand was comprised primarily of winged elm, hickory species, Eastern redcedar, American holly and a limited quantity of white and post oaks. Herbaceous groundcover was sparse prior to harvest.

Procedures

Twenty-eight 0.33-acre circular RBA treatment plots were organized into seven replicated blocks and arranged across the site to account for variability in slope position, aspect, and soil type. Each block of four RBA treatment plots contained one randomly assigned replicate of each of four RBA treatments: zero (RBA0), 15 (RBA15), 30 (RBA30) and 45 (RBA45) square feet per acre. Seedlings of three different 1-0 shortleaf pine stock types were underplanted within a 0.10-acre circular seedling measurement plot originating from RBA treatment plot center. The three stock types included bareroot (BR), containerized seedlings with small plugs (SP) and containerized seedlings with large plugs (LP). Bareroot seedlings were grown using a Virginia seed source. The SP and LP seedlings were produced using North Carolina seed sources but we are unable to verify whether they were from the same seed source. The plugs of the SP seedlings measured 1.6 inches in diameter by 3.5 inches in depth. The LP seedlings plugs measured 1.5 inches in diameter by 4.75 inches in depth.

Overstory trees required to obtain the RBA targets within the treatment plots were selected based primarily on species and visual assessment of health. Tree location was also considered to ensure that RBA was evenly distributed across the plots. Healthy dominant and co-dominant oak and hickory species were targeted for retention although other species were retained where necessary to meet RBA targets and ensure appropriate overstory distribution. Treatment plots were operationally harvested to their assigned RBA targets in the summer and early fall of 2012. A broadcast burn was completed in November, 2012 to prepare the site for planting.

Seedlings were underplanted by hand in January and February of 2013. The rocky soils of the study site prevented a uniform planting spacing. Instead, each seedling measurement plot was divided into four equal quadrants. Nine each of the SP and BR were underplanted within each quadrant resulting in 36 each of these two stock types per seedling measurement plot. Limited seedling availability permitted only 20 to 22 of the LP stock to be planted in each plot using the same distribution method. Proper seedling storage, handling, and planting practices were followed during the reforestation process (Mexal 1992).

Measurements

Initial seedling height and GLD were measured and recorded in February, 2013. Seedlings were assigned a unique identification number and were tagged and flagged for future location and measurement. First year and second year seedling survival, GLD, and height were collected in September, 2013 and January and February, 2015, respectively. Initial seedling size measurements were subtracted from second year measurements to calculate seedling growth.

Statistical analysis

The four RBA levels and three stock type categories formed the treatments in this study. The plot mean survival, height, and GLD growth for each of these stock types represented response variables. A blocking factor accounted for differences in aspect, soil, slope and slope position across the site. Initial seedling height and GLD were included in the model as covariates, but removed from the analysis through backwards selection if they lacked significant effects. Mean percent survival, height growth and GLD growth were analyzed for treatment differences using analysis of covariance (ANCOVA) through a General Linear Model. The study therefore utilized a 4 x 3 factorial randomized complete block design that allowed for statistical control of the potential confounding variables associated with this study. Analysis was conducted with a significance level of α=0.05. Tukey's Honest Significant Difference was used to compare means when significant differences were detected among treatments.

Site Damage

The study site experienced several incidents of damage between establishment and the second growing season. A straight-line wind toppled overstory trees in several treatment plots in July, 2013. Ice damage in the winter of 2013/14 also eliminated several overstory trees. The RBA treatment plots were inventoried in the winter of 2013/14 and again in 2014/15 to account for the damage. Redheaded pine sawfly (*Neodiprion lecontei*) impacted approximately 29 percent of the underplanted seedlings during the late fall of the first growing season. Deer herbivory damaged approximately 30 percent of the seedlings during the winter following the first growing season. Seedlings were inspected in February 2014 to record whether or not they had been browsed or impacted by sawfly.

The biotic and abiotic damage to both the overstory trees and the seedlings within the study area necessitated substantial data filtering to remove confounded treatment levels and experimental units from the analysis. Treatment plots with RBA measurements more than 7.5 square feet outside of their target have been removed from the analysis. Plot mean height and GLD growth values calculated from fewer than five live and unaffected seedlings have also been excluded from the study. The biotic damaging agents were presumed to have had some effect on survival. Survival analysis was limited to data which was collected after the first growing season before the sawfly damage and deer browse occurred. Analysis on height and GLD growth has been applied to data collected after the second growing season that has been filtered of damaged seedlings. All statistical analyses were performed using SAS Enterprise Guide 7.1 (SAS Inc., Cary, NC).

RESULTS AND DISCUSSION

Results

We found significant differences in mean initial seedling height (fig. 1) and GLD (fig. 2) by stock type. BR seedlings had the tallest mean heights and the largest GLD at time of planting. LP seedlings were smallest in both measures and SP seedlings were intermediate between the two. There were no significant differences in seedling size within stock type by RBA treatments.

RBA and stock type had significant effects on percent seedling survival after one growing season (fig. 3). Survival was poorest for all three stock types in the RBA0 plots which lead to the significance of the RBA effect. There were no significant differences in survival between the RBA15, RBA30, and RBA45 treatment levels. LP seedlings had the highest survival (99 percent) followed by the SP seedlings (91 percent) and the BR seedlings (64 percent), respectively. Differences in survival among all three stock types were significant.

RBA and stock type had significant effects on seedling height (fig. 4) and GLD (fig. 5) growth after two growing seasons. Height and GLD growth were best under the lower RBA treatments of RBA0 and RBA15. There were no significant differences between those two treatments for either measurement. Likewise, there were no significant differences between the pair of higher RBA treatments of RBA30 and RBA45 where seedling height and GLD growth were poorest. The groups of low and high RBA treatments were significantly different from each other for GLD. A similar grouping of significant differences between high and low RBA treatments existed for height but the two intermediate mean height values by treatment, RBA0 and RBA30, were not significantly different from each other.

The LP seedlings had significantly greater mean height growth (1.7 feet) after two growing seasons than the SP (1.1 feet) or BR (0.9 feet) seedlings. The SP seedlings had marginally better mean height growth than the BR seedlings but the differences were not significant. The LP seedlings also had significantly better GLD growth (0.4 inches) than the SP (0.3 inches) and BR (0.2 inches) seedlings. The SP seedlings had significantly better GLD growth than the BR seedlings.

Discussion

Residual overstory basal area—The significant effect of RBA on survival was due primarily to the differences in survival between the RBA0 and RBA15 plots for both the SP and BR stock types. Retaining as little as 15 square feet of residual overstory basal area significantly improved survival, although the improvements were most operationally meaningful for the BR stock where mean survival increased by over 25 percent. Survival was not significantly improved by increasing overstory

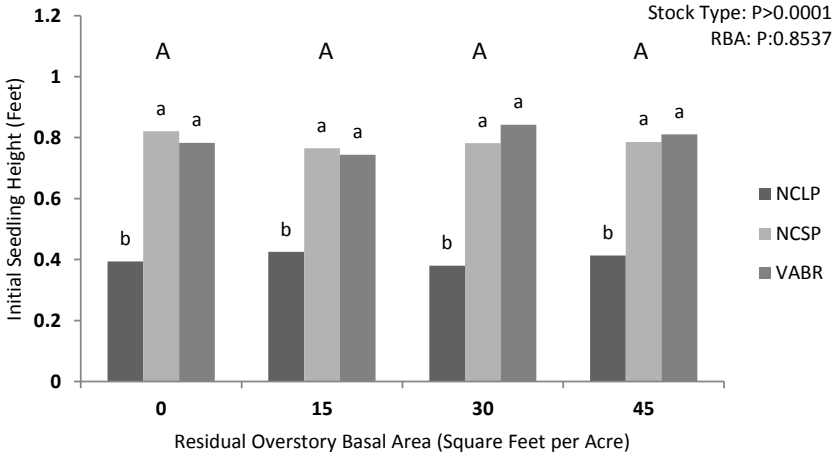


Figure 1—Initial seedling height by stock type and RBA. Same or shared letters indicate no significant differences at α=0.05.Different capital letters indicate significant differences among RBA treatments. Different lower case letters indicate significant differences between stock types within each RBA treatment.

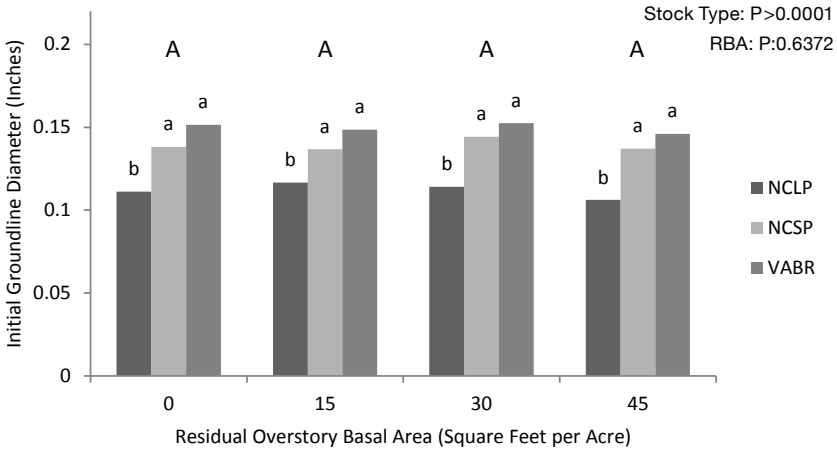


Figure 2—Initial seedling groundline diameter by stock type and RBA. Same or shared letters indicate no significant differences at α=0.05.Different capital letters indicate significant differences among RBA treatments. Different lower case letters indicate significant differences between stock types within each RBA treatment.

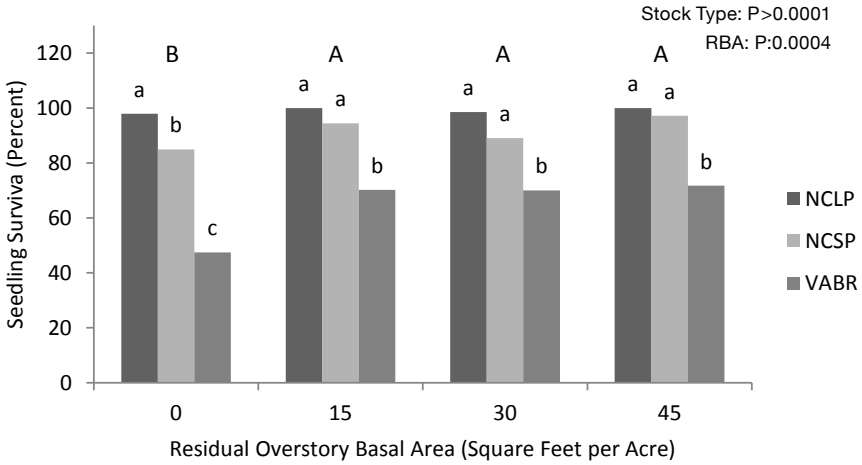


Figure 3—Percent seedling survival by stock type and RBA. Same or shared letters indicate no significant differences at α=0.05.Different capital letters indicate significant differences among RBA treatments. Different lower case letters indicate significant differences between stock types within each RBA treatment.

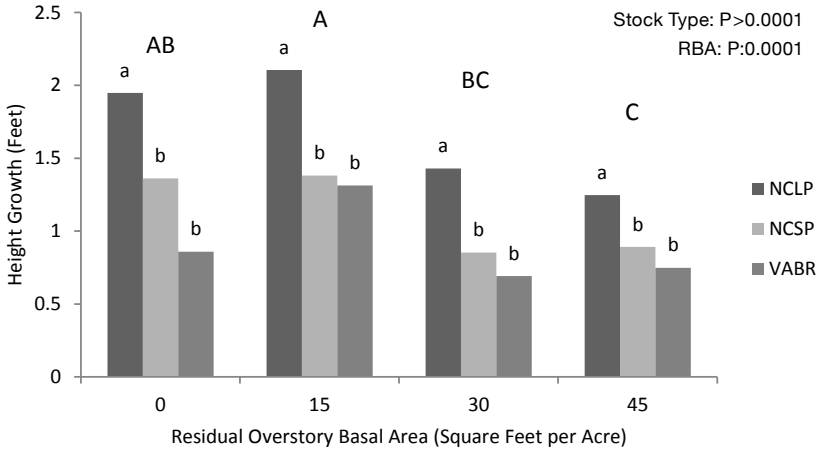


Figure 4—Seedling height growth after two growing seasons by stock type and RBA. Same or shared letters indicate no significant differences at α=0.05.Different capital letters indicate significant differences among RBA treatments. Different lower case letters indicate significant differences between stock types within each RBA treatment.

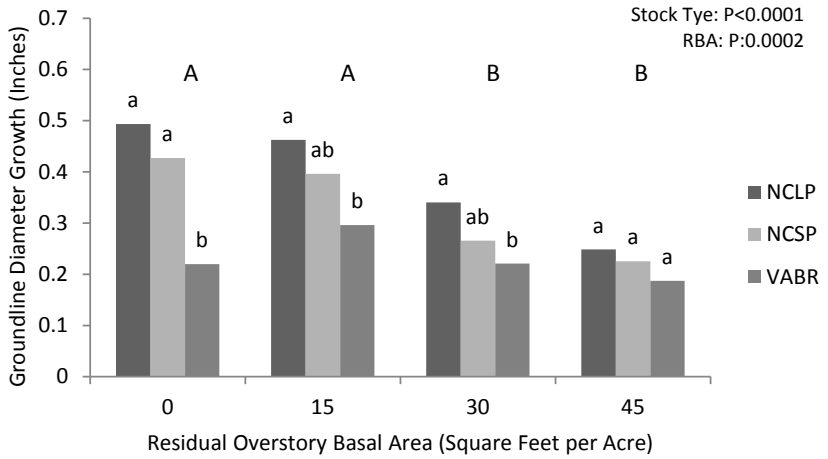


Figure 5—Seedling groundline diameter growth after two growing seasons by stock type and RBA. Same or shared letters indicate no significant differences at α=0.05.Different capital letters indicate significant differences among RBA treatments. Different lower case letters indicate significant differences between stock types within each RBA treatment.

stocking beyond 15 square feet of basal area for any of the three stock types. The positive yet mostly insignificant effect of overstory stocking on seedling survival is in line with the findings of Kabrick and others (2011). Unfortunately, Kabrick and others (2011) is the only past shortleaf pine underplanting study where seedlings were established after the stand had been thinned to different stocking levels and is therefore the only study to which we can make a comparison.

Our analysis of growth by RBA revealed a similar relationship of increasing residual overstory to decreasing seedling growth as has been found by others (Guldin and Heath 2001, Jensen and others 2007, Jensen and Gwaze 2007, Kabrick and others 2011). The lack of significant differences in height growth between the RBA0 and RBA30 treatment levels is likely do the very poor height growth of the BR stock in the RBA0 plots.

We speculated that the relationships we found between RBA and seedling survival and growth were functions of both overstory and understory competition as well as the gradient of site harshness present across the different RBA levels. Increasing levels of overstory shade resulted in decreasing levels of seedling growth. We attributed the inverse relationship between RBA and seedling growth in plots with at least 15 square feet of overstory basal area to the overstory shade. Site visits also revealed that overstory shade appeared to suppress competing vegetation. We observed high levels of competing vegetation in the RBA0 plots

that decreased dramatically as RBA increased. Fully exposed clearcut sites like the RBA0 plots can also have very harsh microclimatic conditions including higher soil and air temperatures near the ground level compared to sites with some overstory cover (Guldin and Barnett 2004). We surmised that the harsh microclimatic conditions and the high amount of competing vegetation in the RBA0 plots increased seedling mortality and retarded height growth when compared to the more moderate microclimates and reduced herbaceous competition in plots with at least 15 square feet of RBA.

Stock Type—The two containerized stock types achieved better survival than bareroot stock. Containerized stock exceeded 84 percent mean survival under all of the RBA treatment levels. On the other hand, survival for the BR seedlings ranged from approximately 47 percent to 72 percent. Our results on seedling survival differ from those of Barnett and Brissette (1989) and Gwaze and others (2006) who did not find significant differences in survival between 1-0 containerized and bareroot stock. However, Gwaze and others (2006) did find a stock type x seed family interaction in survival and growth indicating that certain families of shortleaf pine perform better as containerized stock while others perform better as bareroot and also indicated that a small sample size may have influenced their results.

Containerized stock also exhibited better height and GLD growth than bareroot stock after two growing seasons. The differences between containerized and bareroot stock were most pronounced in GLD growth with both containerized stock types significantly exceeding the bareroot stock in the lower RBA treatments. The LP seedlings exceeded the height growth of the BR seedlings across all treatments, but the SP seedlings actually grew similarly to the BR in plots with at least 15 square feet of RBA. Our finding that at least one containerized stock exhibited better growth than bareroot stock is in line with those of Brissette and Barnett (1989) and Barnett and Brissette (2004). They are in contrast those of Gwaze and others (2006) who did not find significant differences in growth between 1-0 bareroot and containerized stock in Missouri.

We attributed the significantly greater survival and growth of the containerized stock to the more intact root systems and higher root mass that containerized seedlings often have compared to bareroot seedlings (Barnett 1992). We speculated that LP seedlings would have had slightly higher root mass at the time of planting than the SP seedlings given their larger container size, potentially leading to better survival and improved growth. However, the extensive range of shortleaf has resulted in a strong relationship between climate at seed source and growth at the outplanted site (Schmidtling 2001). Without knowing the specific origin of the SP stock and already knowing that the BR and LP stock are from different orchard mixes, we cannot rule out that the differences in survival and growth by stock type might be a function of seed source.

Management Implications

Future research will be necessary to determine how long the benefits provided by limited levels of residual overstory basal area persist. The residual overstory trees will presumably continue to grow and eventually reach stocking levels under which this study shows that seedling growth will decline. Future research on this site will focus on how underplanted seedling survival and growth are affected by the continually changing and likely increasing levels of residual overstory basal area and understory competition.

CONCLUSIONS

The results of this study indicated that underplanting may be a suitable method for establishing shortleaf pine on marginal sites in the Central Appalachian Piedmont. They also showed that containerized shortleaf pine performed very well on such sites compared to bareroot stock. The first year survival and second year growth analysis indicated that retaining low levels of residual overstory basal area can improve early seedling survival and growth compared to clearcut plots where competing vegetation has not been controlled.

LITERATURE CITED

Bailey, R.G. 1995. Descriptions of Ecoregions of the United States. 2d ed., rev. and expanded (1st ed. 1980). Misc. Publ. No. 1391. Washington, DC: U.S. Department of Agriculture Forest Service. 108 P. with separate map at 1:7,500,000. Available online at: http://fs.fed.us/land/ecosysmgmt/index.html.

Barnett, J P. 1992. Production of shortleaf pine seedlings. In: Brissette, J.C.; Barnett, J.P., eds. Proceedings of the shortleaf pine regeneration workshop. Gen. Tech. Rep. SO-090. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 20-31.

Barnett, J.P.; Brissette, J.C. 2004. Stock type affects performance of shortleaf pine planted in the Ouachita Mountains through 10 years. In: Connor, K. F., ed. Proceedings of the 12th biennial southern silviculture research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 420-422.

Brissette, J.C.; Barnett, J.P. 1989. Comparing first year growth of bare-root and container plantings shortleaf pine half-sib families. In: Proceedings of the 20th southern forest tree improvement conference. Atlanta, GA: Southern Forest Tree Improvement Committee: 354-361.

Guldin, James M.; Barnett, James P. 2004. Microclimatic conditions after reproduction cutting in shortleaf pine stands in the Ouachita Mountains. In: Connor, K. F., ed. Proceedings of the 12th biennial southern silviculture research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 92-98.

Guldin, James M.; Heath, Gerald 2001. Underplanting shortleaf pine seedlings beneath a residual hardwood stand in the Ouachita Mountains: results after seven growing seasons. Res. Note SE-9. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 8 p.

Gwaze, D.; Melick, R.; Studyvin, C.; Hoss, G. 2006. Survival and growth of container and bareroot shortleaf pine seedlings in Missouri. In: Riley, L.E.; Dumroese, R.K.; Landis, T.D., tech. coords. 2006. National proceedings: forest and conservation nursery associations - 2005. RMRS-P-43. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 123-126.

Jensen, J.; Gwaze, D. 2007. Underplanting shortleaf pine at Coldwater Conservation Area in Missouri. In: Kabrick, J.M.; Dey, D.C.; Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 117-118.

Jensen, J.; Smith, C.; Johanson, M.; Gwaze, D. 2007. Underplanting shortleaf pine in the Missouri Ozarks. In: Kabrick, J.M.; Dey, D.C.; Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 112-116.

Kirby, R.M. 1976. Soil Survey of Durham County, North Carolina. Washington, DC: U.S. Department of Agriculture Soil Conservation Service. 80 p.

Kabrick, J.M.; Dey, D.C.; Shifley, S.R.; Villwock, J.L. 2011. Early survival and growth of planted shortleaf pine seedlings as a function of initial size and overstory stocking. In: Fei, S.; Lhotka, J.M.; Stringer [and others], eds. Proceedings to the 17th central hardwood forest conference. Gen. Tech. Rep. NRS-P-78. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 277-286.

Mexal, J M. 1992. Artificial regeneration of shortleaf pine: put it all together for success. In: Brissette, J.C.; Barnett, J.P., eds. Proceedings of the shortleaf pine regeneration workshop. Gen. Tech. Rep. SO-090. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 172-186.

North Carolina Geological Survey. 1985. Geologic map of North Carolina: North Carolina Geological Survey, General Geologic Map. Scale 1:500000. http://portal.ncdenr.org/web/lr/1985-state-geologic-map. [Date accessed: May 5, 2015].

Perry, K. 1996. Average growing season for selected North Carolina locations. North Carolina Cooperative Extension Service Horticulture Information Leaflet. http://content.ces.ncsu.edu/average-growing-season-for-selected-north-carolina-locations/. [Date accessed: May 5, 2015].

Schmidtling, Ronald C. 2001. Southern pine seed sources. Gen. Tech. Rep. SRS-44. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 25 p.

State Climate Office of North Carolina, NC State University. 2015. CRONOS [internet database]. http://www.nc-climate.ncsu.edu/cronos/normals.php?station=311285. [Date accessed: May 5, 2015].

AGE AND SIZE COMPARISONS OF REGENERATING SHORTLEAF PINE SEEDLINGS BURNED MULTIPLE TIMES IN ECOSYSTEM RESTORATION AREAS

David C. Clabo, James M. Guldin, and Wayne K. Clatterbuck[1]

Abstract—Shortleaf pine (*Pinus echinata* Mill.) ecosystem restoration has been a major management goal in the Ouachita National Forest since the early to mid-1990s. Restoration efforts have focused on periodic prescribed burning and thinning operations to restore disturbance dependent vegetation communities suitable for the recovery of the endangered red-cockaded woodpecker, as well as other species able to utilize open forests and woodlands. Questions exist about how frequent burning will affect recruitment of shortleaf pine regeneration into larger age and size classes. Determining shortleaf pine's sprouting capabilities in response to repeated periodic prescribed burns could aid managers in recruitment of new age classes, while still promoting fire-dependent vegetation communities. The three stands in this study have undergone restoration activities for the past 6 to 12 years. The objectives of this study were: (1) to compare shortleaf pine root and shoot age, weight, and volume values, as well as root diameter, basal diameter, sprout height growth, live foliage weight, regeneration densities, and sprout production parameters in three analogous stands, and (2) develop regression equations for predicting sprout height following a burn from seedling root characteristics. Root ages were older than stem ages, and there were statistical differences in root ages indicating that the majority of seedlings and saplings regenerated following the same event. Seedlings and saplings in the stand that had been burned fewer times with a lower overstory basal area were characterized by larger morphological characteristics than seedlings in stands with greater burn frequencies for most variables.

INTRODUCTION

Restoration of shortleaf pine (*Pinus echinata* Mill.) communities has gained momentum over the last several years across the Ouachita and Ozark Highlands of Arkansas, Missouri, and Oklahoma. Historically, fire has been an important anthropogenic disturbance in the Ouachita Highlands. Fire return intervals across the area prior to European settlement around 1820 were 7-20 years, from 1820-1920 2 to 4 years, and from 1920-2000 4 years or longer, but this estimate is very area dependent with some locations having 50-year plus return intervals (Guyette and others 2006, Johnson and Schnell 1985, Stambaugh and Guyette 2006). Efforts to restore shortleaf pine-bluestem ecosystems on the Ouachita National Forest (NF) began in the region approximately 20 years ago with a primary goal of restoring the endangered red-cockaded woodpecker, but a variety of species of flora and fauna benefit from this work. Strong timber markets in the region promote restoration treatments such as midstory removals and prescribed burning because a portion of the income from harvests are allowed to fund the restoration treatments through the Knutson-Vandenberg Act of 1933 and the National Forest Management Act of 1976 (Bukenhofer and Hedrick 2013, Bukenhofer and others 1994, Guldin and others 2004, Hedrick and others 2007). This restoration work is scheduled for roughly 268,000 acres, or 20 percent, of the 1.8-million acre Ouachita NF, and it is estimated that roughly 50,000 acres are fully restored to date (Hedrick and others 2007, Zhang and others 2012).

Shortleaf pine seedlings regenerating in many of these restoration areas are damaged by prescribed burning every two to three years on average and by logging activity during thinnings (Lilly 2010). The species' ability to sprout may assist in the survival of a young cohort after a disturbance, but little is known about the number of times seedlings can resprout after repeated burns. One year-old seedlings achieve variable survival rates of about 40-90 percent or better after late dormant season burns, but burn timing, intensity, and seedling age can drastically alter survival rates (Cain and Shelton 2000, Clabo 2014, Little and Somes 1956, Shelton and Cain 2002). Older seedlings should have higher growth and survival rates compared to younger seedlings due to thicker bark, larger, more developed basal crooks that contain more dormant buds, and larger root systems

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based on a collection of past studies and observations (Cain and Shelton 2000, Lilly and others 2012, Little and Somes 1956).

Knowledge of how time since a burn and number of burns affect seedling survival and growth is ambiguous at best. Early in the twentieth century, Mattoon (1915) reported that seedlings can resprout at least three times from three individual disturbances even though the type of disturbance and survival rates following the disturbance(s) were not identified. The same account states that shortleaf pine seedlings can completely make up for height lost in a disturbance within 2 to 4 years. In pine-bluestem restoration areas of Arkansas, seedlings have resprouted 6-8 times in association with each successive burn. Some of these seedlings have had a 3.1 inch diameter root with a stem only 3.2 feet tall, which indicates a much older root system than stem (Lilly 2010). The previous findings were observations and were not carried out with an organized study design in areas with varying frequencies and intervals of burn occurrences. A non-replicated study in New Jersey in the 1930s found that 46 percent of shortleaf seedlings can produce ¼ to ¾ of their pre-disturbance height in less than a year following a late spring burn and 86 percent can produce ¾ of their pre-burn height or more within 2 years (Moore 1936).

Several important questions remain concerning shortleaf pine regeneration and sprouting in restoration areas. The first is whether managers must interrupt cyclic prescribed burning in order for saplings to grow large enough to withstand burns and develop into merchantable size classes. The second question is whether seedlings can survive successive burns and reliably resprout. Therefore, the objectives of this study are (1) to compare shortleaf pine root and shoot age, weight, and volumes, as well as root diameter, basal diameter, sprout height growth, live foliage weight, regeneration densities, and sprout production parameters in three analogous stands with a differing number of occurrences and time lapses since prescribed burning last occurred, and (2) develop regression prediction equations for each stand for sprout height following a burn from a variety of seedling root characteristics.

METHODS

This study was conducted on the Ouachita NF on the Poteau/Cold Springs Ranger District in Scott County, Arkansas. The Ouachita Mountains are characterized by east to west oriented ridges with broad U-shaped valleys. Elevations range from 500 to 2,700 feet (Hedrick and others 2007). Three stands that had received varying amounts and intensities of restoration treatments, such as midstory removals, regeneration harvests, thinnings, and prescribed burns for the last 6 to 12 years were chosen for this study. All three stands were within 6 miles of one another and on similar landforms. Shortleaf pine was the dominant overstory species in all three stands.

Inventories of the three stands were conducted on July 22-24, 2013. The first stand was located the farthest east of the three stands and was located on the Buffalo Creek Road (34°48'31.44"N 94°02'21.95"W) on a south aspect at an upper slope position. The soils in this stand are typic hapludults and lithic dystrudepts consisting of the Carnasaw Series, Carnasaw-Sherless Complex, and Carnasaw-Sherless-Clebit Complex types. The site index for shortleaf pine on these sites is 60 to 65 feet at base age 50 years (Vodrazka 1998). Prescribed burns were completed on 29 August 2002, 25 March 2006, 30 June 2009, and 22 April 2013. The second stand was located to the west of stand one along the Buffalo Creek Road and had a similar aspect and slope position as stand one. A commercial thinning was conducted in spring 2001 and left 60-70 square feet per acre of basal area. Prescribed burns were completed on 15 March 2003, 27 February 2007, 19 March 2010, and 8 September 2011. Stand 3 was located farthest west of the three stands adjacent to the Boles Motorway (34°48'46.11"N 94°09'42.16"W). This stand had a similar aspect, slope position, and contained similar soils as stands 1 and 2. A shelterwood harvest was completed on stand 3 in July 2007, leaving about 30 square feet of basal area per acre. Prescribed burns were completed on 4 March 2010 and 27 February 2012.

Sampling grids were originated at a random point located at least one chain from a stand boundary. Plots were located every two chains along a predetermined azimuth perpendicular to the prevailing elevation gradient in the stand. In between lines separated plots by two chains resulting in a 12x6 chain grid totaling 18 plots per stand. Each sampling location was a 1/100th acre circular plot. All shortleaf pine seedlings less than or equal to 4.5 feet tall were counted to estimate per acre seedling densities. The closest seedling to plot center (regardless if the seedling was located in the 1/100th acre plot) was measured for height, basal diameter, and the number of sprouts. A note was recorded if the seedling had a dead stem still present alongside the live stem(s). This seedling was then removed from the ground with a shovel in a manner so as to preserve as much of the root system as possible.

The extracted seedlings (N=54), were returned to the lab for additional measurements. All needles were removed from the seedlings in order to facilitate stem and root volume measurements. Needles from three random seedlings in each stand were kept and oven dried to a constant weight in a VWR Scientific 1380 FM forced air oven for 48 hours at 97 degrees F and then weighed. Taproot length and longest lateral root length were measured to determine taproot and lateral root length

thresholds for root volume measurement. A threshold was set because not all of the root system could be extracted in many instances because of the rocky soil conditions at some plot locations. Average length and standard deviation statistics were used to select the threshold lengths. The threshold length was set at 5.1 inches for taproots and lateral roots. Root systems were then severed from the stems for volume determination using the water displacement technique outlined in Burdett (1979). Volume determinations were done only for live stems and roots.

Stems and roots from stands 1 and 2 were oven dried to a constant weight for 48 hours at 124 degrees F. Stems and roots from stand 3 were oven dried to a constant weight for 72 hours at 140 degrees F due to their larger average size. Stems and roots were then weighed to the nearest ounce. Root diameter was measured and marked at the widest point of the basal crook. Roots and stems were aged following volume and weight measurements. All roots were examined prior to cutting for aging to determine how well the basal crook was developed. Well-developed basal crooks were those where the angle of deflection from upright on the stem was 45 to 90 degrees (Will and others 2013). Forty-nine of the 54 seedlings returned to the lab had well-developed basal crooks. The five seedlings that did not have well developed crooks all had nearly vertical taproots, which may indicate intermediate morphological traits that occur when hybridization with loblolly pine occurs (Tauer and others 2012). For seedlings that had well-developed basal crooks, cuts were made at the widest point of the basal crook and at the point below the basal crook where the taproot turns vertical again. For seedlings without a well-developed basal crook, the root was cut at the widest point of the taproot. Stems were aged just above the root collar. Sections were then sanded using a Delta® Shopmaster belt/disc sander with 320 and 400 grit sandpaper until rings were clearly visible. Sections were examined and

Table 1—ANOVA means and standard errors in parentheses are presented for each shortleaf pine seedling variable by stand for the shortleaf pine restoration study on the Ouachita National Forest in Arkansas, 2013. A variable name followed by an "*" indicates significant differences among stands

Stand	Root Volume *(Fluid Ounces)**	Stem Volume *(Fluid Ounces)**	Stem Weight *(Ounces)**	Root Weight *(Ounces)**	Basal Crook Widest Point Average *(Inches)**
One	0.8 (+/-0.21)	0.06 (+/-0.017)	0.05 (+/-0.02)	0.68 (+/-0.17)	0.72 (+/-0.08)
Two	0.62 (+/-0.15)	0.23 (+/-0.06)	0.17 (+/-0.05)	0.45 (+/-0.12)	0.78 (+/-0.09)
Three	3.06 (+/-0.81)	0.77 (+/-0.21)	0.67 (+/-0.2)	2.02 (+/-0.56)	1.26 (+/-0.15)

Table 1 Continued—ANOVA means and standard errors in parentheses are presented for each shortleaf pine seedling variable by stand for the shortleaf pine restoration study on the Ouachita National Forest in Arkansas, 2013. A variable name followed by an "*" indicates significant differences among stands

Stand	Stem Age *(Years)**	Root Age Average *(Years)*	Root Age at Widest Point of Basal Crook *(Inches)*	Root Age Below the Basal Crook *(Inches)*	Sprout Number
One	1 (+/-0.08)	9.82 (+/-0.71)	10.35 (+/-0.74)	10.09 (+/-0.71)	10.89 (+/-2.3)
Two	3.71 (+/-0.27)	7.94 (+/-0.6)	9.06 (+/-0.64)	8.56 (+/-0.57)	3.42 (+/-0.74)
Three	4.58 (+/-0.35)	9.73 (+/-0.76)	10 (+/-0.78)	9.87 (+/-0.76)	10.71 (+/-2.41)

Table 1 Continued—ANOVA means and standard errors in parentheses are presented for each shortleaf pine seedling variable by stand for the shortleaf pine restoration study on the Ouachita National Forest in Arkansas, 2013. A variable name followed by an "*" indicates significant differences among stands

Stand	Tallest Sprout *(Feet)**	Basal Diameter *(Inches)**	Needle Weight *(Ounces)**	Seedlings Per Acre*
One	0.87 (+/-0.1)	0.1 (+/-0.02)	0.21 (+/-0.08)	74.5 (+/-47.6)
Two	1.64 (+/-0.18)	0.24 (+/-0.03)	0.53 (+/-0.2)	11.2 (+/-7.7)
Three	2.43 (+/-0.27)	0.46 (+/-0.06)	0.73 (+/-0.27)	4099.5 (+/-2750.3)

aged using a magnifying glass or a Fisher Scientific binocular microscope.

Multivariate analysis of variance (MANOVA) was used to test for differences among the following dependent variables: stem volume, root volume, stem weight, root weight, basal crook average width, stem age, root age at the widest point of the basal crook, average root age, tallest sprout height, and sprout number. A MANOVA was used to reduce the likelihood of Type I errors that can occur when using many univariate analysis of variance (ANOVA) analyses by themselves. MANOVA also takes into consideration the correlation of closely related variables. ANOVA was used to test for stand differences if the MANOVA found significant differences initially in the dependent variables. In addition, a one-way ANOVA was used to test for differences in needle weights and seedling density per acre among the three stands. Tukey mean separation (alpha=0.05) was used for all ANOVA analyses. Log transformations were completed as necessary for non-normality issues and back transformed estimates were reported. Multiple linear regression was used to test which root variables were the best predictors of sprout height in seedlings that had been topkilled. Tested variables included: taproot length, taproot diameter, largest lateral root diameter, number of lateral roots, and longest lateral root length. All analyses were completed using SAS 9.4.

RESULTS

The four MANOVA tests indicated significant differences among the three stands. Wilk's Lambda ($p<0.0001$), Pillai's Trace ($p<0.0001$), Hotelling-Lawley Trace ($p<0.0001$), and Roy's Greatest Root ($p<0.0001$) were all significant, thus individual variable ANOVA tests were conducted with stand as a fixed factor. Results are presented in Table 1. Stem volume ($p<0.0001$) and root volume ($p=0.0002$) both displayed significant differences across stands, with seedlings in stand three having much greater averages than stands 1 or 2. Stand 2 had the smallest root volume average with 0.62 fluid ounces displacement, whereas stand 1 had the smallest stem volume average with 0.06 fluid ounces displacement. Stem ($p<0.0001$) and root weight ($p=0.0008$) were significantly different as well. Again, stand 3 had the greatest average weights. Stand 1 had the smallest average stem weight (0.05 ounces), and stand 2 had the smallest average root weight (0.45 ounces). There were significant differences among stands for the widest average point of the basal crook ($p=0.0031$). Stand 3, displayed the greatest average width at 1.26 inches, while stands 1 and 2 were statistically similar. Stem age significantly differed among stands ($p<0.0001$). Stand 1 stems averaged one-year-old making them the youngest on average. Stands 2 and 3 were statistically similar and averaged 3.7 and 4.6 years-old on average.

There were no significant differences in average root age ($p=0.1621$), root age at the widest point of the basal crook ($p=0.3644$), or root age below the basal crook ($p=0.059$). Root ages for all three variables ranged from 7.9 to 10.3 years old across stands. Sprout number displayed significant differences among stands ($p<0.0004$). Stands 1 and 3 were statistically similar (10.89 and 10.71 sprouts per seedling), but stand 2 differed in sprout production with 3.42 sprouts per seedling on average. Sprout height was statistically different across the three stands ($p<0.0001$). Stand 3 had an average sprout height of 2.43 feet, stand 2, 1.64 feet, and stand 1, 0.87 feet. Significant differences in average basal diameter were found as well ($p<0.0001$). Stand 1 had the smallest basal diameters on average at 0.1 inches, followed by stand 2 at 0.24 inches, and finally stand 3 at 0.46 inches. There were no statistical differences in average needle weights across stands ($p=0.1161$). There were significant differences in seedling densities across the three stands ($p<0.0001$). Stands 1 and 2 were statistically similar (74.5 and 11.2 seedlings per acre), whereas stand 3 had a much greater density of 4099.5 seedlings per acre.

The multiple linear regression results for predicting sprout height in feet following a burn by taproot length, lateral root number, longest lateral root length, largest lateral root diameter, and taproot diameter had different results for stand 1 as compared to stands 2 and 3 which were very similar. For stand 1, largest lateral root diameter and taproot diameter did not contribute to prediction of sprout height and were thus removed from the model (table 2). These three variables predicted 84 percent of sprout height differences. Root variables from stand 1 predicted the following equation for sprout height:

(1) Sprout Height = 0.25 + 0.024a + 0.024b + 0.054c
where a=taproot length in inches, b=number of lateral roots in inches, and c=longest lateral root length in inches.

Regression parameters for stand 1 are found in Table 2. Largest lateral root diameter was the only significant prediction variable for stand 2. This variable predicted 56 percent of differences in sprout height. This root variable produced the following regression equation for stand 2 for prediction of sprout height:

(2) Sprout Height = -0.0409 + 2.63a
where a=largest lateral root diameter in inches.

Largest lateral root diameter was also the only variable for stand 3 seedlings that contributed to a prediction of sprout height following a burn (table 2). This variable predicted 41 percent of the differences in sprout height.

Table 2—Multiple regression results for predicting dominant shortleaf pine sprout height in stands 1, 2, and 3 for the shortleaf pine restoration study on the Ouachita National Forest in Arkansas, 2013

Stand 1: (R^2=0.84)	Parameter Estimate	SE	t-value	p-value	Partial R-Squares
Intercept	0.25182	0.09176	2.74	0.0158	—
Taproot Length	0.02391	0.01119	2.14	0.0507	24.607
Lateral Root Number	0.02422	0.01031	2.35	0.0339	28.293
Lateral Root Length	0.05359	0.01078	4.97	0.0002	63.813
Stand 2: (R^2=0.56)					
Intercept	-0.0409	0.144	-0.34	0.74	—
Largest Lateral Root Diameter	2.63	0.585	4.5	0.0004	—
Stand 3: (R^2=0.41)					
Intercept	1.35	0.46	2.92	0.01	—
Largest Lateral Root Diameter	3.34	0.99	3.37	0.0039	—

This root variable produced the following equation for stand 3 for prediction of sprout height:

(3) Sprout Height = 1.35 + 3.34a
where a=largest lateral root diameter in inches.

MANAGEMENT IMPLICATIONS AND CONCLUSIONS

Past research has shown that prescribed fire is the cheapest and least intensive site preparation or intermediate operation to achieve suitable shortleaf pine regeneration densities (Yocom and Lawson 1977). The fire return interval averaged across the three stands in this study was just over three years. The lack of significant differences among root ages indicate that the majority of seedlings regenerated around the same time before most of the burns associated with restoration activities began. The younger stem ages in comparison to the roots and the presence of a dead, charred stem on 96 percent of the seedlings sampled across stands indicates that prescribed burns are topkilling the majority of seedlings each time they occur. The multiple linear regression equations indicate that fewer root parameters are important for predicting sprout height in stands that have been burned longer ago than more recently. In addition, stand 3, which had received two fewer burns than stands 1 or 2 when this study was conducted, tended to have larger seedling attributes across many of the variables that were measured even though it had the second shortest time lapse since a burn. One conflicting factor that may account for this finding is that stand 3 is past the seed cut of a shelterwood regeneration harvest, whereas the other two stands are still in late-rotation mature stand conditions; the larger size of saplings in stand 3 is thus not an unexpected finding. Those saplings are on their way to becoming a second age cohort in that regenerating stand. These findings indicate that burning intervals in restoration areas will have to be more variable until seedlings and saplings reach minimum age and size thresholds as suggested by Guldin (2007), Stambaugh and others (2007), and Walker and Wiant (1966) to avoid topkill. Walker and Wiant (1966) reported that saplings 2-6 years old can survive a moderate intensity burn if crown scorch is less than 70 percent, basal diameter is greater than or equal to 0.5 inches, and height is greater than or equal to 5 feet. Competition from hardwoods would likely not be a problem with slightly longer burn intervals as shortleaf pine can typically compete successfully with or outcompete hardwoods around ages 5-7 on shortleaf sites in the region (Cain 1991).

The much lower regeneration densities per acre in stands 1 and 2 as compared to stand 3 indicate that more periodic prescribed burns result in fewer regenerating seedlings and saplings per acre, but not necessarily overstory density. Stands 1 and 2 would be considered extremely understocked according to most guidelines (e.g. Blizzard and others 2007), indicating that some type of change in management will be necessary in order to achieve necessary seedling densities and stocking prior to the end of overstory timber rotations. Past research has shown newly established seedlings can persist under a partial or full overstory for a period of time with limited effects on

survival, but height growth will be negatively affected (Kabrick and others 2011, Shelton 1995). Other research by Fan and others (2012) on periodic prescribed burning in mixed pine-hardwood forests in Missouri has shown that shortleaf pine seedling and sapling mortality is high compared to associated species as indicated by the results in this study.

Thus, when the time eventually comes to regenerate stands similar to stands 1 or 2 with the shelterwood method, foresters should inventory the shortleaf advance growth to see if the seedling bank of shortleaf seedlings and saplings is sufficient to regenerate the stand. If not, some degree of reliance on seedfall from the seed trees in the shelterwood will still be needed to fully stock the new age cohort. This study and others indicate that after a good regeneration year burning should stop in restoration areas until seedlings reach adequate sizes and ages to survive burns.

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LITERATURE CITED

Blizzard, E.M.; Henken, D.; Kabrick, J.M. [and others]. 2007. Shortleaf pine abundance and growth in pine-oak stands in the Missouri Ozarks. In: Kabrick, J.M.; Dey, D.C.; Gwaze, D. eds. Shortleaf pine restoration and ecology in the Ozarks: Proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 138-146.

Bukenhofer, G.A.; Hedrick, L.D. 2013. Shortleaf pine/bluestem renewal. U.S. Department of Agriculture Forest Service, Ouachita National Forest, Arkansas. http://www.fs.usda.gov/detailfull/ouachita/home/?cid=fsm9039689&width=fulll. [Date accessed: March 18, 2013].

Bukenhofer, G.A.; Neal, J.C.; Montague, W.G. 1994. Renewal and recovery: shortleaf pine/bluestem grass ecosystem and red-cockaded woodpeckers. In: Trauth, S. ed. Proceedings of the Arkansas Academy of Science. Monticello, AR: Arkansas Academy of Science: 243-245.

Burdett, A.N. 1979. A nondestructive method for measuring the volume of intact plant parts. Canadian Journal of Forest Research. 9(1): 120-122.

Cain M.D. 1991. Hardwoods on pine sites: competition or antagonistic symbiosis? Forest Ecology and Management. 44: 147-160.

Cain, M.D., M.G. Shelton. 2000. Survival and growth of *Pinus echinata* and *Quercus* seedlings in response to simulated summer and winter prescribed burns. Canadian Journal of Forest Research. 30(11): 1830-1836.

Clabo, D.C. 2014. Shortleaf pine sprout production capability in response to disturbance. Knoxville, TN: University of Tennessee. 76 p. M.S. thesis.

Fan, Z.; Ma, Z.; Dey, D.C.; Roberts, S.D. 2012. Response of advance reproduction of oaks and associated species to repeated prescribed fires in upland oak-hickory forests, Missouri. Forest Ecology and Management. 266: 160-169.

Guldin, J.M. 2007. Restoration and management of shortleaf pine in pure and mixed stands – science, empirical observation, and the wishful application of generalities. In: Kabrick, J.M.; Dey, D.C.; Gwaze, D. eds. Shortleaf pine restoration and ecology in the Ozarks: Proceedings of a Symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service. Northern Research Station: 47-58.

Guldin, J.M; Strom, J.; Montague, W.; Hedrick, L.D. 2004. Shortleaf pine-bluestem habitat restoration in the interior highlands: Implications for stand growth and regeneration. In: Shepperd, W.D.; Eskew, L.G., comps. Silviculture in special places: proceedings of the national silviculture workshop. RMRS-P-34. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 182-190.

Guyette, R.P.; Spetich, M.A.; Stambaugh, M.C. 2006. Historic fire regime dynamics and forcing factors in the Boston Mountains, Arkansas, USA. Forest Ecology and Management. 234: 293-301.

Hedrick, L.D.; Bukenhofer, G.A.; Montague, W.G. [and others]. 2007. Shortleaf pine bluestem restoration in Ouachita National Forest. In: Kabrick, J.M.; Dey, D.C. Dey, Gwaze, D. eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service Northern Research Station: 206-213.

Johnson, F.L.; G.D. Schnell. 1985. Wildland fire history and the effects of fire history at Hot Springs National Park, Arkansas. Report to National Park Service, Santa Fe, New Mexico. Oklahoma Biological Survey. Norman, OK: University of Oklahoma. p. 49.

Kabrick, J.M.; Dey, D.C.; Shifley, S.R.; Villwock, J.L. 2011. Early survival and growth of planted shortleaf pine seedlings as a function of initial size and overstory stocking. In: Fei, S.; Lhotka, J.M.; Springer, J.W.; Gottschalk, K.W.; Miller, G.W. eds. Proceedings of the 17th central hardwoods forest conference. Gen. Tech. Rep. NRS-P-78. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 277-286.

Lilly, C.J. 2010. Shortleaf pine: The basal crook adaptation and the traits it infers to its hybrid with loblolly pine. Stillwater, OK: Oklahoma State University. 174 p. M.S. thesis.

Lilly, C.J., Will, R.E.; Tauer, C.G. [and others]. 2012. Factors affecting the sprouting of shortleaf pine rootstock following prescribed fire. Forest Ecology and Management. 265: 13-19.

Little, S.; Somes, H.A.1956. Buds enable pitch and shortleaf pines to recover from injury. Station Pap. NE-81. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. p. 14.

Mattoon, W.R. 1915. Life history of shortleaf pine. Bull. 244. Washington, DC: U.S. Department of Agriculture. 46 p.

Moore, E.B. 1936. Seedling sprout growth of shortleaf and pitch pine in New Jersey. Journal of Forestry. 34(9): 879-882.

Shelton, M.G. 1995. Effects of seedbed production, seedbed condition, and overstory basal area on the establishment of shortleaf pine seedlings in the Ouachita Mountains. Res. Pap. SO-293. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 13 p.

Shelton, M.G.; Cain, M.D. 2002. The sprouting potential of loblolly and shortleaf pines: Implications for seedling recovery from top damage. In: Walkingstick, T.; Kluender, R.; Riley, T. eds. Proceedings of the 2002 Arkansas Forest Resources Center Arkansas forestry symposium. Little Rock, AR: 17-21.

Stambaugh, M.C.; Guyette, R.P. 2006. Fire regime of an Ozark wilderness area, Arkansas. The American Midland Naturalist. 156(2): 237-251.

Stambaugh, M.C.; Guyette, R.P.; Dey, D.C. 2007. What fire frequency is appropriate for shortleaf pine regeneration and survival? In: Kabrick, J.M.; Dey, D.C. Dey, Gwaze, D. eds. Shortleaf pine restoration and ecology in the Ozarks: Proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 121-128.

Tauer, C.G.; Stewart, S.F.; Will, R.E. [and others]. 2012. Hybridization leads to loss of genetic integrity in shortleaf pine: Unexpected consequences of pine management and fire suppression. Journal of Forestry. 110(4): 216-224.

Vodrazka, F.M. 1998. Soil survey of Scott County, Arkansas. Vol. 57. U.S. Department of Agriculture, Natural Resources Conservation Service. 185 p.

Walker, L.C.; Wiant, H.V. 1966. Silviculture of shortleaf pine. Bulletin 9. Nacogdoches, Texas: Stephen F. Austin State College School of Forestry. p. 60.

Will, R.E.; Lilly, C.J.; Stewart, J. and others. 2013. Recovery from topkill of shortleaf pine x loblolly pine hybrids to their parent populations. Trees. 27: 1167-1174.

Yocom, H.A.; Lawson, E.R. 1977. Tree percent mortality from naturally regenerated shortleaf pine. Southern Journal of Applied Forestry. 1(2): 10-11.

Zhang, D.; Huebschmann, M.M; Lynch, T.B.; Guldin, J.M. 2012. Growth projection and valuation of restoration of the shortleaf pine-bluestem grass ecosystem. Forest Policy and Economics. 20: 10-15.

SITE SUITABILITY FOR SHORTLEAF PINE RESTORATION IN THE EASTERN ALABAMA FALL LINE REGION

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Abstract—Littleleaf disease is the most significant disease of shortleaf pine and has decimated shortleaf pine forests in the Piedmont region of the Southeast. This study used the littleleaf disease hazard soil rating method to evaluate the littleleaf hazard of Piedmont sites owned by Auburn University in the Auburn, Alabama area. The results indicate that a few of these stands are suitable for shortleaf pine management. Furthermore, soil cores were an accurate replacement for soil profiles, and soil series descriptions from the NRCS soil survey were not adequate for remotely assessing a site's littleleaf disease hazard.

INTRODUCTION

Shortleaf pine (*Pinus echinata*) forests were once one of the most widespread ecosystems in the Piedmont and Upper Coastal Plain regions of the southeastern United States (Lawson 1990, Mohr 1901 and 1897, Sargent 1884). Over the last half century, the area of shortleaf pine forests has declined sharply, particularly east of the Mississippi River (Oswalt 2012). Littleleaf disease, which is found primarily in the Piedmont region, is the most significant disease of shortleaf pine and has been responsible for much of this decline. This is due to both actual infection from the disease and to a general aversion towards shortleaf pine management caused by the disease (Campbell and Copeland 1954, Oak and Tainter 1988).

Littleleaf disease is caused by a water mold, *Phytophthora cinnamomi*, that is found in virtually all soils throughout the Southeast (Campbell and Copeland 1954, Mistretta 1984). Under wet soil conditions, *Phytophthora cinnamomi* attacks the fine roots of pine trees and kills them, which can result in the slow decline and mortality of the tree. Mortality depends on how aggressively the tree can grow new fine roots, the duration and frequency of attacks, and the general fertility of the soil. If intervals between wet soil conditions are too short, shortleaf pine cannot adequately grow new fine roots and will die from nitrogen deficiency (Campbell and others 1953, Campbell and Copeland 1954, Mistretta 1984). Due to the dependence of *P. cinnamomi* on wet soil conditions, littleleaf disease is most prevalent in poorly drained, heavy clay soils such as those commonly found in the Piedmont. Shortleaf pine is the most severely affected species, but loblolly pine (*Pinus taeda*) is also susceptible to a lesser degree.

Even in areas of high littleleaf disease incidence, the disease is site-specific based on soil drainage and erosion qualities (Campbell and Copeland 1954). A method for rating the littleleaf hazard of soils was developed by Campbell and Copeland (1954) that involves scoring four metrics of soil erosion and internal drainage: topsoil erosion, subsoil consistency, depth to the zone of greatly reduced permeability (permeability depth), and subsoil mottling. Topsoil erosion is rated as either "slight" (40 points), "moderate" (30 points), "severe" (20 points), or "rough gullied" (10 points); subsoil consistency is rated as either "very friable" (32 points), "friable" (24 points), "firm" (16 points), "very firm" (8 points), or "extremely firm" (0 points); permeability depth is measured in inches and grouped from 24-36 inches (15 points), 18-23 inches (12 points), 12-17 inches (9 points), 6-11 inches (6 points), or 0-6 inches (3 points); and subsoil mottling is rated as either "none" (13 points), "slight" (9 points), "moderate" (5 points), or strong (1 point). The sum of these four scores determines if a soil is at severe risk (0-50 points), moderate risk (51-74 points), or low risk ("healthy;" 75-100 points) for littleleaf disease (a full description of this soil rating method can be found in Campbell and Copeland 1954). This rating method should make it possible to assess potential shortleaf pine restoration sites in the Piedmont and other areas where littleleaf disease is a concern. With increasing interest in restoring shortleaf pine, extension agents, foresters, and forestry educators can contribute greatly to these efforts by instructing landowners on how to assess sites using this method. This paper outlines such an assessment and the soil testing procedure used.

METHODS

The site specific littleleaf hazard rating system was used to evaluate fifty stands on three tracts owned by

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Auburn University. Streamside Management Zones (SMZs) and open areas were not examined. The smallest stand evaluated was 1.2 acres, the largest was 51.9 acres, and the mean stand size was 16.1 acres. Four plots were semi-randomly placed in each stand using a topographical map to ensure coverage of major topographical features (e.g. slopes, ridgetops). Nine stands had less than four plots, due to small size or boundaries inconsistent with the stand map. Plots were navigated to using a F4 Devices Forge hand-held GPS unit (F4 Devices. http://f4devices.com/products/forge). At each plot, soil cores were collected using a 1 inch diameter by 16 inch deep cylindrical metal soil push probe. Prior to collection, the litter layer was cleared away and the soil probe inserted into the ground until substantial resistance was encountered. Once the soil core sample was obtained, it was evaluated for subsoil consistency, permeability depth, and subsoil mottling.

Subsoil consistency was measured by feeling the friability of the B-horizon soil layer (as described in Campbell and Copeland 1954). Depth to zone of greatly reduced permeability was measured as the depth of the soil core, measured in inches. The B-horizon layer was evaluated for subsoil mottling as described in Campbell and Copeland (1954). Erosion was assessed within each stand while traversing the stand, based on the degree of gullying and other visible signs of soil erosion (as described in Campbell and Copeland 1954). However, different ratings were given to individual plots if local conditions were substantially different from those in the rest of the stand.

The shortleaf pine component of each stand was assessed for the amount, size, and over-all health of individual trees. This assessment was limited to anecdotal observations along traverse routes. Stands with apparent, large, many and healthy shortleaf pine individuals were noted as having an important shortleaf pine remnant. Site sampling was conducted from December, 2014 to January, 2015.

Once initial data were analyzed, the numbers of plots required to achieve 10 percent +/- error in each stand were determined, and additional plots were added to thirteen stands to achieve this level of accuracy. Stand ratings were determined from the mean of the plots within each stand, and single sample, one-way t-tests were conducted on stand level data to determine which ones were equal to or over either 75 or 51, using Minitab Statistical Software version 17 (Minitab, Inc. 2015. www.minitab.com). The NRCS Web Soil Survey was used to determine which soil series each plot fell in, and plots were pooled by soil series and compared to each other using ANOVA tests, Tukey's range test, and t-tests. The number of plots required for each stand to achieve 10 percent +/- error was compared to stand size using simple regression analysis. Stands were then pooled based on whether they contained one or more soil series, and the average number of plots required in each group compared with a t-test. The soil ratings of stands noted for shortleaf pine were compared to those of stands without shortleaf pine using a t-test. Minitab Statistical Software version 17 and a confidence level of 90 percent (P-value < 0.1) were used for all tests.

Follow up testing was done in March, 2015, which involved digging seven soil profiles in one stand, evaluating the littleleaf hazard of each profile, and comparing the littleleaf hazard rating obtained from the soil profile to the littleleaf hazard rating obtained from a soil core taken from the center of the soil profile prior to digging the profile. The profiles were placed semi-randomly throughout the stand. Both the permeability depths and total littleleaf hazard ratings obtained from the profiles and cores at each location were compared using simple linear regression.

RESULTS

A total of 216 soil plots were taken in the fifty stands. Of these plots, 157 fell in Pacolet sandy loams, 52 fell in Gwinnett sandy loams, six fell in Enoree silt loams, and one fell in a Hiwassee sandy loam. Between one and eight plots were taken in each stand, with a mean of 4.3 plots per stand. The mean number of plots required for 10 percent +/- error was 1.7. Eleven of the fifty stands had important shortleaf pine remnants.

Two stands had a mean rating of 75 or higher and were rated as healthy. The remaining 48 stands had mean ratings between 50 and 75 and were classified as moderate littleleaf hazard stands. However, 15 of the moderate hazard stands were rated just below the threshold for healthy (between 70 and 75, P < 0.1).

There was no significant difference between littleleaf hazard ratings based on soil series type. Slope had no affect either, both within individual soil series or between soil series. There was no significant difference between ratings of stands with or without a shortleaf pine component, and both groups fell within the moderate hazard rating.

There was no significant trend between the size of a stand and the number of plots needed to sample the soil to 10 percent +/- error (slope = 0.01157 R-sq. = 0.6 percent), no difference between the number of plots required per stand based on soil series type, and no difference based on whether stands had one or two soil series (no stands with more than two soil series were tested). Both Pacolet and Gwinnett sandy loams were rated lower than suggested by their soil series survey descriptions. Oak and Tainter (1988) predicted that both of these soil series would be classified as healthy, but in this study both were classified as moderate hazard with a mean rating of 71 each.

Soil profiles yielded deeper permeability depth measurements than soil cores, and this difference increased with permeability depth (slope = 1.84, R-sq. = 74.6 percent, lower 90 percent CI = 1.07). However, this difference did not cause soil profiles to yield higher total littleleaf hazard ratings (slope = 1.03, R-sq. = 98 percent, lower 90 percent CI = 0.89).

DISCUSSION

Between two healthy stands and fifteen stands that were rated on the high end of moderate hazard (70-75), there should be suitable sites for shortleaf pine management on the tested tracts. Data from Campbell and Copeland (1954) indicate that mortality rates may be lower on sites rated from 70-75 than on sites rated below 70. The lack of difference between the littleleaf hazard of stands with or without shortleaf pine indicates that the presence of remnant shortleaf pine is probably due to past management rather than littleleaf disease severity. However, the shortleaf pine assessment did not fully, systematically, or equally cover the sampled stands. To fully compare stands with remnant shortleaf pine to those without, an intentional survey for remnant shortleaf pine would have to be conducted on all the sampled stands.

This study suggests that taking between four and eight plots should yield accurate results for most stands 50 acres or less with two or less soil series. Despite differences in at least one individual measure, soil cores appear to give equivalent results to soil profiles and may be used in their place. Because soil cores can be taken and measured easily and quickly, eight cores are recommended when sampling just one stand. If many stands are being sampled and time is limited, four plots per stand would likely yield suitable results. It is likely that these numbers would work for stands larger than 50 acres, as well.

Soil series descriptions were not adequate for remotely assessing littleleaf hazard. Both Pacolet and Gwinnett soils should be rated healthy based on their soil series descriptions, but both these series were rated moderate in this study. This discrepancy was likely due to erosion that has occurred since the original soil survey was conducted, and suggests that soil sampling is required for accurate littleleaf hazard determination, especially in soils predicted to have low hazard. More soil series should be tested and the results compared to predictions made from the series' survey descriptions in order to further explore this effect. It is likely that soil series descriptions can be used to determine unsuitable sites, because there is a low likelihood of soils showing substantial drainage and depth improvements since original soil surveys were conducted.

CONCLUSION

The Piedmont and Upper Coastal Plain were two of the areas of greatest historical shortleaf pine forest extent, and for current restoration efforts to be complete they must include sites in these regions. In the Piedmont in particular, the littleleaf hazard soil rating method developed by Campbell and Copeland (1954) may be used to assess a potential restoration site's risk of littleleaf disease. An initial evaluation should be made based on the described drainage properties of a site's soil series. If the soil series description suggests that the site is safe from littleleaf disease, eight soil cores should be taken throughout the stand, ensuring coverage of the major topographic features, and rated for littleleaf hazard using Campbell and Copeland's (1954) methodology. If many stands are being measured and time is limited, as few as four cores per stand will likely be sufficient. More cores may be needed if three or more soil series are found on the site. Testing additional soil series and stands with three or more soil series should contribute towards refining this easy and cheap method of evaluating the littleleaf disease hazard on potential shortleaf pine restoration sites in the littleleaf disease range.

LITERATURE CITED

Campbell, W.A.; Copeland, O.L. 1954. Littleleaf disease of shortleaf and loblolly pines. Cir. 940. Washington, DC: U.S. Department of Agriculture Forest Service. 41 p.

Campbell, W.A.; Copeland, O.L.; Hepting, G.H. 1953. Littleleaf in pines in the Southeast. Yearbook of Agriculture. Washington, DC: U.S. Department of Agriculture: 855-857.

Lawson, E.R. 1990. *Pinus echinata* Mill.—shortleaf pine. In: Burns, R.M.; Honkala, B.H., tech. cords. Silvics of North America. Vol. 1. Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service: 316-326.

Mistretta, P.A. 1984. Littleleaf disease. Forest Insect and Disease Leaflet 20. Washington, DC: U.S. Department of Agriculture. 6 p.

Mohr, C. 1901. Plant life of Alabama. Contributions from the U.S. National Herbarium, vol. 6. Washington, DC: Government Printing Office. 921 p.

Mohr, C.; Roth, Filibert. 1897. The timber pines of the southern United States: together with a discussion of the structure of their wood. Agric. Bull. 13. Washington, DC: U.S. Department of Agriculture Division of Forestry: 91-112.

Oak, S.W.; Tainter, F.H. 1988. How to identify and control littleleaf disease. Protection Report R8-PR 12. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region. 14 p.

Oswalt, C. 2012. Spatial and temporal trends of the shortleaf pine resource in the eastern United States. In: Kush, J.; Barlow, R.J.; Gilbert, J.C. eds. Proceedings of the shortleaf pine conference: east meets west. Special Report 11. Auburn, AL: Alabama Agricultural Experiment Station: 33-37.

Sargent, C.S. 1884. Report on the forests of North America (exclusive of Mexico). Washington, DC: U.S. Department of the Interior Census Office. 612 p.

SILVICULTURE'S IMPACT ON THE HISTORICAL SHORTLEAF COMPONENT OF PINE FORESTS IN THE UPPER WEST GULF COASTAL PLAIN

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Abstract—Silvicultural practices and human-induced alterations to natural disturbance regimes have contributed to a dramatic decline in shortleaf pine (*Pinus echinata*) across most of the Upper West Gulf Coastal Plain (UWGCP). The increased preference for faster-growing loblolly pine (*Pinus taeda*) in natural-origin stands, coupled with the spread of loblolly plantations and less fire on the landscape, have selected against shortleaf pine. While many are interested in reversing shortleaf's decline, remarkably little is known about the composition and structure of historical UWGCP natural-origin pine stands, and this lack of knowledge constrains our ability to establish restoration goals. As a first step, this review first contrasts the shortleaf composition of the virgin forest with that of well-stocked second-growth in the first half of the 20th century using a variety of sources. For instance, two extensive inventories conducted by the Forest Service, U.S. Department of Agriculture in the early 1930s in northeastern Louisiana and southeastern Arkansas surveyed cutover pinelands. Shortleaf pine was a prominent component in both inventories across all size classes, accounting for 20 to 40 percent of the pine sawtimber volume in many second-growth forests. These statistics are supported by later regional inventories as well as other UWGCP-based studies on wood decay, selective logging, site quality, pulpwood thinning, and seed tree management. Such examples can help identify the silvicultural contributions to this decline, thereby laying the foundation for conservation options.

INTRODUCTION

From southern Pennsylvania and New Jersey to eastern Oklahoma and Texas across to northern Florida, shortleaf pine (*Pinus echinata*) covers the widest geographic extent of any southern pine (Lawson 1990). However, a recent analysis of rangewide abundance using Forest Inventory and Analysis (FIA) data noted that shortleaf pine-dominated forests declined from 12.6 million acres in 1980 to 6.1 million acres in 2010—a decrease of 52 percent (Oswalt 2012). Although some regions (for example, the Ouachita Mountains in Arkansas and Oklahoma) have been less affected, others including the coastal plains across the southeastern U.S. have witnessed dramatic decreases across all size classes (Moser and others 2007, Oswalt 2012). Historically, the Upper West Gulf Coastal Plain (UWGCP) has been considered the pinnacle of shortleaf pine's distribution, with the biggest individual trees and highest stand volumes (Mohr and Roth 1897, Mattoon 1915). According to Mohr and Roth (1897, p. 94): "[w]est of the Mississippi River the Shortleaf pine finds its region of greatest profusion, forming forests of vast extent on the uplands of the undulating plain and table-lands of the hill country, which in their timber wealth and economic importance rival the great lumbering regions farther south." Yet, across the UWGCP today, shortleaf pine has become an increasingly minor species. According to the most recent FIA data, the previously codominant shortleaf now comprises only 11.7 percent of the region's 56 billion board feet of pine sawtimber, and has particularly declined in southwestern Arkansas and northern Louisiana.[1]

As with virtually all declines of once prominent species, the diminishment of shortleaf pine has its roots in both natural and anthropogenic causes. During the forest exploitation period of the southeastern U.S. (from 1880 to 1930), all of the major southern pines were heavily lumbered and untold billions of board feet of shortleaf were felled. Traditionally, shortleaf pine was preferred over loblolly pine (*Pinus taeda*) for lumber. Bray (1904, p. 52) described the wood of shortleaf pine as "…heavy, hard, strong, and generally coarse-grained…as lumber it is inferior only to longleaf pine," while loblolly wood was considered "…light, not strong, brittle, very coarse grained, and not durable." However, when grown under

[1] Gross sawtimber in terms of board feet, International ¼-inch rule. Includes data from all pine species for 2013 for Bailey's Ecological Subregion 231E (Mid-coastal plains, western region) for Arkansas, Louisiana, Oklahoma, and Texas from the U.S. Forest Service's Forest Inventory Data Online (FIDO) website (http://apps.fs.fed.us/fia/fido/index.html), accessed 12 February 2015. Note that other board foot reports in this paper could be in terms of International ¼ inch, Doyle, Scribner, or other log rules (some sources are not specific).

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comparable circumstances, the wood quality between old-growth loblolly and shortleaf pine probably differed less than that noted by Bray. Commercially, there was little distinction between these species, especially when contrasted to old-growth longleaf (*Pinus palustris*) and slash pine (*Pinus elliottii*). The label "shortleaf" was applied when marketing the lumber of both *Pinus echinata* and *Pinus taeda* (Davis 1931) and was often used interchangeably to describe either species in the field.

Over time, opinions of loblolly pine timber changed, aided by a fast growth rate and the comparative ease of regenerating this species. Loblolly took advantage of favorable old-field sites and cutover timberlands now protected from fire to rapidly expand its distribution across the UWGCP, becoming the dominant pine species across the region by the mid-20th century. This dominance has only increased in recent years, as many natural-origin pine, pine-hardwood, and hardwood forests have been replaced by loblolly pine plantations (Klepzig and others 2014). A concurrent decline of shortleaf pine went largely unheralded until recently (Moser and others 2007, Oswalt 2012). Growing concern helped prompt the formation of the "Shortleaf Pine Initiative" (http://www.shortleafpine.net/) to educate landowners and foresters about the decline of shortleaf and promote silviculture to help this species.

While many are interested in reversing the wane of shortleaf, little is known about its role in historical UWGCP natural-origin forests and the contributions of silviculture to its decline. This lack of knowledge constrains our ability to establish restoration goals for shortleaf pine. Fundamentally, there is no expectation that this timber-producing region will ever revert back to pre-Euroamerican dynamics sufficiently to return to an extensive shortleaf pine-dominated forest. However, a review of historical pine forest conditions across the UWGCP should provide valuable insights into what management options may be most effective in restoring shortleaf.

METHODS

The literature reviewed here is not intended to be an exhaustive list of the material available, but rather was chosen to help illustrate the role of silviculture in the decline of the shortleaf pine resource across the UWGCP. The UWGCP covers over 22 million acres of southern Arkansas, northern Louisiana, eastern Texas, and extreme southeastern Oklahoma and was chosen to highlight a portion of the range of shortleaf pine in which land ownership patterns, management practices, and site conditions remain potentially favorable for restoration. The UWGCP is still dominated by pine, although most is now loblolly and virtually no old growth remains (Klepzig and others 2014).

A large number of historical maps, photographs, inventory reports, scientific papers, and other documents were consulted; those that featured descriptions of well-stocked, second-growth pine and pine-hardwood UWGCP forests were chosen for further analysis. This resulted in most sources coming from after 1930. Before this time, there was little quantifiable information on relative shortleaf pine abundance and only sparse photographic evidence of stand composition and structure. Most references also came from before 1970, when naturally regenerated silviculture of second-growth stands dominated UWGCP landscapes. As the 20th century progressed, loblolly pine plantation management became the prevailing practice, and interest in natural-origin pine silviculture dropped greatly.

RESULTS AND DISCUSSION

Virgin Shortleaf Pine in the UWGCP

Almost without exception, the earliest reliable documentation indicates that upland pine-dominated forests across the UWGCP were heavily shortleaf pine. For example, Sargent's (1884) census report for the UWGCP in Arkansas, Louisiana, and Texas showed shortleaf as the most prominent pine of this region. Mohr and Roth (1897) provided semi-quantitative maps of the distribution of the major southern pine species, and from these (fig. 1) it is clear that shortleaf pine was dominant or codominant with loblolly pine (and, to a lesser degree, longleaf) across nearly the entire UWGCP. Olmsted (1902) noted that in some uncut lands near Pine Bluff, Arkansas, the "pine ridge" type averaged 5,945 board feet of merchantable pine sawtimber per acre, of which 64.2 percent was shortleaf and the remaining 35.8 percent was loblolly. His less common "pine flat" type on small stream terraces was better stocked (6,646 board feet per acre in pine) and was composed of 67.3 percent loblolly and 32.7 percent shortleaf. Parts of the UWGCP were even more shortleaf-dominant: a selective logging study in virgin pine timber from eastern Texas reported only a few hardwoods and a small fraction (about 7.5 percent) of loblolly (Garver and Miller 1933). The 1930s-vintage Forest Service, U.S. Department of Agriculture inventories for the UWGCP reported a major (25 to 50 percent) shortleaf component in the remaining old-growth pine-dominated forests (table 1).

Most other early UWGCP descriptions agree with these assessments, although they tend to be more qualitative (for example, Mattoon 1915). Bray (1904) mapped most of northeastern Texas as shortleaf pine forest (he called it the only important timber species in this region), with loblolly pine and longleaf pine more prominent in southeastern Texas. Foster (1912, p. 9) noted similar dominance of shortleaf pine across much of northern Louisiana—his "shortleaf pine uplands"

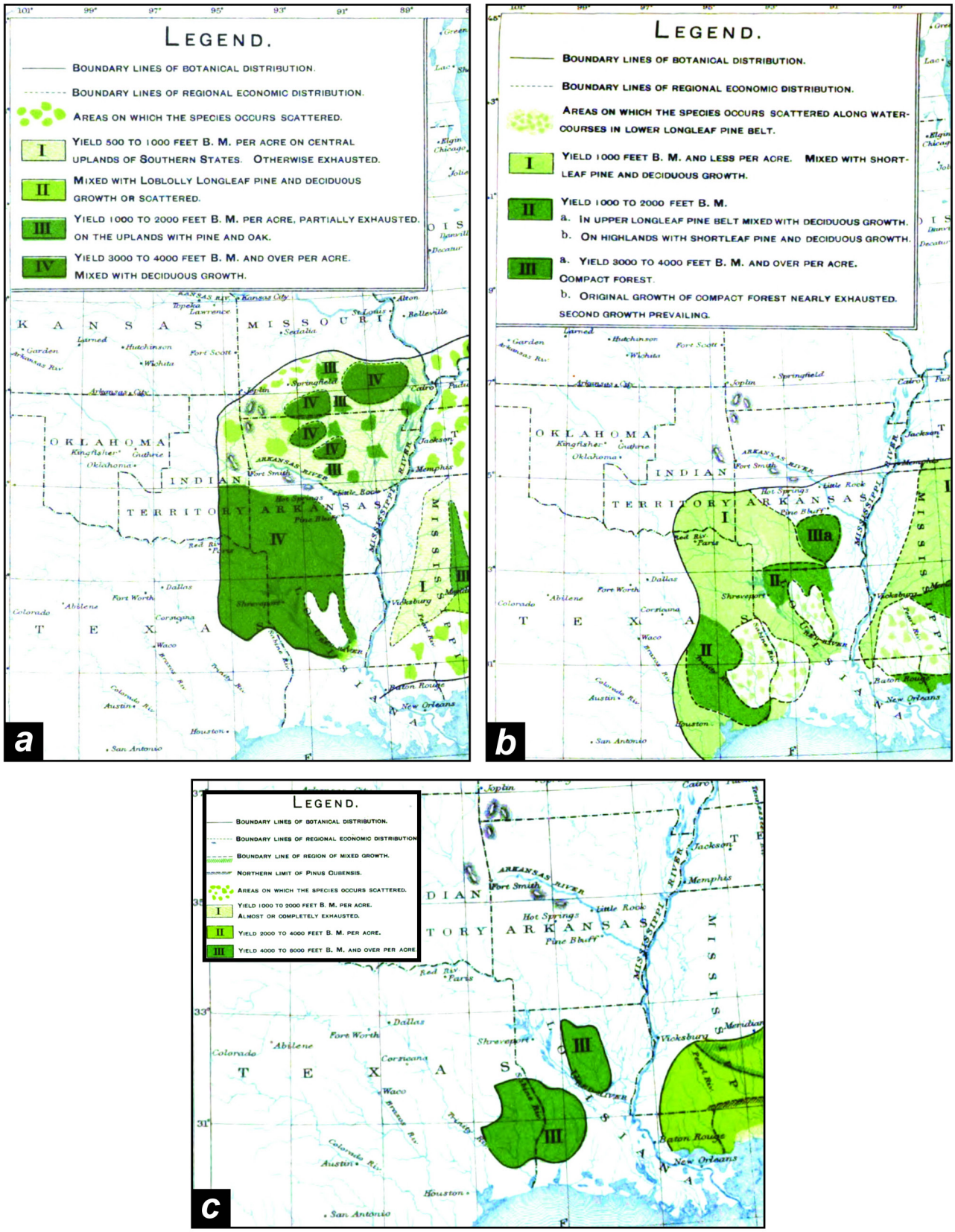


Figure 1—Historical distribution of the pine forests of the Upper West Gulf Coastal Plain, as drawn by Mohr and Roth (1897) and excerpted from their original maps of (*a*) shortleaf pine, (*b*) loblolly pine, and (*c*) longleaf pine (including slash pine east of the Mississippi River).

Table 1—Coverage, estimated pine sawtimber volume (foot, board measure (fbm)), and fractions of this sawtimber volume by species in uncut old-growth pine-dominated forests of the Upper West Gulf Coastal Plain as identified by U.S. Forest Service regional inventories in the mid-1930s

Region/State	Year of survey	Uncut old-growth area	Sawtimber volume	Pine species			Source
				Short-leaf	Loblolly	Long-leaf	
		-- acres --	*-- 1000s fbm --*	*-- percent of sawtimber --*			
SW Arkansas	1936	48,300	628,000	30.8	69.2	0.0	Eldredge (1937)
NW Louisiana	1935	61,100	387,800	39.9	60.1	0.0	Eldredge (1938a)
NE Texas	1935	33,600	341,700	49	46-50	1-5	Cruikshank (1938)
SE Texas	1935	180,900	1,177,800	24.7	58.8	16.5	Cruikshank and Eldredge (1939)

region had "pure" stands of shortleaf on the dry ridges and a mixture of hardwoods, shortleaf, and loblolly in the "intermediate" lands. Even in the better soils of northeastern Louisiana, Foster (1912) only mentioned shortleaf pine mixed with the hardwoods. Chapman (1913, p. 4) reported the virgin forests of a 27,000-acre tract in southeastern Arkansas and northeastern Louisiana were "…shortleaf and loblolly pine in almost equal mixture... [loblolly] gradually gives place to shortleaf on drier soils and on the driest the shortleaf grows pure." Russell R. Reynolds, the first scientist at the Crossett Experimental Forest, later described the virgin timber in this same area as 50 percent loblolly, 25 percent shortleaf, and 25 percent hardwoods (Reynolds and others 1984).

There are other qualitative accounts of the historical dominance of shortleaf pine across the UWGCP, including some photographs taken by U.S. Forest Service staff (fig. 2). Other historical photographs of the piney woods of the UWGCP are often labeled as showing "shortleaf" pine, although it is often not clear if the pines shown are *Pinus echinata*, *Pinus taeda*, or some mixture of these species. This taxonomic uncertainty is an unfortunate consequence of some early reports failing to distinguish between loblolly and shortleaf pine—many were only interested in differentiating between longleaf pine and other southern pines (Davis 1931, Bragg 2002, Bragg 2008). For instance, Morbeck (1915) described the uncut pine-hardwood forests of the Fordyce Lumber Company exclusively as shortleaf pine, even though loblolly was present in most of the upland virgin forests across southern Arkansas, particularly along the smaller stream bottoms (Mohr and Roth 1897, Olmsted 1902, Mattoon 1915, Chapman 1942).

Historical Accounts of Shortleaf in Second-Growth Pine Forests

Shortleaf's prominence on most upland sites in the UWGCP waned as the old growth was cleared or otherwise disrupted by widespread Euroamerican settlement. Regrettably, most data on the abundance and size-class distribution of second-growth shortleaf pine in the UWGCP came decades after most of the virgin timber had been cleared. During the first decades of the 20th century, the nascent forestry profession had yet to offer meaningful management options for second-growth forests, and very few publications considered the prospects for shortleaf pine silviculture. For example, Bray (1904, p. 39) dedicated only one small paragraph on "conservative lumbering" in shortleaf pine forests of northeastern Texas, as he was convinced that "…the [second-growth shortleaf] forests of this region will find their chief usefulness as protective forests and woodlots."

Early U.S. Forest Service inventories—During their early years, U.S. Forest Service experiment stations often aided private landowners and government agencies interested in sustainable forestry by inventorying their cutover timberlands. In 1931, the Southern Forest Experiment Station assessed the lands of the Union Saw Mill Company in northern Louisiana and southern Arkansas. Union Saw Mill was trying to determine if their uncut second-growth and recently



Figure 2—Photograph from the fall of 1937 of a stand of young and old shortleaf pine-dominated timber near Hamburg, Arkansas. U.S. Forest Service image from the files of the Crossett Experimental Forest.

cutover pine lands could sustain their mill in Huttig, Arkansas, and Forest Service forester A.E. Wackerman cruised their approximately 152,000 acres of uncut second-growth timber. Wackerman estimated these lands averaged 5,210 board feet (International ¼-rule log scale) per acre of pine, of which 1,574 board feet were shortleaf (just over 30 percent of the pine; fig. 3) (Unpublished 1936 report to Mr. F.W. Scott, President, Union Saw Mill Company, Huttig, Arkansas, on a management plan for sustained yield. On file with D.C. Bragg). Most of the shortleaf pine sawtimber volume was less than 15 inches diameter at breast height, suggesting that shortleaf had robustly regenerated following the clearing of the virgin pine. Similarly, Russell R. Reynolds provided the data and analysis for second-growth upland forests in UWGCP for the National Plan for American Forestry (also known as the "Copeland Report"; USDA Forest Service 1933). According to Reynolds' work (pages 945-946 in the Copeland Report), an "average-stocked" acre of pine-hardwood (inventoried across 345,000 acres and thought to be "typical of conditions on 7 million acres of this type" across southern Arkansas, northern Louisiana, and eastern Texas) had 1,000 board feet of shortleaf pine, 1,768 board feet of loblolly pine, and 1,920 board feet of hardwoods in sound trees (fig. 4a). Although the extent of "better-stocked shortleaf-loblolly-hardwood stands" was not specifically given (it was probably in the hundreds of thousands of acres), this stand condition likewise had a substantial amount of shortleaf pine sawtimber (4,371 board feet per acre, or 37.6 percent of all pine sawtimber; fig. 4b). In both stand types, shortleaf pine was well distributed across the range of tree diameters.

The U.S. Forest Service's Southern Forest Survey conducted the first formal inventories of the UWGCP in the 1930s. These inventories found shortleaf to be a prominent component of uncut second-growth stands, both young and old. With the notable exception of northeastern Texas, shortleaf averaged between 18.6 and 24.1 percent of all sawtimber, and 26.4 to 47.6 percent of pine sawtimber (table 2). As mentioned earlier, northeastern Texas was even more shortleaf dominated, with between 30.4 and 44.2 percent of all sawtimber and 65.7 to 70.8 percent of pine sawtimber. Unfortunately, these inventories do not detail relative pine abundance in the corner of the UWGCP that occurs in extreme southeastern Oklahoma (the southern half of McCurtain County). However, both loblolly and shortleaf are known to have been common in this small (less than 80,000 acres) enclave of pine-hardwood (Little and Olmsted 1936, Blair and Hubbell 1938, Eldredge 1938b, Duck and Fletcher 1945).

Other technical forestry reports—By the mid-20th century, technical forestry reports had increased substantially, but many authors did not distinguish between loblolly and shortleaf pine, limiting their utility for this paper. Hence, this section focuses on papers that detail species abundance. For example, Garver and Miller's (1933, p. 10) evaluation of selective logging in UWGCP forests noted that a 17-acre tract of somewhat understocked, approximately 60-year-old second-growth pine stand in southern Arkansas was "…about equally divided between shortleaf and loblolly pine…." This was considerably less than the over 90 percent shortleaf sawtimber from some companion virgin stands in eastern Texas, but far exceeded the roughly 6 percent they noted in a 13-acre old-field stand in northern Louisiana (Garver and Miller 1933). Old-field stands that arose following the abandonment of agriculture across the UWGCP are usually considered loblolly pine habitat. However, shortleaf pine can also dominate old fields, assuming an adequate seed source was present (Mattoon 1915). For instance, Forbes and Stuart (1930, p. 10) noted old-field stands "containing 80 per cent or more" of shortleaf pine commonly occurred in the UWGCP. Likewise, Turner's (1936) site index study of mature (greater than 50 years old), mostly old-field pine stands across southern Arkansas sampled shortleaf as often as loblolly pine, suggesting these species occurred in approximately equal frequencies. Representation was not always equivalent—Guttenberg (1954) noted that a dense, 44-year-old old-field pine stand on the Crossett Experimental Forest was 80 percent loblolly pine and 20 percent shortleaf pine in 1937—but still a much higher proportion of shortleaf than assumed today.

Other studies can be used to infer shortleaf abundance. In the 18 soils series Turner (1936) compared for site index, shortleaf pine was absent from the two most poorly drained soils considered, and was less commonly sampled than loblolly in only one other soil series. A later, large-scale study of non-randomly chosen stands across southern Arkansas and northern Louisiana placed 9 percent of their sites in "pure" shortleaf pine stands and another 24 percent in stands classified as mixed loblolly and shortleaf pine (Zahner 1958). In an assessment of wind firmness in shortleaf and loblolly seed trees on the Crossett Experimental Forest (Grano 1953), harvests were done on two 40-acre compartments in 1949, retaining 13 to 21 seed trees per acre. Of these seed trees, 55 percent were loblolly and 45 percent were shortleaf pine (Grano 1953). While shortleaf pines may have been disproportionately retained, Grano (1953, p. 116) noted that "…only the best individuals were selected for seed trees" and at that time the Crossett Experimental Forest did not prefer one pine over the other, suggesting that almost half of the pines were probably shortleaf prior to cutting. This assertion is supported by Reynolds (1959, p. 5), who stated that the second-growth forests of southern Arkansas and northern Louisiana were "… often half loblolly and half shortleaf…." Stephenson

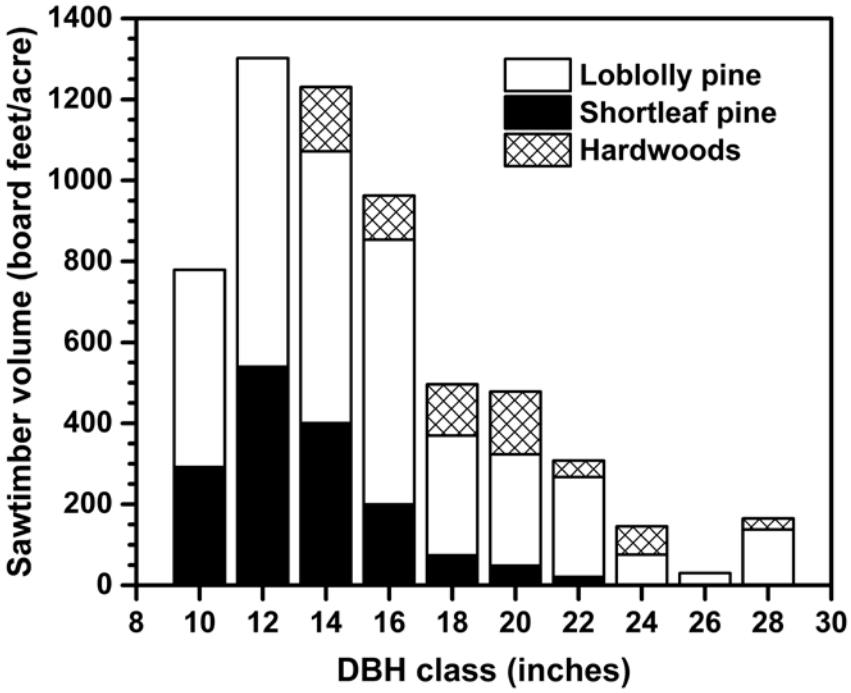


Figure 3—Size and sawtimber volume distribution by species for 152,000 acres of second-growth pine-dominated forests in Union County, Arkansas and Union Parish, Louisiana in the early 1930s adapted from A.E. Wackerman's unpublished 1936 report to the Union Saw Mill Company. Note: data did not include trees less than 10 inches DBH (diameter at breast height).

(1963) noted second-growth stands on the E.L. Kurth Experimental Forest in eastern Texas were at least 90 percent shortleaf pine, with only a few plots having more than 25 percent loblolly.

The Likely Impacts of Silvicultural Practices on Shortleaf's Decline

Undoubtedly, the application of silviculture in the UWGCP has contributed to the decline of shortleaf pine. This waning almost certainly began with the clearing of the virgin forest and was then continued by treatments that, either directly or indirectly, discriminated against shortleaf. Reynolds (1951, p. 4) noted as much in a guidebook he prepared: "Management [on the Crossett Experimental Forest] is gradually converting the original shortleaf-loblolly pine-hardwood stands to a shortleaf-loblolly pine type in which loblolly is increasingly dominant." This admission is significant because shortleaf was not treated differently from loblolly at Crossett, yet loblolly was being favored by the uneven-aged practices of Reynolds.

As silvics information swelled, foresters quickly realized that loblolly outgrew shortleaf pine on the same UWGCP sites, especially when protected from fire (for example, Record 1907, Chapman 1942). It is possible that the smaller size of shortleaf pine may have led to higher rates of removal during thinning operations, which were often done from below to remove what were thought to be intermediate or suppressed trees. Furthermore, although it was not necessarily appreciably different than loblolly pine, shortleaf developed a reputation for being a less prolific seed producer (Barnett and Haugen 1995) and more prone to certain forest health issues, including susceptibility to southern pine beetle (*Dendroctonus frontalis*) (Ku and others 1980), greater windthrow of seed trees (Grano 1953), and periodic unexplained declines (Williams and Tainter 1971). This perception often led to recommendations to select against shortleaf in natural-origin pine stands. For example, Grano (1953) advised managers that shortleaf seed trees either be retained in 'mutually protective' small groups or loblolly pine should be favored. With these concerns (deserved or not), it is not surprising that foresters increasingly selected against shortleaf pine in second-growth forests.

The recent success of loblolly pine across the UWGCP is not due solely to harvest practices. As cutover and agriculturally abandoned lands reforested, loblolly's proclivity to occupy favorable sites allowed it to rapidly expand. Concurrently, fire suppression in the UWGCP became widespread during the 1930s, with forest coverage increases being attributed in part to better

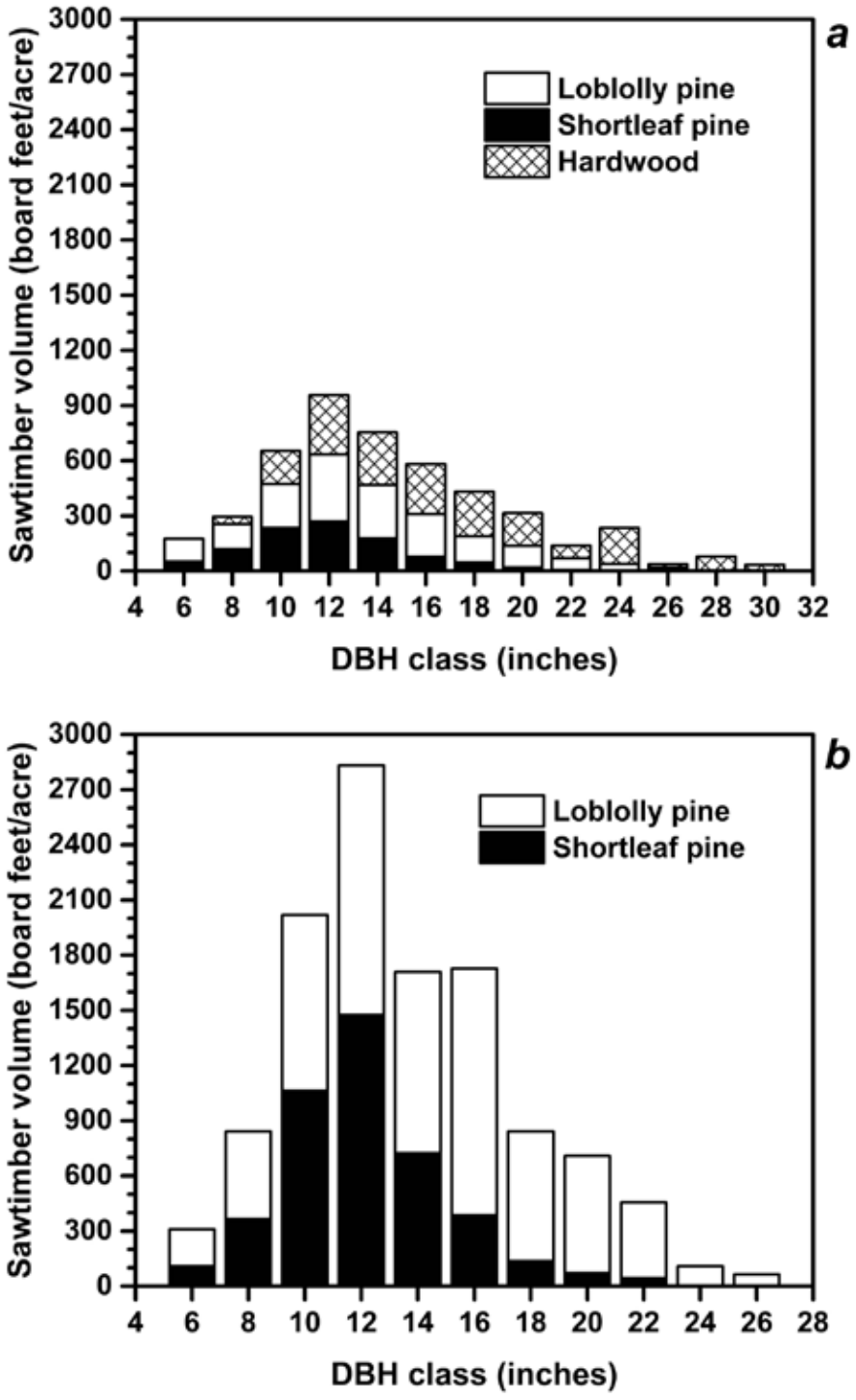


Figure 4—Size and sawtimber volume distributions by species for (*a*) average and (*b*) exceptionally well-stocked second-growth pine-dominated forests thought to be representative of hundreds of thousands to millions of acres across the Upper West Gulf Coastal Plain in southern Arkansas, northern Louisiana, and northeastern Texas in the early 1930s (USDA Forest Service 1933). (DBH = diameter at breast height).

Table 2—Quantity and proportion of sawtimber (foot, board measure (fbm)) in uncut second-growth pine-dominated forests in the Upper West Gulf Coastal Plain by different species or species groups estimated by U.S. Forest Service regional inventories in the mid-1930s

Region/State	Stand type [a]	Shortleaf	Loblolly/ longleaf [b]	All hardwood	All species	Pines only	Source
		---------- billions fbm ----------			----- percent -----		
SW Arkansas	sawlog	2.053	3.632	2.954	23.8	36.1	Eldredge (1937)
SW Arkansas	under	0.030	0.033	0.079	21.1	47.6	Eldredge (1937)
NW Louisiana	sawlog	1.240	2.054	1.855	24.1	37.6	Eldredge (1938a)
NW Louisiana	under	0.022	0.031	0.053	20.8	41.5	Eldredge (1938a)
NE Texas	sawlog	1.744	0.910	1.295	44.2	65.7	Cruikshank (1938)
NE Texas	under	0.034	0.014	0.064	30.4	70.8	Cruikshank (1938)
SE Texas	sawlog	1.134	3.155	1.793	18.6	26.4	Cruikshank and Eldredge (1939)
SE Texas	under	0.058	0.133	0.118	18.7	30.4	Cruikshank and Eldredge (1939)

[a] Stand types are either "sawlog size" or "under sawlog size"; under sawlog size includes reproducing stands.

[b] This combines loblolly and longleaf pine, when present--there is no longleaf pine in the Arkansas inventory. Longleaf is typically a minor component in the NW Louisiana and NE Texas inventories.

fire protection (for example, Eldredge 1937). Prior to this, much of the region burned frequently (for example, Bruner 1930), which helped shortleaf pine more than loblolly, given shortleaf's ability to sprout from the basal crook when young (Mattoon 1915). It is hard to overstate the impact of fire on the success of shortleaf pine regeneration. Mattoon (1915, p. 20) claimed that "…the majority of all standing shortleaf timber examined in various portions of Arkansas [including the UWGCP] was found to be of coppice origin." During the first few decades of the 20th century, fire suppression was often considered the only silvicultural option. As an example, Peters (1916) spent over three pages on fire in an assessment of forest conservation in southern pine, compared to only a half-page on forest management. After all, without effective fire control, it was thought to be impossible to get a properly stocked stand established (Bruner 1930).

Prescribed fire eventually returned as a means of competition control and as a habitat restoration tool. By the mid-1980s, some UWGCP landowners burned tens of thousands of acres annually (for example, Georgia-Pacific's Mid-Continent Division in Arkansas, Louisiana, and Mississippi; Williams 1985), a practice that is still common on public lands across the region. The use of fire to clear logging slash, prepare seedbeds, and restrict non-pine competition was implemented to have minimal impact on established loblolly or shortleaf pine, and it is likely that both species benefited from prescribed fire during this period. However, under these circumstances, fire no longer limits loblolly pine regeneration, with some important genetic consequences (described later). In recent years, increasingly effective mechanical and chemical site preparation techniques have largely replaced prescribed fire as the preferred stand establishment tools, particularly as investment-heavy loblolly pine plantations have become the norm. Restoration-focused prescribed fire use has increased as of late, but continued population growth and urbanization pressures are expected to further erode UWGCP forest cover over the next 50 years (Klepzig and others 2014), and this does not bode well for the large-scale use of fire to help restore shortleaf pine.

The rapid and large-scale expansion of loblolly pine plantations across the UWGCP over the last 30 years has probably had the biggest impact on the decline of shortleaf pine. Installing loblolly pine plantations on upland sites is now the industry standard and has also been adopted by many small private landowners. Decades of loblolly pine tree improvement programs, better herbicides, intensive site preparation methods (including bedding and fertilization), superior (often containerized) seedlings, and custom-tuned density management strategies have dramatically increased volume yields over increasingly shorter rotations (see reviews in Borders and Bailey 2001, Stanturf and others 2003, Allen and others 2005, Fox and others 2007, Jokela and others 2010). With the considerable investment required to achieve these gains, yield losses to competing vegetation (including natural-origin pine) are to be avoided, prompting many foresters to intensify silvicultural treatments over increasingly larger areas. The net result has been that many of the natural-origin pine and pine-hardwood forests, with their heretofore substantial shortleaf pine component, have been converted to loblolly plantations where prescribed fire is avoided.

The combination of more loblolly on the landscape and the loss of fire have yet other unanticipated consequences for shortleaf pine. There appears to be an increasing degree of introgression in the genetic integrity of shortleaf (Tauer and others 2012). Recent studies have documented that shortleaf and loblolly pines have been crossbreeding, suggesting that many of the putative shortleaf seedlings in naturally regenerated stands are, in fact, hybrids (Stewart and others 2012). One of the shortleaf pine traits that appears to be lost in these crosses is the strong basal crook and the concurrent ability to resprout following topkilling (Will and others 2013), which conveyed a distinct advantage over loblolly in fire-prone environments (Mattoon 1915, Walker and Wiant 1966, Stewart and others 2015). Frequent fire likely helped maintain the genetic distinctness of both shortleaf and loblolly by killing most fire-susceptible hybrid seedlings (Will and others 2013, Stewart and others 2015). In the absence of fire, the shortleaf-loblolly hybrids, with their enhanced growth rates, can outcompete genetically pure shortleaf pine seedlings, thereby further pressuring an already declining species (Tauer and others 2012, Will and others 2013).

IMPLICATIONS

This review presents a preliminary (and, admittedly, superficial) approximation of the challenges facing foresters and landowners interested in restoring shortleaf pine. The loss of natural-origin pine-dominated stands to loblolly pine plantations has contributed to a loss of genetic diversity in both shortleaf and loblolly pine. This trend, when coupled with fire suppression and the expansion of loblolly pine's range, suggests that returning significant amounts of shortleaf pine to the UWGCP may prove exceedingly difficult (Tauer and others 2012). However, modifications to a number of silvicultural practices (such as the retention of frequent prescribed fire) could help stem shortleaf's decline, even on a small scale. Further work will be needed to determine if even more effective strategies will be needed, but the rapidly changing genetics of shortleaf pine in the UWGCP, coupled with impending climate change, invasive species, and other landscape pressures, strongly favor quick action.

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LITERATURE CITED

Allen, H.L.; Fox, T.R.; Campbell, R.G. 2005. What is ahead for intensive pine plantation silviculture in the South? Southern Journal of Applied Forestry. 29(2): 62-69.

Barnett, J.P.; Haugen, R.O. 1995. Producing seed crops to naturally regenerate southern pines. Res. Pap. SO-286. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Blair, W.F.; Hubbell, T.H. 1938. The biotic districts of Oklahoma. American Midland Naturalist. 20(2): 425-454.

Borders, B.E.; Bailey, R.L. 2001. Loblolly pine—pushing the limits of growth. Southern Journal of Applied Forestry. 25(2): 69-74.

Bragg, D.C. 2002. Reference conditions for old-growth pine forests in the Upper West Gulf Coastal Plain. Journal of the Torrey Botanical Society. 129(4): 261-288.

Bragg, D.C. 2008. The prominence of pine in the Upper West Gulf Coastal Plain during historical times. In: Hardy, L.M., ed. Freeman and Custis Red River expedition of 1806: two hundred years later. Bull. 13. Shreveport, LA: Louisiana State University-Shreveport, Museum of Life Sciences: 29-54.

Bray, W.L. 1904. Forest resources of Texas. Bull. 47. Washington, DC: U.S. Department of Agriculture Forest Service Bureau of Forestry. 71 p.

Bruner, E.M. 1930. Forestry and forest fires in Arkansas. Circular 281. Fayetteville, AR: University of Arkansas, Agriculture Extension Service. 30 p.

Chapman, H.H. 1913. Prolonging the cut of southern pine. Part I. Possibilities of a second cut. Bull. 2. New Haven, CT: Yale University Press, Yale Forest School. 32 p.

Chapman, H.H. 1942. Management of loblolly pine in the pine-hardwood region in Arkansas and in Louisiana west of the Mississippi River. Bull. 49. New Haven, CT: Yale University Press, Yale Forest School. 150 p.Cruikshank, J.W. 1938. Forest resources of northeast Texas. Forest Survey Release 40. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 25 p.

Cruikshank, J.W.; Eldredge, I.F. 1939. Forest resources of southeastern Texas. Misc. Publ. 326. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 37 p.

Davis, E.M. 1931. The defects and some other characteristics of virgin-growth and of second-growth commercial shortleaf pine lumber. Journal of Forestry. 29: 54-63.

Duck, L.G.; Fletcher, J.B. 1945. A survey of the game and furbearing animals of Oklahoma. State Bull. 3. Ponca City, OK: Oklahoma Game and Fish Commission. 144 p.

Eldredge, I.F. 1937. Forest resources of southwest Arkansas. Forest Survey Release 27. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 21 p.

Eldredge, I.F. 1938a. Forest resources of northwest Louisiana. Forest Survey Release 31. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 21 p.

Eldredge, I.F. 1938b. Forest resources of southeast Oklahoma. Forest Survey Release 37. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 25 p.

Forbes, R.D.; Stuart, R.Y. 1930. Timber growing and logging and turpentining practices in the southern pine region. Tech. Bull. 204. Washington, DC: U.S. Department of Agriculture Forest Service. 115 p.

Foster, J.H. 1912. Forest conditions in Louisiana. Bull. 114. Washington, DC: U.S. Department of Agriculture. 39 p.

Fox, T.R.; Jokela, E.J.; Allen, H.L. 2007. The development of pine plantation silviculture in the southern United States. Journal of Forestry. 105(7): 337-347.

Garver, R.D.; Miller, R.H. 1933. Selective logging in the shortleaf and loblolly pine forests of the Gulf States region. Tech. Bull. 375. Washington, DC: U.S. Department of Agriculture. 53 p.

Grano, C.X. 1953. Wind-firmness of shortleaf and loblolly pines. Southern Lumberman. 187(2345): 116.

Guttenberg, S. 1954. Growth and mortality in an old-field southern pine stand. Journal of Forestry. 52: 166-168.

Jokela, E.J.; Martin, T.A.; Vogel, J.G. 2010. Twenty-five years of intensive forest management with southern pines: important lessons learned. Journal of Forestry. 108(7): 338-347.

Klepzig, K.; Shelfer, R.; Choice, Z. 2014. Outlook for Coastal Plain forests: a subregional report from the Southern Forest Futures Project. Gen. Tech. Rep. SRS-196. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 68 p.

Ku, T.T.; Sweeney, J.M.; Shelburne, V.B. 1980. Site and stand conditions associated with southern pine beetle outbreaks in Arkansas—a hazard-rating system. Southern Journal of Applied Forestry. 4(2): 103-106.

Lawson, E.R. 1990. *Pinus echinata* Mill.—shortleaf pine. In: Burns, R.M.; Honkala, B.H., tech. cords. Silvics of North America. Vol. 1. Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 316-326.

Little, E.L., Jr.; Olmsted, C.E. 1936. Trees and shrubs of the southeastern Oklahoma protective unit. Proceedings of the Oklahoma Academy of Science. 16: 52-61.

Mattoon, W.R. 1915. Life history of shortleaf pine. Bull. 244. Washington, DC: U.S. Department of Agriculture. 46 p.

Mohr, C.; Roth, F. 1897. The timber pines of the southern United States. Bull. 13. Washington, DC: U.S. Department of Agriculture Division of Forestry. 176 p.

Morbeck, G.C. 1915. Logging shortleaf pine in Arkansas. Ames Forester. 3: 92-118.

Moser, W.K.; Hansen, M.; McWilliams, W.H.; Sheffield, R.M. 2007. Shortleaf pine composition and structure in the United States. In: Kabrick, J.M.; Dey, D.C.; Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 19-27.

Olmsted, F.E. 1902. A working plan for forest lands near Pine Bluff, Arkansas. Bull. 32. Washington, DC: U.S. Department of Agriculture Bureau of Forestry. 48 p.

Oswalt, C.M. 2012. Spatial and temporal trends of the shortleaf pine resource in the eastern United States. Special Rep. 11. In: Kush, J.; Barlow, R.J; Gilbert, J.C., eds. Proceedings of the shortleaf pine conference: east meets west. Auburn, AL: Alabama Agricultural Experiment Station: 33-37.

Peters, J.G. 1916. Forest conservation for states in the southern pine region. Bull. 364. Washington, DC: U.S. Department of Agriculture. 14 p.

Record, S.J. 1907. The forests of Arkansas. Forestry Quarterly. 5(3): 296-301.

Reynolds, R.R. 1951. Guide to Crossett Experimental Forest. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 65 p.

Reynolds, R.R. 1959. Eighteen years of selection timber management on the Crossett Experimental Forest. Bull. 1206. Washington, DC: U.S. Department of Agriculture Forest Service. 68 p.

Reynolds, R.R.; Baker, J.B.; Ku, T.T. 1984. Four decades of selection management on the Crossett Farm Forestry forties. Bull. 872. Fayetteville, AR: University of Arkansas, Agricultural Experiment Station. 43 p.

Sargent, C.S. 1884. Report on the forests of North America (exclusive of Mexico). Washington, DC: U.S. Department of the Interior, Census Office. 612 p.

Stanturf, J.A.; Kellison, R.C.; Broerman, F.S.; Jones, S.B. 2003. Productivity of southern pine plantations: where are we and how did we get here? Journal of Forestry. 101(3): 26-31.

Stephenson, G.K. 1963. Ten years of shortleaf pine seed crops in Texas. Journal of Forestry. 61(4): 270-272.

Stewart, J.F.; Tauer, C.G.; Nelson, C.D. 2012. Bidirectional introgression between loblolly pine (*Pinus taeda* L.) and shortleaf pine (*P. echinata* Mill.) has increased since the 1950s. Tree Genetics & Genomes. 8(4): 1-11.

Stewart, J.F.; Will, R.E.; Robertson, K.M.; Nelson, C.D. 2015. Frequent fire protects shortleaf pine (*Pinus echinata*) from introgression by loblolly pine (*P. taeda*). Conservation Genetics. 16: 491-495.

Tauer, C.G.; Stewart, J.F.; Will, R.E.; Lilly, C.J.; Guldin, J.M.; Nelson, C.D. 2012. Hybridization leads to loss of genetic integrity in shortleaf pine: unexpected consequences of pine management and fire suppression. Journal of Forestry. 110(4): 216-224.

Turner, L.M. 1936. Factors influencing the rate of growth of pine in Arkansas. Ecology. 17(2): 227-240.

USDA Forest Service. 1933. A national plan for American Forestry, volume I. Letter from the Secretary of Agriculture transmitting in response to S. Res. 175 (Seventy-Second Congress) (Copeland Report). Washington, DC: Government Printing Office. 1050 p.

Walker, L.C.; Wiant, H.V., Jr. 1966. Silviculture of shortleaf pine. Bull. 9. Nacogdoches, TX: Stephen F. Austin State College, School of Forestry. 60 p.

Will, R.E.; Lilly, C.J.; Stewart, J.; Huff, S.; Tauer, C.G. 2013. Recovery from topkill of shortleaf pine x loblolly pine hybrids compared to their parent populations. Trees. 27: 1167-1174.

Williams, R.A. 1985. Use of prescribed fire on industrial lands in the Gulf Coastal Plain and uplands. In: Wade, D.D., comp. Prescribed fire and smoke management in the South: conference proceedings. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 33-38.

Williams, R.A.; Tainter, F.H. 1971. Pine decline in southern Arkansas. Plant Disease Reporter. 55(11): 1036.

Zahner, R. 1958. Site-quality relationships of pine forests in southern Arkansas and northern Louisiana. Forest Science. 4(2): 162-176.

MULTIPLE TREATMENTS YIELD EARLY SUCCESS IN A SHORTLEAF PINE WOODLAND RESTORATION PROJECT IN THE MISSOURI OZARKS

Matthew G. Olson and Elizabeth K. Olson[1]

Abstract--Shortleaf pine woodland communities were more extensive in the southeastern Missouri Ozarks prior to Euro-American settlement than today. In 2000, the Missouri Department of Conservation initiated a shortleaf pine woodland restoration project on state land in the Ozarks of southeast Missouri at an area called the Midco Pine Flats Restoration Area. The purpose of this investigation was to evaluate shortleaf pine woodland restoration efforts at Midco over the 14-year period from 2000-2013. Our analysis suggested that treatments impacting both the overstory and understory, including mechanical release of both planted and natural pine regeneration, yielded the best short-term results. In contrast, the failure of burning alone to significantly enhance both pine and woodland ground flora in the short-term highlights the importance of canopy disturbance to increase light to the understory to restore this woodland natural community. Increased competition from regenerating hardwood species appeared to limit both pine recruitment and woodland herbaceous cover expansion. Future release treatments will likely be needed to promote shortleaf pine and woodland flora at this and similar sites where shortleaf pine woodland restoration is a management goal.

INTRODUCTION

Prior to Euro American settlement, shortleaf pine (*Pinus echinata*) was a dominant tree species of the southeastern Missouri Ozarks, where it grew in pure stands and in mixtures with hardwood species (Hanberry and others 2012). At the time of settlement in the Missouri Ozarks, shortleaf pine grew as part of a shifting landscape mosaic of forest and woodland natural communities shaped by ecological site variation and frequent fire (Ladd 1991, Batek and others 1999, Hanberry and others 2012).

Woodlands are terrestrial communities dominated by trees, with sparse midstory and understory tree layers (Nelson 2010). Although woodlands can have high variability in overstory tree density, the canopies of woodlands are generally more open than forest canopies. This open structure allows for a conspicuous ground layer of diverse forb, grass, and sedge species (Nelson 2010). Historical shortleaf pine woodland communities likely supported a rich understory of about 300 species of forbs, grasses, and sedges (Ladd and others 2007). With frequent fire, understory woody stems would be suppressed, and grasses would become dominant in the understory (Masters 2007). Historic accounts describe the ground layer of shortleaf pine woodlands as composed of blueberry shrubs (*Vaccinium* spp.), bluestem grasses (*Andropogon gerardii* and *Schizachyrum scoparium*), and other herbaceous perennials associated with high light intensities (Ladd and others 2007).

Multiple historical factors have led to the loss of shortleaf pine woodland communities from the Missouri Ozarks. Extensive logging from 1880-1920 targeted the removal of shortleaf pine (Cunningham and Hauser 1989) and caused an immediate loss of mature trees, which, in turn, diminished the seed source for new pine seedlings. Associated with extensive logging and human population density was an increase in fire frequency and intensity (Guyette and others 2002), which acted as a barrier to the recruitment of shortleaf pine beyond the seedling stage (Guyette and others 2007). Although fire can facilitate shortleaf pine seedling establishment by reducing litter depth and woody competition (Little and Moore 1949), recurrence at too frequent intervals can prohibit shortleaf pine from reaching a size large enough to resist top-kill (Dey and Hartman 2005). Shortleaf pine is one of only a few North American pine species capable of basal sprouting (Lawson 1990), but shortleaf pine sprout growth is less vigorous than that of co-occurring oak species. In concert, extensive pine harvesting and frequent fire diminished shortleaf dominance, while simultaneously increasing the dominance of oak species. Finally, active fire suppression starting in the 1940s enabled an accumulation of midstory and overstory trees on sites that previous supported more open canopies

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(Hanberry and others 2013). The resulting forests, dominated by oak species, did not admit enough light to the understory for light-demanding woodland ground flora species to survive and reproduce or for shade-intolerant shortleaf pine regeneration to persist, grow, and eventually recruit into the canopy. Collectively, these factors reduced the extent of pine woodland communities in the southeastern Missouri Ozarks, making these natural communities some of the most imperiled in the state.

In 2000, the Missouri Department of Conservation (MDC) initiated a shortleaf pine woodland community restoration project on state land in the Ozarks of southeastern Missouri at an area called the Midco Pine Flats Restoration Area (Midco). This area experienced severe and widespread red oak decline – mainly black oak (*Quercus velutina*) and scarlet oak (*Q. coccinea*) – in the mid to late-1990s, prompting MDC managers to prescribe extensive salvage harvesting and to critically consider alternative management options for mitigating future oak decline. With the predominance of dry sites at Midco, pine-dominated communities were thought to be more appropriate for much of the area than forests dominated by red oak species. Shifting composition from red oak to shortleaf pine would lower the vulnerability of these communities to red oak decline (Law and others 2004; Blizzard and others 2007) and restore an imperiled Missouri natural community.

The purpose of this investigation was to evaluate shortleaf pine woodland restoration efforts at Midco over the last 14 years and to determine if treatments have yielded early success. Specifically, we are interested in identifying the treatment that has produced the greatest gains in shortleaf pine abundance, woodland herbaceous plant richness and abundance, and overall floristic quality.

METHODS

Midco is a 2,233-acre site located on Peck Ranch Conservation Area (PRCA). PRCA is located in the Current River Subsection of the Ozark Highlands Section (Nigh and Schroeder 2002). Most of Midco falls within the Current River Oak-Pine Woodland/ Forest Hills landtype association (LTA) and the rest within the Current River Pine-Oak Woodland Dissected Plain LTA (Nigh and Schroeder 2002). Topography is predominantly rolling with elevation ranging from 625-970 feet. Soils are mainly dry to dry-mesic, cherty Ultisols and Alfisols derived within residuum of sandstone and dolomite and have low base saturation.

Midco was established as an adaptive restoration project with the primary objective of developing a high-quality shortleaf pine woodland community. An additional objective of this study is to evaluate the use of prescribed fire, mechanical release, and herbicide release treatments as restoration tools. The first wave of restoration at Midco was a salvage harvest of dead and dying red oak throughout Midco from 1998-2003. The intensity of overstory removal varied substantially depending on local variation in the severity of red oak decline and presence of species targeted for retention (shortleaf pine, white oak (*Quercus alba*), and post oak (*Q. stellata*). Nearly all of Midco received a prescribed burn from 2001-2002. Since this first wave of extensive burning, treatments have been applied adaptively and selectively to portions of the project area in need of management action, including: partial overstory removal (2003), site preparation slashing and planting of shortleaf pine seedlings (2004-2007), mechanical release of shortleaf pine (2009 and 2011), herbicide release (2006 and 2009), and prescribed fire (2007 and 2010-2013). Monitoring plots (n=24) were used to help guide adaptive management at Midco. As a result of low sample size and targeted application of treatments, the network of monitoring plots does not cover the full range of treatments implemented to date. For this investigation, monitoring plots were partitioned into three categories based on treatment history: 1) burn only (n=5), 2) harvest and burn (HB; n=4), and 3) harvest, burn, and mechanical release (HBR; n=6). Most of the plots falling into HBR group were also treated with site preparation slashing after salvage harvesting and planted with shortleaf pine seedlings on a spacing of 12 feet x 12 feet. Unfortunately, the source of pine regeneration was not tracked in these monitoring plots so we were unable to determine the relative contribution of artificial versus natural pine regeneration.

The data for this study comes from 15, 0.2-acre vegetation monitoring plots that were randomly established in 2000. The diameter at breast height (dbh; taken at 4.5 feet) and species of all woody plants ≥ 4.5 feet tall and rooted within the 0.2-acre plot were recorded. Woody vegetation < 4.5 feet tall and herbaceous plants were captured using 12, 10.8 square-foot quadrats nested within each 0.2-acre plot; quadrats were located at 16.4-foot intervals along four transects radiating in cardinal directions from the plot center. All vascular plant species within each quadrat were identified to species and assigned to one of seven percent cover classes (<1, 1-5, 5-25, 25-50, 50-75, 75-95, and 95-100). Monitoring plot data were collected in 2000, 2002, 2006, 2009, and 2013.

Multiple response variables were used in this study to assess the treatments. The abundance of shortleaf pine relative to that of all tree species was calculated separately for four size classes: 1) overstory (dbh ≥ 4.5 inches), 2) large sapling (0.5 inch ≥ dbh < 4.5 inches), 3) small sapling (≥ 4.5 feet tall, dbh < 0.5 inch), and 4) seedling (< 4.5 feet tall). For overstory pine, relative density (stems per acre) and relative basal area were calculated as percentages of total tree density and

basal area, respectively. For the other size classes, only relative density was calculated. All herbaceous species were placed into one of four ecologically-based guilds: forest, woodland, generalist, and ruderal (Unpublished data. Susan Farrington. 2009. Natural History Biologist, Missouri Dept. of Conservation, 550 Joe Jones Blvd, West Plains, MO 65775). The woodland guild is composed of species that are more abundant in open-canopied woodlands and on drier exposed slopes, while the forest guild species are more abundant on shady, mesic, protected slopes in the Ozarks. Generalist species may occur on all sites, and ruderals are associated with recent disturbance. Shrubs and woody vines were placed into separate guilds. Each species was assigned to only one guild; tree seedlings were not included in the ground flora analysis. Ground flora species richness was calculated at the plot level by summing all of the species recorded in the 12 quadrats. Frequency was calculated at the plot level as the number of quadrats in which a species was recorded. We calculated Importance Values (IV) at the plot level for each guild group as the sum of the relative cover (based on cover class midpoints) and relative frequency of all species in each guild. The IV sums to 200 percent, but was re-scaled to 100 percent for ease of interpretation.

We used the Floristic Quality Index (FQI) to assess the floristic integrity (Ladd and Thomas, 2015). This index is based on assigning each species a coefficient of conservatism (CC) – a rating on a scale of 1-10 based on the species' tolerance for human disturbance and obligate predilection for intact habitats. Species with low CC values are weedy; species with high CC are more restricted to specific habitats and do not readily colonize disturbed ground. FQI is calculated by multiplying the mean CC by the square root of species richness.

We used PROC TTEST in SAS to test for differences among treatments within an inventory year and PROC MEANS to test for differences over time within each treatment individually. Because we were mainly interested in pine dynamics over the 14-year timeframe of this study, we limited our analysis of shortleaf pine relative abundance to data in 2000 and 2013. However, since ground flora often increase in richness and cover soon after disturbance, we also included data from 2006 to capture this short-term response. PROC MEANS was used to test for differences over time at the site level (i.e., all plots pooled together). Square-root or log transformations were used to meet distributional assumptions when necessary. Significance was set at $p=0.05$ for all analyses.

The control plots used by Rimer (2004) were not maintained as such, so we were not able to use them in this analysis. We summarized pretreatment conditions using data from a set of plots that had not yet been treated in 2000.

RESULTS

Tree Community Response

Prior to the start of restoration at Midco, mean basal area and stem density of the overstory were 77.1 square feet per acre and 136 stems per acre, respectively (data not shown). The overstory was dominated by oak species with red oak species (mainly black and scarlet oaks) constituting over half of the total basal area. In contrast, shortleaf pine accounted for approximately 10 percent of overstory basal area and stem density. White oak species (principally white and post oaks) comprised a much larger portion of total stem density of the large sapling layer than either red oak species or shortleaf pine. No shortleaf pine stems were recorded in the small sapling layer, and less than 1 percent of the seedling layer was shortleaf pine.

As expected, overstory basal area and stem density in plots treated with salvage harvesting were lower than in the burn only plots in 2000, mainly due to lower basal area of red oak species in harvested plots (figs. 1A and 1B). By 2013, basal area declined to 15 square feet per acre in HBR plots, which was driven largely by decreasing red oak basal area. Shortleaf pine basal area of all treatment groups increased over the 14-year period, while overstory density increased in harvested plots but not the burn only plots. No statistically significant differences ($p<0.05$; table 1) in the relative basal area (RBA) or relative density (RD) of overstory shortleaf pine were detected among treatment groups at the start of restoration in 2000. Overstory shortleaf pine basal area and stem density increased during the study period with the largest increases in HBR plots (figs. 1A and 1B). Pine RBA and RD for all plots combined was significantly greater in 2013 than in 2000 ($p=0.0245$ and 0.0480, respectively).

The densities of large and small saplings of all tree species increased substantially in the HBR plots (figs. 2A and 2B). Large saplings of shortleaf pine in these plots increased dramatically over the 14 years and, by 2013, averaged over 50 percent (122 stems per acre) of all large saplings at the plot level. RD of large pine saplings in the HBR plots increased significantly over the study period ($p=0.0171$; table 1) and was significantly higher than the burn only plots in 2013 ($p=0.0241$). Large sapling RD also significantly increased from 2000 to 2013 across all plots ($p=0.0125$). By 2013, RD of large pine saplings in the HB plots reached 25 percent (22 stems per acre), which was driven more by a large decrease in red oak density than increasing pine abundance. Pine only made up a small component of the small sapling and seedling layers of all treatments with the largest RD in HB plots in 2013

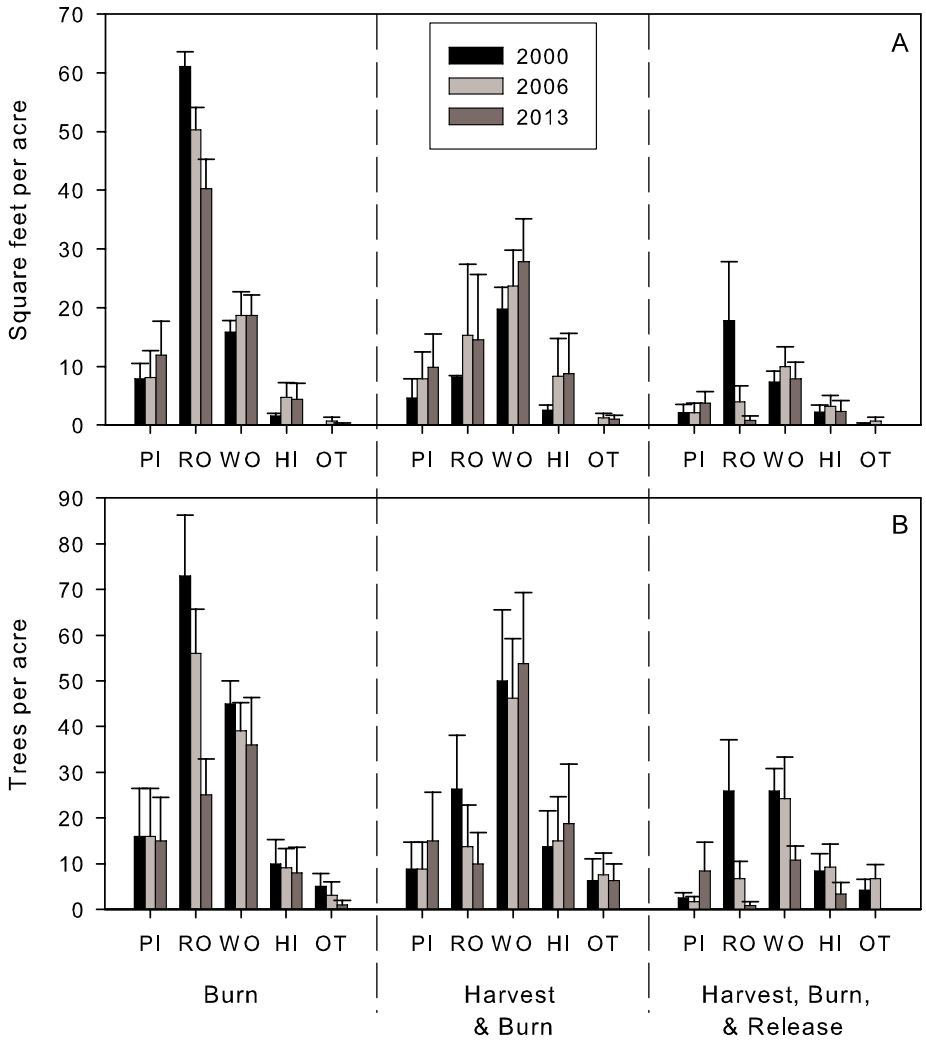


Figure 1—Mean basal area (A) and stem density (B) of five species within three treatment groups in the overstory of Midco in 2000, 2006, and 2013. Error bars equal one standard error. Species are: PI=shortleaf pine, RO=red oak group, WO=white oak group, HI=hickories, and OT=other (mainly black cherry, blackgum, sassafras, sumacs and maples).

(9.3 percent and 9.8 percent, respectively). Meanwhile, there were 5-6 fold increases in the density of other woody species (mainly black cherry (*Prunus serotina*), blackgum (*Nyssa sylvatica*), sassafras (*Sassafras albidum*), sumac (*Rhus* spp.), and maple (*Acer* spp.)) in the seedling layer of harvested plots, where they comprised over 14,000 stems per acre, or > 65 percent of all seedling stems, by 2013 (fig. 2C).

Ground Flora Response

Prior to treatments, woodland species were infrequently recorded and had low cover (data not shown). There was an average of 3.5 woodland guild species per 0.2-acre plot. The cover of the woodland guild averaged 1 percent per quadrat. The average IV of the woodland guild was 26 percent and the average FQI was 13.9 per plot.

By 2013, 27 new species (excluding tree species) were recorded that were not recorded in 2000. These new species were primarily native woodland flora such as bastard toadflax (*Comandra umbellata*), largebract tick trefoil (*Desmodium cuspidatum*), smooth tick trefoil (*D. laevigatum*), Maryland tick clover (*D. marilandicum*), Arkansas bedstraw (*Galium arkansanum*), hairy bedstraw (*G. pilosum*), hairy bush clover (*Lespedeza hirta*), prairie bush clover (*L. frutescens*), beebalm (*Monarda bradburiana*), hairy skullcap (*Scutellaria elliptica*), dropseed (*Sporobolus clandestinus*), and bashful bulrush (*Trichophorum planifolium*). New species in the study site also included two non-natives: Kentucky bluegrass (*Poa pratensis*) and sericea lespedeza (*Lespedeza cuneata*). A few woodland species were recorded in 2000 but were not found in 2013, including St. Andrew's cross (*Hypericum*

Table 1—Mean relative basal area (RBA) and relative density (RD) of shortleaf pine within four size classes by three treatment groups and for all groups combined at the start of restoration (2000) and the end of the 14-year study period (2013). Treatment group means within a size class and year, excluding "All groups", followed by a different letter are statistically different ($p<0.05$). In the 2013 column only, a mean followed by a plus (+) indicates a significant increase ($p<0.05$) over the study period for that treatment group

Size class	Treatment	2000	2013
Overstory RBA	Burn	8.7	14.4
	HB	13.7	18.6
	HBR	15.4	35.1
	All groups	12.6	22.7 +
Overstory RD	Burn	7.3	14.0
	HB	13.2	18.2
	HBR	10.9	34.3
	All groups	10.5	22.2 +
Large sapling RD	Burn	0.0	7.1b
	HB	4.4	25.0
	HBR	0.0	53.6 a +
	All groups	1.5	28.6 +
Small sapling RD	Burn	0.0	0.0
	HB	0.0	9.3
	HBR	0.0	2.9
	All groups	0.0	4.1
Seedling RD	Burn	3.8	1.6
	HB	0.0	9.8
	HBR	0.3	1.2
	All groups	1.4	12.6

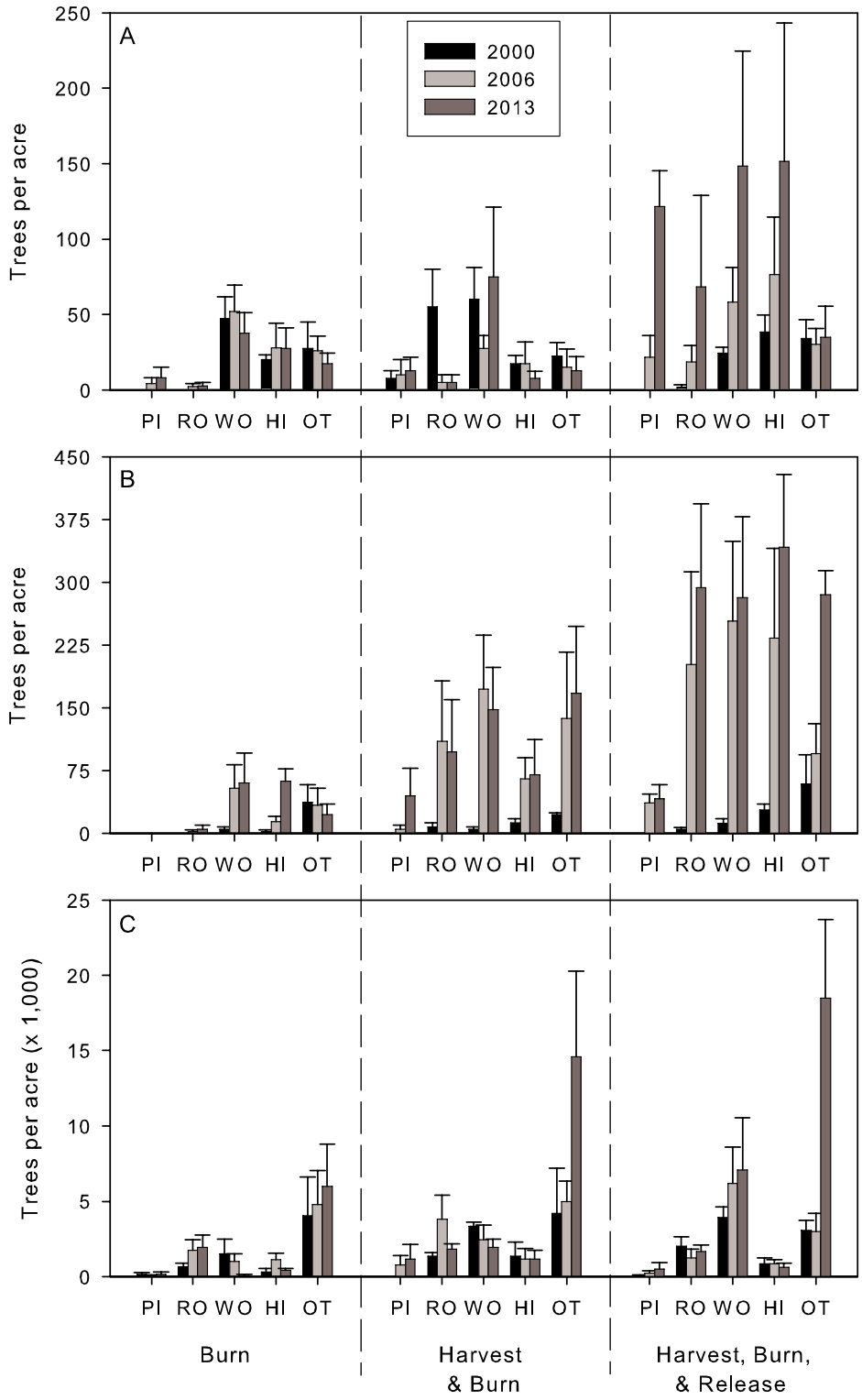


Figure 2—Mean stem density of large saplings (A), small saplings (B), and seedlings (C) of five species within three treatment groups in 2000, 2006, and 2013. Error bars equal one standard error. Species are: PI=shortleaf pine, RO=red oak group, WO=white oak group, HI=hickories, and OT=other (mainly black cherry, blackgum, sassafras, sumacs and maples).

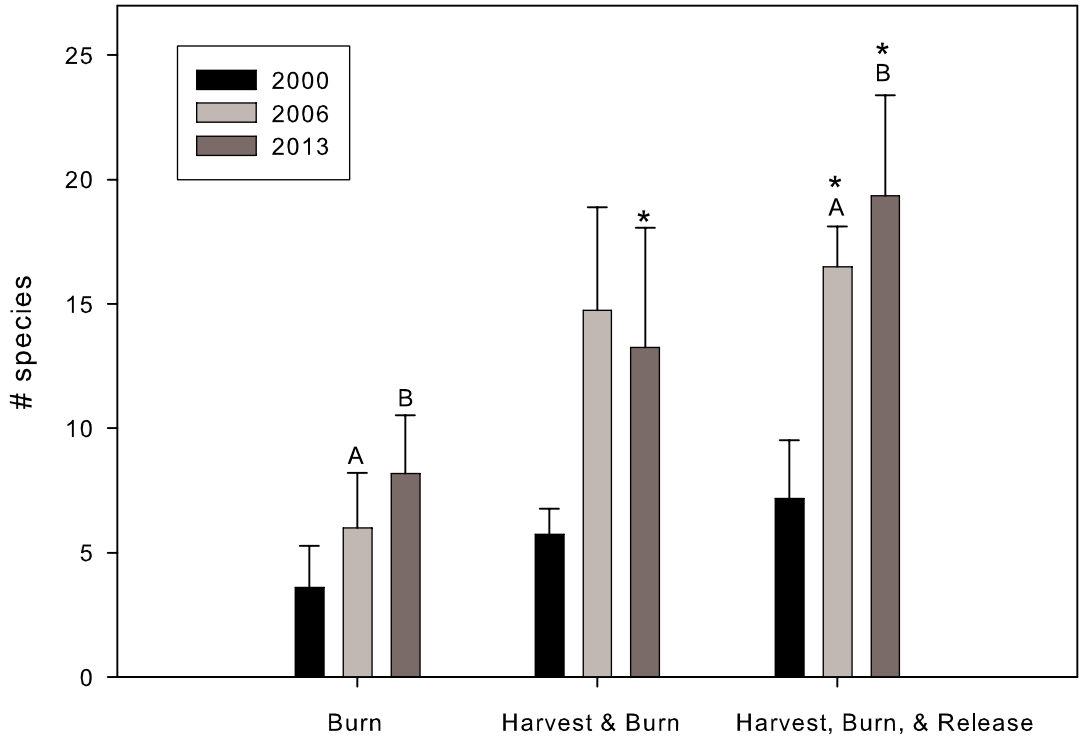


Figure 3—Woodland guild richness; bars are treatment means and error bars are one standard error. Within each year, the same letter indicates a significant difference between treatments. Within each treatment type, asterisks indicate a significant difference from the mean in 2000.

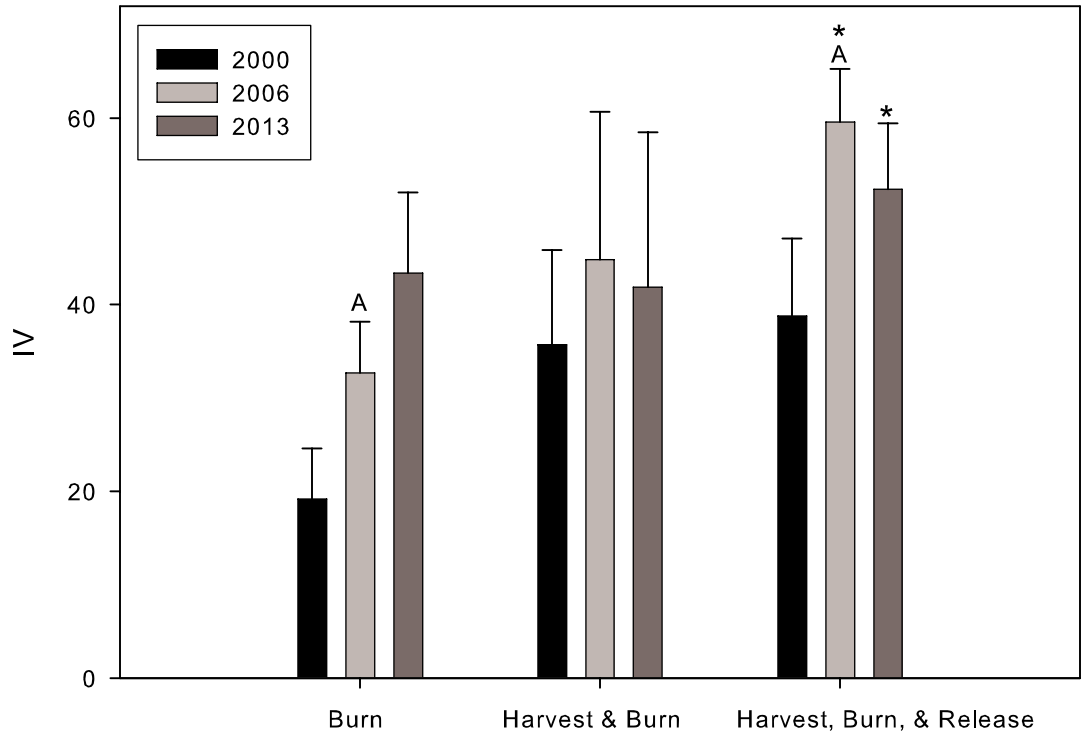


Figure 4—Woodland guild importance value (IV); bars are treatment means and error bars are one standard error. Within each year, the same letter indicates a significant difference between treatments. Within each treatment type, asterisks indicate a significant difference from the mean in 2000.

hypericoides), orange dwarf dandelion (*Krigia biflora*), sensitive brier (*Mimosa quadrivalvis*), and bird's foot violet (*Viola pedata*).

In 2000, woodland guild richness, woodland guild IV, and FQI were similar among the treatments (figs. 3, 4, and 5). Woodland guild richness remained low in the burn treatment, but increased significantly in the HB treatment in 2013 (p=0.0168), and in the HBR treatment in 2006 (p=0.0002) and in 2013 (p=0.0062) (fig. 3). The HBR plots gained significantly more woodland species than the burn plots in 2006 (p=0.0035) and 2013 (p=0.0506). Only the HBR treatment significantly increased the woodland IV in 2006 (p=0.0092) and 2013 (p=0.0472) (fig. 4). Pooling all plots together, the woodland IV increased significantly for the entire Midco restoration area in 2006 (40 percent IV, p=0.0007) and 2013 (44 percent IV, p=0.0048). Although we did not find significant changes in floristic quality between the treatments, the HBR treatment increased FQI in 2006 (p=0.0037) and 2013 (p=0.0303) (fig. 5). Pooling all plots together, FQI increased significantly for the entire Midco research area in 2006 (p=0.0003) and 2013 (p=0.0048).

DISCUSSION

Restoring the composition and structure of historic shortleaf pine woodland natural communities will likely require a long-term commitment of management resources on many sites in the Missouri Ozarks, especially where pine is a relatively minor component. At Midco, overstory pine made up < 10 percent of total basal area and < 5 percent sapling and seedling density at the start of this restoration project. Therefore, it is important to view the results of this study as early findings of a long-term restoration effort at Midco.

Despite significant increases in the pine RBA of all plots over the 14-year study period, the absolute BA of pine was still fairly low, ranging from 4 square feet per acre for HBR plots to 12 square feet per acre for burn plots. Recent estimates of pre-settlement structure of Ozark oak-pine woodlands suggest that basal area may have ranged from 80-130 square feet per acre (Hanberry and others 2014). A more conservative target for natural community management developed for the Mark Twain National Forest recommends that stocking of woodlands should fall between 30 percent and 90 percent, which corresponds to 30-125 square feet per acre (quadratic mean diameter of 7-22 inches) for upland central hardwood stands (Gingrich 1967). Applying this more conservative target, restoring overstory pine basal area to at least 30 square feet per acre will take time, especially areas treated with only prescribed fire.

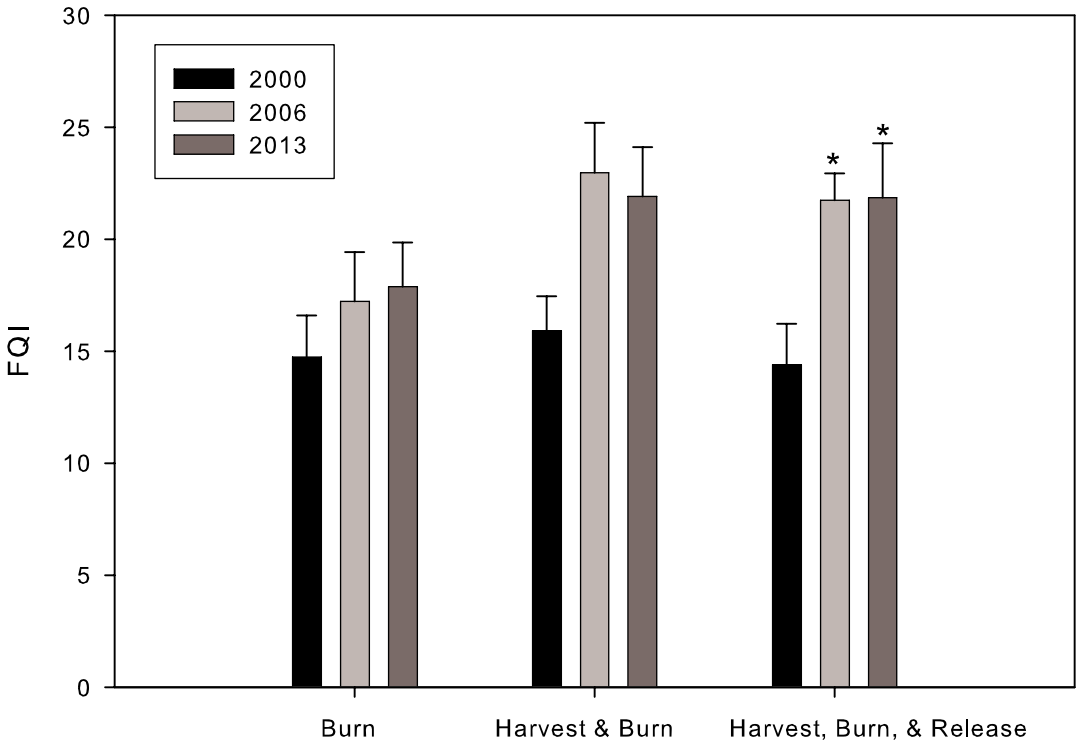


Figure 5—Floristic quality index (FQI); bars are treatment means and error bars are one standard error. Asterisks indicate a significant difference from the mean in 2000 within the same treatment type.

The greatest early gains in shortleaf pine densities occurred in the most heavily disturbed plots; those treated with harvesting, burning, and mechanical release. On these plots, the absolute and relative densities of large pine saplings increased to 122 stems per acre and 54 percent, respectively, by 2013. Most of this resulted from a pulse of ingrowth that took place between 2006 and 2013, which added 100 stems per acre compared to just 20 between 2000 and 2006. This pulse was likely initiated by mechanical release in 2009 and 2011. By 2013, burn only and HB plots supported just 8 and 13 large pine saplings per acre. HBR plots were also the most open stands based on basal area, which declined to only 15 square feet per acre. Shortleaf pine is a shade-intolerant species capable of rapid height growth of free-to-grow seedlings and saplings (Waldrop and others 1989). A higher rate of pine advancement into the large sapling class of the HBR plots was likely an outcome of both competitor release and lower overstory density. Most of the HBR plots also occur within areas treated with site preparation slashing and subsequently planted with nursery-grown shortleaf pine seedlings. Unfortunately, we were unable to determine the relative contribution of artificial versus natural pine regeneration in this study. However, it is likely the early success of HBR in promoting large sapling pine was likely facilitated by deliberate mechanical competition control around both artificial and natural shortleaf pine regeneration.

Our results suggest that a few burns alone (without the addition of mechanical treatments) were not sufficient to increase seedling or sapling shortleaf pine during the first 14 years of this project. Other studies have also shown that a single or even a few burns may not yield a desired understory vegetation response when starting with a mature, undisturbed forest. Elliott and Vose (2005) found that a single, low severity prescribed fire did not increase shortleaf pine establishment in a mature, southern Appalachian forest, which they partially attributed to a lack of fire-induced mortality of overstory trees and, consequently, a poor understory light environment for pine regeneration. Burn-only plots did have the highest pine BA but also maintained the highest total overstory basal area (>75 square feet per acre). According to Brinkman and Smith (1968), a minimum of 6-8 mature trees per acre is enough to regenerate shortleaf pine on suitable seedbeds using the seed-tree method. On average, burn-only plots had 16 overstory shortleaf pine per acre, suggesting that there was adequate density of seed producers for regeneration. Shortleaf pine germination is also limited by deep leaf litter (Grano 1949, Stambaugh and Muzika 2007), and it may take a series of burns to improve a seedbed for light-seeded species (Elliott and Vose 2005). Since shortleaf pine produces a good seed crop every 3-10 years (Lawson 1990), it is possible that transient improvement of the seedbed from fire was not followed by a good seed crop. This suggests that pine recruitment in the burn-only plots was limited more by seed availability, seedbed suitability, and light environment than the density of mature pine. Repeated burning may have also limited new pine establishment by killing pine seedlings.

The ground flora community responded positively to all restoration treatments, with the magnitude of response related to the intensity of treatments. Burning without additional treatment gradually increased the woodland composition and floristic quality, but the response over the 14 year period was not statistically significant. The effect of the HB treatment was statistically similar to both the burn and HBR treatments. HBR was the most effective in changing the species composition to woodland flora and increasing the floristic quality. Our findings are similar to those of Rimer (2004), who also found a significant increase in FQI on Midco in 2002 after burning and thinning.

CONCLUSIONS

Monitoring and evaluation are integral to a full understanding of the effectiveness of ecological restoration projects. Our analysis of monitoring data from Midco indicated that the treatments which caused the most intense disturbance, impacting both the overstory and understory, and also deliberately released both planted and natural pine regeneration, yielded the best short-term results. In contrast, the failure to enhance both pine regeneration and woodland ground flora after just a few burns highlights the importance of canopy disturbance to increase light to the understory for promoting a shortleaf pine woodland natural community. Elliott and Vose (2005) also concluded that mechanical overstory removal will likely be necessary to establish and recruit shortleaf pine in mature stands. Competition from hardwoods can also limit shortleaf pine regeneration and woodland herbaceous cover expansion. Pine seedling and small sapling densities in this study were vastly out-numbered by broadleaf shrubs and trees. This suggests that future release treatments will likely be needed to promote pine recruitment and the maintenance of woodland flora. However, the benefits of more intensive approaches should be weighed against the higher financial costs associated with more intensive treatment.

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LITERATURE CITED

Batek, M.J.; Rebertus, A.J.; Schroeder, W.A. [and others]. 1999. Reconstruction of early nineteenth-century vegetation and fire regimes in the Missouri Ozarks. Journal of Biogeography. 26: 397-412.

Blizzard, E.M. 2007. The state of mixed shortleaf pine-upland oak management in Missouri. In: Kabrick, J.M.; Dey, D.C.; Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 153-157.

Brinkman, K.A.; Smith, R.C. 1968. Managing shortleaf pine in Missouri. Stn. Bull. 875. Columbia, MO: University of Missouri Agricultural Experiment Station. 33 p.

Cunningham, R.J.; Hauser, C. 1989. The decline of the Missouri Ozark forest between 1880 and 1920. In: Waldrop, T.A., ed. Proceedings of pine-hardwood mixtures: a symposium on management and ecology of the type. Gen. Tech. Rep. SE-58. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 34-37.

Dey, D.C.; Hartman, G. 2005. Returning fire to Ozark Highland forest ecosystems: effects on advance regeneration. Forest Ecology and Management. 217: 37-53.

Elliott, K.J.; Vose, J.M. 2005. Effects of understory prescribed burning on shortleaf pine (*Pinus echinata* Mill.)/mixed-hardwood forests. Journal of the Torrey Botanical Society. 132(2): 236-251.

Gingrich, S.F. 1967. Measuring and evaluating stand density in upland hardwood forests in the Central States. Forest Science. 13: 38-53.

Grano, C.X. 1949. Is litter a barrier to the initial establishment of shortleaf and loblolly pine reproduction? Journal of Forestry. 47: 544-548.

Guyette, R.P.; Muzika, R.M.; Dey, D.C. 2002. Dynamics of an anthropogenic fire regime. Ecosystems. 5: 472-486.

Guyette, R.P.; Muzika, R.; Voelker, S.L. 2007. The historical ecology of fire, climate, and the decline of shortleaf pine in the Missouri Ozarks. In: Kabrick, J.M., Dey, D.C., Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 8-18.

Hanberry B.B.; Dey, D.C.; He, H.S. 2012. Regime shifts and weakened environmental gradients in open oak and pine ecosystems. PLoS ONE. 7(7): e41337.

Hanberry B.B.; Kabrick, J.M; He, H.S. 2013. Densification and the state transition across the Missouri Ozarks landscape. Ecosystems. 17(1): 66-81.

Hanberry B.B.; Jones-Farrand, T.D.; Kabrick, J.M. 2014. Historical open forest ecosystems in the Missouri Ozarks: reconstruction and restoration targets. Ecological Restoration. 32(4): 407-416.

Ladd, D. 1991. Reexamination of the role of fire in Missouri Ozark woodlands. In: Burger, G.V.; Ebinger, J.E.; Wilhelm, G.S., eds. Proceedings of the oak woods management workshop. Charleston, IL: Eastern Illinois University: 67-80.

Ladd, D.; Heumann, B.; Nelson, P. 2007. The Pineknot Project: restoration of a Missouri Ozark pinery. In: Kabrick, J.M., Dey, D.C., Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 204-205.

Ladd, D.; Thomas. J.R. 2015. Ecological checklist of the Missouri flora for floristic quality assessment. Phytoneuron. 12: 1-274.

Law, J.R., Melick, R., Studyvin, C., and Steinman, J.R. 2004. Oak decline in Missouri: History revisited. In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 257-258.

Lawson E.R. 1990. *Pinus echinata* Mill., shortleaf pine. In: Burns, R.M.; Honkala, B.H., tech. cords. Silvics of North America. Vol. 1. Conifers. Agric. Handb. 654. Washington, D.C.: U.S. Department of Agriculture: 316-326.

Little, S.; Moore, E.B. 1949. The ecological role of prescribed burns in the pine– oak forests of southern New Jersey. Ecology. 30: 223-233.

Masters, R.E. 2007. The importance of shortleaf pine for wildlife diversity in mixed oak-pine forests and in pine-grassland woodlands. In: Kabrick, J.M., Dey, D.C., Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 35-46.

Nelson, P. 2010. The Terrestrial Natural Communities of Missouri. Jefferson City, MO: Missouri Department of Natural Resources. 550 p.

Nigh, T.A.; Schroeder, W.A. 2002. Atlas of Missouri Ecoregions. Jefferson City, MO: Missouri Department of Conservation. 212 p.

Rimer, R. 2004. Project evaluates burning and timber harvest for restoring Ozark pine-oak woodlands (Missouri). Ecological Restoration. 22(3): 219-220.

Stambaugh, M.C.; Muzika, R. 2007. Successional status of six mature shortleaf pine forests in Missouri. In: Kabrick, J.M., Dey, D.C., Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 59-67.

Waldrop, T.A.; Loyd, F.T.; Abercrombie Jr., J.A. 1989. Fell and burn to regenerate mixed pine-hardwood stands: an overview of research on stand development. In: Waldrop, T.A., ed. Proceedings of pine-hardwood mixtures: a symposium on management and ecology of the type. Gen. Tech. Rep. SE-58. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 75-82.

Hardwood Regeneration

Moderator:

Brian Lockhart
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RESPONSE OF PLANTED NORTHERN RED OAK SEEDLINGS TO REGENERATION HARVESTING, MIDSTORY REMOVAL, AND PRESCRIBED BURNING

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Abstract—Oak (*Quercus*) is difficult to naturally regenerate in many mature oak stands on productive sites in the southeastern United States, and artificial regeneration alternatives should be considered. Artificial regeneration can potentially restore or enrich the oak component at the stand level. We examined genetic and silvicultural effects on artificially regenerated northern red oak (*Quercus rubra*) seedlings three years after planting under three silvicultural prescriptions and a control. We used quality-grown seedlings from open-pollinated families to improve probabilities of success. The seedlings averaged 101 cm in height and 11.2 mm in root-collar diameter at the time of planting. Genetic differences were significant for survival and growth, but these differences may have been due to a residual nursery effect. Families with large seedlings at the time of planting were generally larger and had better survival after three years than families with smaller seedlings at the time of planting. A commercial shelterwood harvest was the only successful silvicultural treatment for artificial regeneration in this study. Trees planted in this treatment grew a total of 41 cm in height and 8.1 mm in ground-line diameter in three years. Seedlings planted in uncut stands, whether stands had been burned, treated with a midstory removal, or left untreated, had relatively poor survival (30 to 72 percent) and negligible growth (≤15 cm height, ≤2 mm ground-line diameter).

INTRODUCTION

In the southern United States, upland oak (*Quercus*) forests are an important resource, but have been declining due to regeneration failures and mortality of aging overstory trees over the last several decades (Abrams 2003, McEwan and others 2011). The loss of oak species can have drastic ecological and economic effects, particularly in southeastern forests (Oswalt and others 2009). Natural regeneration methods have been tested, but often require a series of noncommercial treatments and many years to increase the density of large oak seedlings (i.e., advanced reproduction) (Arthur and others 2012, Loftis 1990). Artificial regeneration can be used to supplement natural oak regeneration, but we currently have limited knowledge on how silvicultural treatments affect planted oak seedlings.

The idea that larger seedlings will perform better than smaller seedlings has been tested for many decades (reviewed in Dey and others 2008). However, recent developments in nursery technology to produce seedlings with relatively large aboveground and belowground systems (e.g., high-quality seedlings; Kormanik and others 2002) have gone relatively untested, particularly on productive upland sites in the southeastern United States. Furthermore, testing of genetic effects and interactions in silvicultural studies is rare. The objective of this study was to test quality-grown northern red oak (*Quercus rubra*) seedlings planted in three silvicultural treatments and a control while accounting for variation associated with genetics of seedling seed source. Results will provide managers with information on how to best use limited resources to artificially regenerate oaks on productive sites in this region.

METHODS

The study area was located in the Blue Ridge Mountain physiographic region of North Carolina on Cold Mountain Game Lands, owned and managed by the North Carolina Wildlife Resources Commission. Site characteristics were described by Keyser and others (in press). Experimental units consisted of 5-ha mature hardwood forests. Stands were similar in structure

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and averaged 32 m^2/ha in basal area in trees greater than 15 cm diameter at breast height (dbh). Treatments were implemented prior to planting and included a prescribed burn (Rx burn), a midstory removal (MR), and a commercial shelterwood harvest (SW). An untreated control was also included. All treatments were replicated three times using a completely random design. Two replications of the Rx burns were conducted in February 2009 and one was conducted in April 2010 prior to planting. Rx burns were ignited using drip torches and were set as backing fires with flanking strip head fires. The MR was conducted according to prescriptions described by Loftis (1990) in September 2008. With the exception of oak and hickory (*Carya*) species, competing midstory trees ≥5.0 cm and <25.0 cm dbh were injected with herbicide Garlon® 3A. Basal area in the control and in the Rx burn units decreased 5 percent, and basal area in the MR units decreased 15 percent. The SW basal area decreased 63 percent, leaving mostly dominant and codominant oak stems.

Northern red oak acorns were collected from six open-pollinated mother trees located in stands of the Pisgah National Forest in the Blue Ridge physiographic region in autumn 2008. Acorns were sown at a density of 65/m^2 separately by family, and the resulting seedlings were grown as 1-0 bare-root seedlings at the East Tennessee Nursery in Delano, TN, using prescriptions developed to produce high-quality seedlings (Kormanik and others 2002). Trees were lifted in March 2010 by family and visually graded to select the largest 30 percent from each seed lot (based primarily on root-collar diameter) to improve seedling quality (Clark and others 2000).

We used a randomized complete block design with single tree plots, and 14 blocks were planted in each experimental unit (84 trees). A total of 168 trees per family (1008 seedlings total) were planted using 3 by 3 m spacing. We collected survival, stem height, and ground-line diameter (GLD) data just after planting (April 2010) and for three years after planting (growing seasons 2010-2012). Data were analyzed using a general linear mixed model (LM) to determine the effects of silvicultural treatment, year since planting, and genetic family on height and GLD. A generalized linear mixed model (GLMM) was used to analyze the effects of treatments on survival (alive=1, dead=0) for each year after planting. We specified a binary response distribution with a logit link function, and GLMMs were modelled on event=1. If main effects of treatments were significant in the LMs and GLMMs, we computed comparisons using Tukey's mean separation method.

RESULTS AND DISCUSSION

Silvicultural treatment and family affected survival, height, and GLD after three growing seasons (table 1 and 2). The interactions between family and silvicultural treatment for survival were significant probably because one family (NRO11) had higher survival than other families in the SW treatment each year after planting. Families had similar survival in all other silvicultural treatments. The two-way interactions between treatment and family were not significant for height or GLD (table 2). The three-way interaction among year, treatment, and family was significant for height because some families performed better in some treatments in some years and performed similarly in some treatments in other years.

When averaged across treatments, NRO11 generally had higher survival than other families each year after planting (fig. 1). This family also exhibited relatively large height and GLD in all years, including at the time of planting (fig. 2). Families maintained similar height and GLD growth rankings over time (fig. 2). These results indicate that larger seedlings at planting will maintain size advantages over smaller seedlings and will have improved survival, which has been shown in other studies (reviewed in Dey and others 2008 and 2012). Family effects in this study, therefore, may be confounded with a nursery effect because seedling size attributes from the nursery could mask true genetic differences after planting (Pinto and others 2011). Our family seed lots were not replicated at the nursery, and

Table 1—Generalized linear mixed model for survival for the first three years after planting northern red oak

	Treatment		Family		Treatment*Family	
Year	F	P	F	P	F	P
1	69.57	<0.0001	67.05	<0.0001	139.45	<0.0001
2	94.51	<0.0001	82.33	<0.0001	130.84	<0.0001
3	78.39	<0.0001	68.61	<0.0001	112.4	<0.0001

initial size advantages exhibited by certain families could be due to nursery growing conditions for that family seed lot (e.g., distance to nearest watering riser and elevation of bed). However, the size advantages could also be related to acorn size at sowing, which is an inherited trait (Kormanik and others 1998, Korstian 1927). Genetic differences in early field performance of northern red oak in the nursery and after planting is a well-known phenomenon (Kriebel 1965), and family rankings can change over time (Schlarbaum and Bagley 1981, Kriebel and others 1988). Managers should use a genetically diverse seed mix to avoid the chance of planting only one or two families that produce small seedlings in the nursery or poor performing families in the field. Families had similar survival in the noncommercial treatments (Rx burn, MR, and control) for each year after planting, indicating genetic and/or seedling size differences could not be discerned in low light environments where growth was negligible.

Seedlings in the SW treatment had the highest survival and had the largest height and diameter after three growing seasons (fig. 3 and table 3). In fact, the SW treatment was the only treatment to have significant

Table 2—General linear mixed models with repeated measures for height and ground-line diameter (GLD) three years after planting northern red oak

		Height		GLD	
Effect	Denominator DF	F	P	F	P
Treatment	8	6.85	0.0134	11.24	0.0031
Family	818	18.78	<0.0001	9.52	<0.0001
Treatment*Family	818	1.45	0.1163	1.01	0.4371
Year	3	90.98	<0.0001	330.17	<0.0001
Year*Treatment	1893	27.09	<0.0001	126.75	<0.0001
Year*Family	1893	1.65	0.0539	1.58	0.0723
Year*Treatment*Family	1893	1.50	0.0179	0.67	0.9575

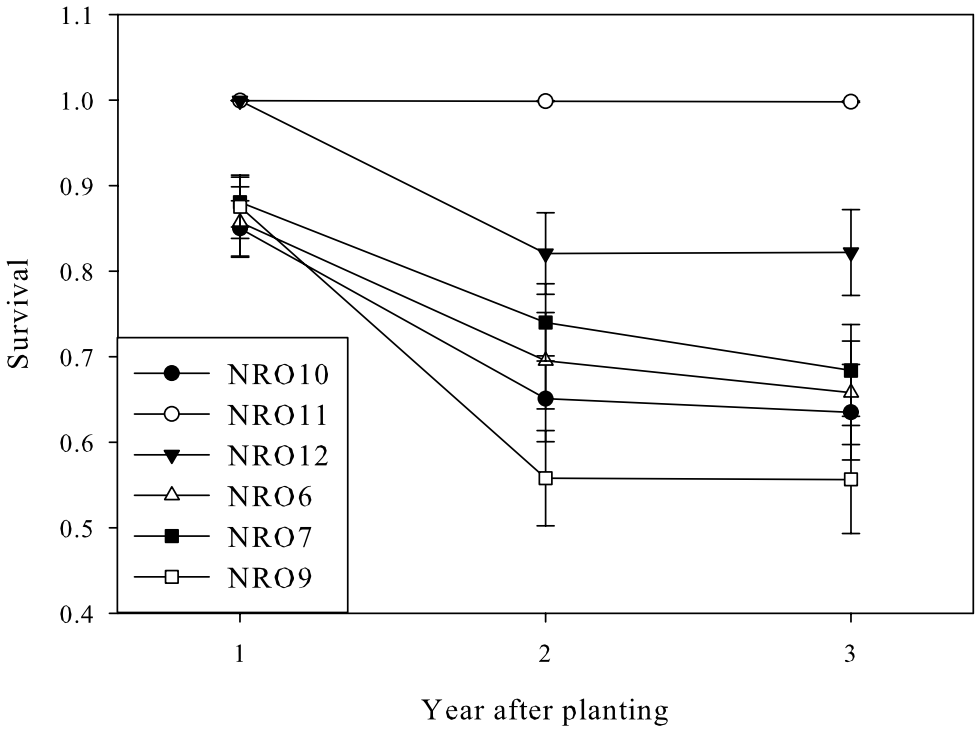


Figure 1—Survival least-squares means three years after planting six genetic families of northern red oak.

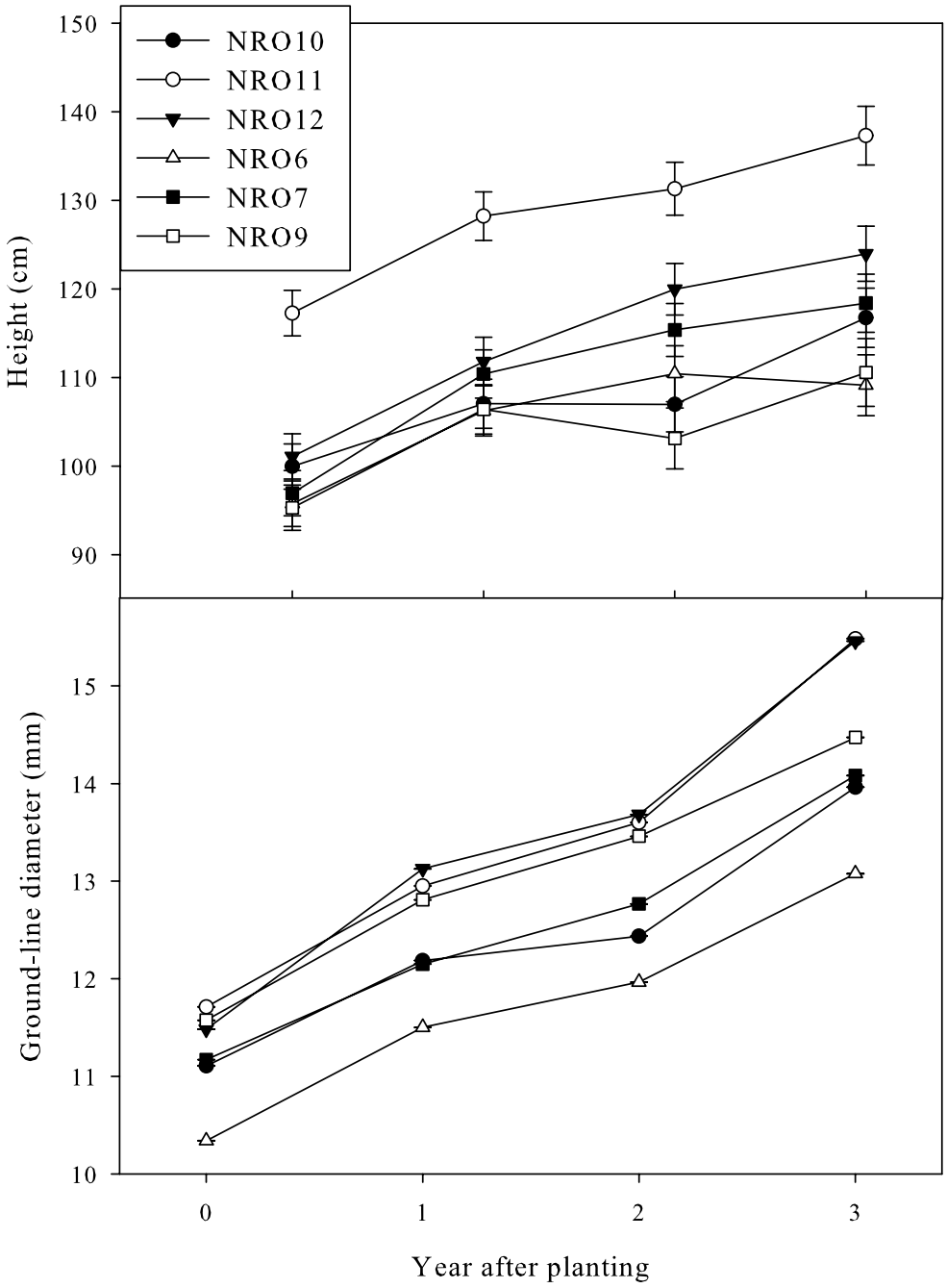


Figure 2—Height and ground-line diameter least-squares means three years after planting six genetic families of northern red oak.

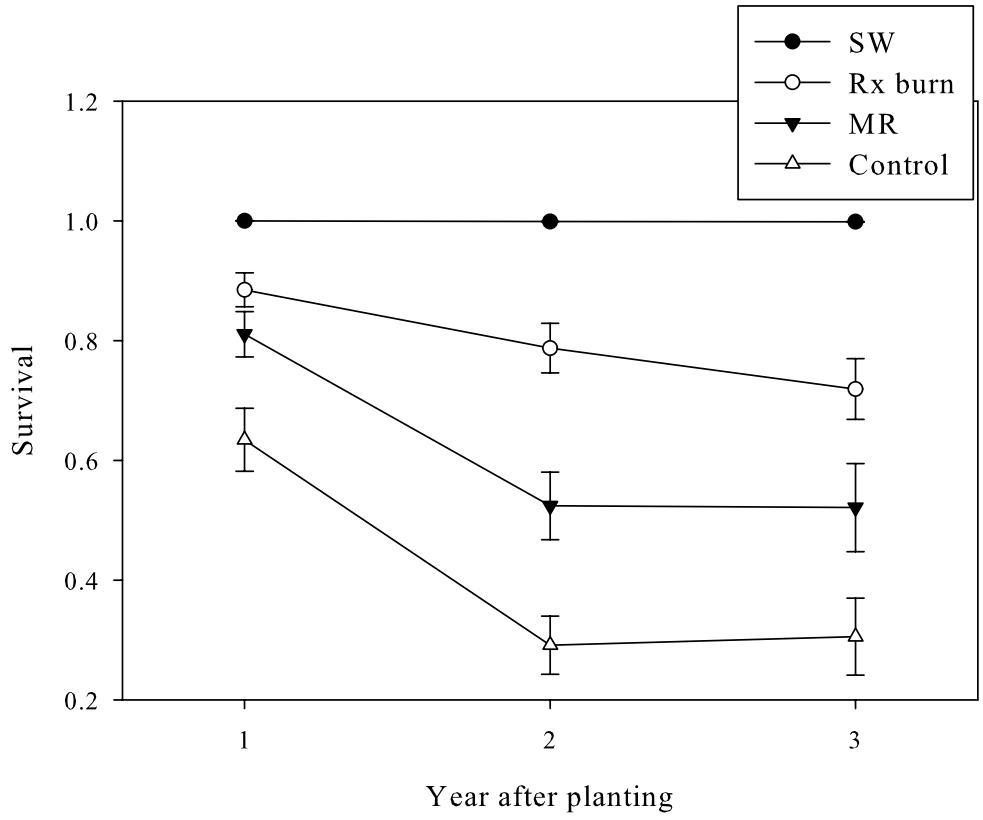


Figure 3—Survival least-squares means for three silvicultural treatments and a control three years after planting northern red oak. SW = commercial shelterwood harvest, Rx burn = prescribed burn, and MR = midstory removal.

Table 3—Height and ground-line diameter (GLD) least-square means at the time of planting and for the first three years after planting northern red oak. Means followed by the same letter are not significantly different

	Year 0		Year 1		Year 2		Year 3	
				Height (cm)				
Control	99	fgh	105	cdefg	102	defg	104	cdefg
MR	101	gh	113	bcdef	113	bcdef	113	bcdef
Rx burn	101	fgh	118	bcd	119	bc	116	bcde
SW	103	eh	112	cdfg	123	b	144	a
				GLD (mm)				
Control	11.2	gjkl	12.6	cdefhi	12.5	cdefhi	13.2	cdef
MR	11.4	fkl	12.1	dij	12.2	cdeghi	12.9	cegh
Rx burn	11.0	il	12.4	cdefgh	11.9	ejk	12.7	cdfgh
SW	11.2	hijk	12.7	cdefg	15.4	b	19.3	a

MR = midstory removal, Rx burn = prescribed burn and SW = commercial shelterwood harvest.

yearly height growth after the first year (table 3). Three-year survival was 99 percent in the SW stands, and trees grew an average of 41 cm and 8.1 mm in height and GLD, respectively. The improved survival and growth in the SW treatment over other treatments was expected because growth rates increase as canopy openness increases from relatively low levels (Johnson 1984, Lhotka and Loewenstein 2009, Morrissey and others 2010, Spetich and others 2002).

The Rx burn and MR treatments had statistically similar survival rates in years 1 and 3 after planting despite a 20-percent higher survival rate in the Rx burn stands. The control had the lowest survival rates, averaging 30 percent in year 3; however, this was not statistically different than the MR treatment (52 percent). The lack of statistical differences in survival among treatment means was due to high variation among replications, and indicates more replications of treatments were needed. Variation among experimental units with the same treatment could be due to differences in timing of treatments for the Rx burn units, or it could be related to site differences among replications of the same treatment (e.g., aspect, elevation, soil moisture availability, and nutrient availability). Exploratory analysis using potential explanatory variables should be conducted to elucidate explanation for the relatively high variation in survival.

Despite the low power of the study, the low survival in the MR treatment would be considered unacceptable for most commercial planting operations. We suspect that the slight increases in survival in the Rx burn treatment over the control and MR treatments could be related to nutrient increases from burning organic matter prior to planting (Blankenship and Arthur 1999). In contrast to this study, other studies have found oak seedlings had adequate survive in noncommercially treated underplantings up to seven years after planting with slight increases in light availability (Buckley and others 1998, Lhotka and Loewenstein 2009, Parrott and others 2012, Schweitzer and others 2006). However, seedlings planted in this study were larger than these previous studies, which may have reduced their ability to maintain proper root/shoot ratios under a low light environment (Struve and others 2000). Oswalt and others (2006) also found reduced survival (58 percent) in uncut stands after two years compared to harvested stands (>80 percent) for seedlings sized similarly to those in our study.

Seedlings in the noncommercial treatments had relatively poor growth rates compared to the SW treatment (table 3). The MR, the Rx burn, and the control treatments did not grow in height from year 1 to year 3, and the MR treatment was the only noncommercial treatment that had significant GLD growth during this time. The poor growth in the control was similar to previous studies that found oak seedlings did not grow under full canopy conditions (Buckley and others 1998, Parrott and others 2012). In contrast to our study, large oak seedlings (~100 cm height) grew 26 cm in height in control stands after two years in a study in western Tennessee (Oswalt and others 2006). The height growth reported by Oswalt and others (2006), however, could have been from new sprouts originating after stem dieback, and may not represent increases in total height (as we measured). The slightly improved GLD growth found in the MR treatment compared to the control treatment was expected because removal of midstory or understory competition above natural or planted seedlings improves growth, particularly of the root system (Lhotka and Loewenstein 2009, Loftis 1990, Parrott and others 2012, Paquette and others 2006). The relatively poor growth of seedlings in the Rx burn was probably due to low available light. The burns were highly variable and were of relatively low intensity overall. Additionally, midstory stems deadened by the fire produced understory sprouts within a year of the burns, which probably increased understory shade to planted seedlings.

If we use GLD as a surrogate for root growth (Grossnickle 2012), trees had more relative growth in the root system compared to stem growth in the SW and control treatments. For example, trees had a 40-percent increase in relative stem height and a 72-percent increase in relative GLD in the SW stands from the time of planting until year 3. The seedlings in the control grew 5 percent in height and 18 percent in GLD over three years. In contrast, the MR and the Rx burn treatments had GLD-relative growth rates similar to their height-relative growth rates (approximately 12 and 15 percent for the MR and Rx burn treatments, respectively). Oak seedlings rebuild root systems damaged during lifting and planting at the expense of stem growth, a process known as transplant shock (Struve and others 2000). Results suggest that at the two extremes of disturbance in this study, with the most extreme in the SW and the least extreme in the control, seedlings built root systems at the expense of aboveground biomass during the early years when undergoing planting shock. Seedlings planted under the intermediate disturbance regimes of the MR and Rx burn treatments recovered their root systems while simultaneously building aboveground biomass in similar proportions. Unfortunately, information comparing growth of oak seedlings planted under a gradient of disturbance regimes is limited. Kolb and Steiner (1990) found direct-seeded northern red oak had relatively balanced root/shoot ratios when grown with an intermediate light intensity and relatively high root/shoot ratios when grown in full sun.

MANAGEMENT IMPLICATIONS

We propose that the MR and control treatments as applied in this study are not viable treatments for artificial regeneration of northern red oak due to low survival and growth rates after the first three growing seasons. The Rx burn was marginally successful in terms of survival, but growth was negligible. The most efficacious treatment was the SW treatment; seedlings were able to recover from transplant shock and grow significantly each year, resulting in a 14-cm gain in height and a 2.7-mm gain in GLD per growing season. We expect height and GLD growth to improve in the SW stands as trees age and root systems fully recover from transplant shock, assuming no major future stress to the trees (Struve and others 2000).

Family differences in survival and growth were significant, but genetic differences may have been confounded with a nursery effect. Regardless, results indicate managers should use a genetically diverse seed mix from locally adapted sources to avoid the possibility of planting only poor-performing families.

A portion of the SW and the Rx burn treatments were recently burned again as part of management prescriptions for this study. Recovery from burning will be important to follow because managers are increasingly using fire to improve oak regeneration (Arthur and others 2012). Predictions that survival and growth after burning will depend on seedling size at planting are currently being tested.

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LITERATURE CITED

Abrams, M.D. 2003. Where Has All the White Oak Gone? BioScience. 53: 927-939.

Arthur, M.A.; Alexander, H.D.; Dey, D.C [and others]. 2012. Refining the oak-fire hypothesis for management of oak-dominated forests of the eastern United States. Journal of Forestry. 110: 257-266.

Blankenship, B.A.; Arthur, M.A. 1999. Soil nutrient and microbial response to prescribed fire in an oak-pine ecosystem in eastern Kentucky. In: Stringer, J.W.; Loftis, D.L. eds., Proceedings of the 12th central hardwood forest conference. Gen. Tech. Rep. SRS-24. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 39-47.

Buckley, D.S.; Sharik, T.L.; Isebrands, J.G. 1998. Regeneration of northern red oak: positive and negative effects of competitor removal. Ecology. 79(1) 65-78.

Clark, S.L.; Schlarbaum, S.E.; Kormanik, P.P. 2000. Visual grading and quality of 1-0 northern red oak seedlings. Southern Journal of Applied Forestry. 24: 93-97.

Dey, D.C.; Gardiner, E.S.; Schweitzer, C.J. [and others]. 2012. Underplanting to sustain future stocking of oak (*Quercus*) in temperate deciduous forests. New Forests. 43: 955-978.

Dey, D.C.; Jacobs, D.F.; McNabb, K [and others]. 2008. Artificial regeneration of major oak (*Quercus*) species in the eastern United States - a review of the literature. Forest Science. 54: 77-106.

Grossnickle, S.C. 2012. Why seedlings survive: influence of plant attributes. New Forests. 43: 711-738.

Johnson, P.S. 1984. Responses of planted northern red oak to three overstory treatments. Canadian Journal of Forest Research. 14: 536-542.

Keyser, T.L.; Greenberg, C.H.; Simon, D.; Warburton, G. [In press]. Assessing the regeneration potential of productive mixed-hardwood stands following single and repeated prescribed fire. In: Proceedings of the 18th biennial southern silvicultural research conference. Gen. Tech. Rep. Asheville, NC: U.S. Department of Agriculture Forest Service Southern Research Station.

Kolb, T.E.; Steiner, K.C. 1990. Growth and biomass partitioning of northern red oak and yellow poplar seedlings: effects of shading and grass root competition. Forest Science. 6: 34-44.

Kormanik, P.P.; Sung, S.J.S.; Kormanik, T.L. [and others]. 1998. Effect of acorn size on development of northern red oak 1-0 seedlings. Canadian Journal of Forest Research. 28: 1805-1813.

Kormanik, P.P.; Sung, S.J.S.; Zarnoch, S.J.; Tibbs T. 2002. Artificial regeneration of northern red oak and white oak on high-quality sites: effect of root morphology and relevant biological characteristics. In: Sambeek, J.W.; Dawson J.O.; Ponder F. Jr. [and others] eds. Proceedings of the 2001 national silviculture workshop. Gen. Tech. Rep. PNW-546. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station: 83-91.

Korstian, C.F. 1927. Factors controlling germination and early survival in oaks. New Haven, CT: Yale University School Forestry Bulletin. 115 p.

Kriebel, H.B. 1965. Parental and provenance effects on growth of red oak seedlings. In: Proceedings of the 4th central states forest tree improvement conference: 19-25.

Kriebel, H.B.; Merritt, C.; Stadt, T. 1988. Genetics of growth rate in *Quercus rubra*: provenance and family effects by the early third decade in the north central USA. Silvae Genetica. 37: 193-198.

Lhotka, J.M.; Loewenstein, E.F. 2009. Effects of midstory removal on understory light availability and the 2-year response of underplanted cherrybark oak seedlings. Southern Journal of Applied Forestry. 33: 171-177.

Loftis, D.L. 1990. A shelterwood method for regenerating red oak in the Southern Appalachians. Forest Science. 36: 917-929.

McEwan, R.W.; Dyer, J.M; Pederson, N. 2011. Multiple interacting ecosystem drivers: toward an encompassing hypothesis of oak forest dynamics across eastern North America. Ecography. 34: 244-256.

Morrissey, R.C.; Jacobs, D.F.; Davis, A.S.; Rathfon, R.A. 2010. Survival and competitiveness of *Quercus rubra* regeneration associated with planting stocktype and harvest opening intensity. New Forests. 40: 273-287.

Oswalt, C.M.; Clatterbuck, W.K.; Houston, A.E. 2006. Impacts of deer herbivory and visual grading on the early performance of high-quality oak planting stock in Tennessee, USA. Forest Ecology and Management. 229: 128-135.

Oswalt, C.M.; Oswalt, S.N.; Johnson, T.G. [and others] 2009. Tennessee's Forests, 2009. Res. Bull. SRS-189. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 136p.

Paquette, A.; Bouchard A.; Cogliastro A. 2006. Successful under-planting of red oak and black cherry in early-successional deciduous shelterwoods of North America. Annals of Forest Science. 63: 823-831.

Parrott, D.L.; Lhotka J.M.; Stringer J.W.; Dillaway D.N. 2012. Seven-year effects of midstory removal on natural and underplanted oak reproduction. North Journal of Applied Forestry. 29: 182-189.

Pinto, J.R.; Dumroese R.K.; Davis A.S.; Landis T.D. 2011. Conducting seedling stocktype trials: a new approach to an old question. Journal of Forestry. 109: 293-299.

Schlarbaum, S. E.; Bagley, W.T. 1981. Intraspecific genetic variation in *Quercus rubra* L., northern red oak. Silvae Genetica. 30:5056.

Schweitzer, C. J.; Gardiner, E. S.; Loftis D. L. (2006). Response of sun-grown and shade-grown northern red oak seedlings to outplanting in clearcuts and shelterwoods in North Alabama. In: Conner, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 269-274.

Spetich, M.A.; Dey, D.C.; Johnson, P.S.; Graney, D.L. 2002. Competitive capacity of *Quercus rubra* L. planted in Arkansas Boston Mountains. Forest Science. 48: 504-517.

Struve, D.K.; Burchfield, L.; Maupin, C. 2000. Survival and growth of transplanted large- and small-caliper red oaks. Journal of Arboriculture. 26: 162-169.

INFLUENCE OF FLOODING, FREEZING, AND AMERICAN BEAVER HERBIVORY ON SURVIVAL OF PLANTED OAK SEEDLINGS

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Abstract—Good seedlings, proper planting, and competition control normally result in successful hardwood planting. However, other factors can have serious impact on planting success, such as the impact of flooding, freezing, and the American beaver (*Castor canadensis*). In 2014, three planting stocks of Nuttall oak (*Quercus nuttallii*) and Shumard oak (*Quercus shumardii*) were planted on two sites in southern Mississippi and survival checks were conducted monthly. The first check revealed that only three seedlings of the containerized stock had survived. Subsequent nursery inquiries revealed that many containerized seedlings died from freezing over the unusually cold winter. During the second check, it was discovered that beavers had uprooted many seedlings and consumed the roots at one site. Rain gauge data confirmed that excessive rainfall had resulted in site flooding and seedlings remained underwater for a portion of the month of May. After the immediate loss from beaver damage, Shumard oak seedlings suffered an additional loss over both remaining planting stocks due to the extended period of inundation. Most of the loss caused by inundation was large potted seedlings. Overall, 99.8 percent mortality occurred in one planting stock due to freezing, 22 percent of mortality was due to beaver damage, and 33 percent of mortality was due to inundation in the remaining planting stocks.

INTRODUCTION

Oak (*Quercus spp.*) seedlings were planted in the spring of 2014 to evaluate and compare survival and growth of different planting stocks of oaks as part of ongoing research focusing on oak restoration on Hurricane Katrina damaged lands throughout southern Mississippi and Louisiana. Planting stock survival is the most important factor in seedling establishment of any species, and it was immediately apparent that the 2014 planting would experience exceptionally low survival rates. Survival was so abnormally low that a separate study was implemented to quantify and identify the sources of seedling loss. With artificial regeneration using high quality seedlings, proper planting, and competition control, mortality is normally low (Self and others 2010). These variables can be controlled when implementing reforestation/afforestation processes. Uncontrollable and out-of-the-ordinary circumstances affecting seedling survival may also arise, particularly in the first growing season. Factors impacting this research were documented as flooding, freezing, and American beaver (*Castor Canadensis*) herbivory.

OBJECTIVES

1. To evaluate survival of different planting stocks of oak seedlings.
2. To identify causes of seedling loss and quantify results for each cause.

MATERIALS AND METHODS

Site Description/ Site Preparation

The two sites utilized in this study are separated by approximately 60 miles. Site one (Odom Site) is located in the northeastern corner of Perry County, MS, approximately six miles north-northeast of Richton, MS. It is a bottomland site surrounded by small streams, but little to no flooding occurs. The Odom Site was previously used for pasture land containing an extensive herbaceous layer growing on soil types Savannah and Stough fine sandy loams (WSS 2015). Site preparation included subsoiling to break the hardpan from livestock compaction. Site two (Welford Site) is located in eastern George County, MS, near the Alabama state line on the Escatawpa River approximately 14 miles southeast of Lucedale, MS. It experiences periodic flooding but usually on a very small scale. The Welford Site was previously a mixed hardwood stand on mostly Lenoir silt loam with a small amount of Harleston fine sandy loam (WSS 2015) near the river. Site preparation consisted of using a bulldozer to clear debris.

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Seedlings

Two oak species were evaluated in this study: Shumard oak (*Quercus shumardii*) and Nuttall oak (*Quercus nuttallii*). Three planting stocks of each species were used including high-quality 1-0 bareroot, conventional 240ml containerized, and 3.8L large potted seedlings. The bareroot stock was produced by Rayonier nursery in Elberta, AL. Mossy oak Native Nursery in Osborn, MS, produced the conventional containerized stock. Resource Environmental Solutions (RES) native tree and coastal marsh grass nursery in Montegut, LA, was contracted by Restore the Earth Foundation to grow and plant the large potted stock. Mississippi State University (MSU) personnel planted both bareroot and conventional containerized stocks in the middle of February. The commercial planting crew planted the large potted seedlings on April 1, with a MSU researcher on site for supervision.

Plot Establishment

A complete randomized block design comprised of three replicates of 100 tree units for each planting stock/species combination (treatment) was used. The Odom Site was planted on a 10'x10' spacing, while the Welford site had to be condensed into an 8'x8' spacing. HWC (herbaceous weed control) was applied to bareroot and containerized stock. No HWC was applied for the large potted stock type, because they are marketed for a plant and walk away approach.

Survival

Survival checks were taken monthly for the first growing season. Trees were not recorded as dead until no green was present on any leaf or in the cambium. Freeze damage was determined by a combination of deduction methods. American beaver herbivory was determined by missing seedlings that were previously healthy before the flood event, and shoots that were left on site with teeth marks that were cut by the animal. Inundation deaths were determined by trees that were weakened then died within two survival checks post flood event.

RESULTS AND DISCUSSION

Freezing

During the first survival check, 99.8 percent or 1,197 out of 1,200 of the conventional containerized stock were discovered dead, a total loss for the purpose of this research. No other planting stock was affected indicating that site conditions could be ruled out as the cause. Slight sub-freezing temperatures do not usually affect seedlings because the roots are not exposed due to the growing medium acting as an insulator. In north Mississippi, January temperatures were recorded below freezing (32 °F) for 25 days, with 4 days (two, 2 day periods) temperature never rising above freezing. During that month 10 days were below 20°F, 6 days below 15 °F, and 3 days below 10 °F. This is unusually low as the 30-year average low is 37 °F. The conclusion of the involved parties was that such deep freezes at night, many days not warming enough to thaw the soil, and small container size allowed the roots to freeze in the nursery containers. Evidence of freezing is not noticeable until seedlings try to break dormancy, and it should be noted that all seedlings appeared normal and healthy at time of planting.

Flooding

Eighteen inches of rain were recorded on the Welford site in April, with eight inches occurring in two days. The Escatawpa River runs approximately 350 feet from the planting area with an elevation change of 11.5 feet to the lowest seedlings and 14 feet to the highest seedlings. The average depth of the river is 4 feet, but during April it rose to over 18 feet according to NOAA flood data (AHPS 2014). Some local residents unofficially measured the water depth at 21 feet during the flooding period. With the proximity to the river and slight elevation change of the site, the rising river water reached the seedlings quickly. The seedlings were inundated to depths ranging from 4 feet-6.5 feet during this period. The flood contributed to mortality by soil displacement, allowing easy access to American beaver, and preventing gas exchange.

Soil displacement—Water covered the site allowing soil particles to break apart more easily and be displaced. In turn, the displaced soil contributed to the root system not being able to support the above ground portion of the seedling. Large potted seedlings have a low root to shoot ratio that already allows the seedling to be overturned easily. The flow of water along with debris pushed approximately 50 of these seedlings over and uprooted 32 others, which were all Nuttall oak. Nuttall oak had a much larger above ground mass and were, therefore, more susceptible to the flooding damage. The loose soil also allowed the American beaver to more easily pull the bareroot stock roots out of the ground.

American beaver—When planting oak seedlings, the normal rodent concern is small rodents girdling the seedlings at or below ground level (Schreiber and Swihart 2009, Tyler and others 2008). American beavers can be a greater concern planting if certain conditions are present. American beaver herbivory seems to be specific to age, time of year, and water level for oak seedlings. They prefer first-year seedlings in late winter/early spring when water is shallow in the area (Krinard and Johnson 1981). It appears that American beaver prefer first-year bareroot seedlings because they are newly planted and have softer roots. The roots have not had time to become well established, flooding reduces any bond with the soil thereby leaving them vulnerable to being pulled out. The preferred time of year is when alternative food sources are scarce, and there is a short

growth period from planting date. American beavers also chewed off large potted seedlings but evidence of root consumption could not be found. Thirty-three percent of all bareroot mortality on the Welford site was directly related to beaver, while only 11 percent of large potted mortality could be attributed to them.

Gas exchange—Inundating plants with water restricts atmospheric exchange of gases. Plants rely on roots and leaves to exchange carbon dioxide and oxygen needed for photosynthesis (Anderson and Pezeshki 1999). Some species can slow respiration while flooded for periods of time then resume once water has receded, some species growth will be greatly reduced even after the water has receded, and others will never resume respiration once flooded. Oaks can function for different periods with their roots covered in water, but not under full inundation. In their first year, seedlings that are overtopped by floodwaters for more than three weeks during growing season will die (Walker and Oswald 2000). However, tolerance to flooding varies among species of oaks. Nuttall oak has shown only about 2 percent mortality that can be directly linked to complete inundation in this study while Shumard oak had greater than 40 percent.

Overall Survival

On the Odom site where no flooding occurred, overall survival was 84 percent excluding containerized stock and 56 percent including containerized stock. On the Welford site where flooding occurred, overall survival was 28 percent excluding containerized stock and 19 percent including containerized stock. Table 1 provides details of survival by species and planting stock for each site. A complete loss of containerized stock from freezing occurred on both sites, totaling 33 percent of all mortality. Twenty-two percent of mortality on the Welford site was directly caused by American beavers. Table 2 presents mortality for the Welford site by cause, species, and planting stock.

CONCLUSION

Nuttall oak has proven to be more resilient than Shumard oak on the bottomland sites in this study.

Table 1—Survival percentages of seedlings by site, species, and planting stock for 2014 growing season (excluding containerized stock)

	Odom Site (no flood)	Welford Site (with flood)
Bareroot overall	84 %	31 %
Shumard oak	69 %	34 %
Nuttall oak	99 %	28 %
Large potted overall	83 %	25 %
Shumard oak	75 %	6 %
Nuttall oak	91 %	43 %

Table 2—Mortality percentages of seedlings on Welford Site by cause, species, and planting stock for 2014 growing season (excluding containerized stock)

	Soil displacement	American beaver	Inundation
Bareroot overall	—	33 %	11 %
Shumard oak	—	21 %	22 %
Nuttall oak	—	45 %	1 %
Large potted overall	5 %	11 %	38 %
Shumard oak	—	16 %	77 %
Nuttall oak	5 %	6 %	3 %

Overall Nuttall oak growth exceeded that of Shumard oak on both sites whether flooded or not flooded. The most likely reason is that Nuttall oak has adapted to growing in wet areas by allocating growth to height while young thus allowing their tops to exceed flood levels as fast as possible. Flooding can contribute to seedling mortality in other ways than just inundation to seedling mortality. American beaver can be detrimental to seedling survival in the first growing season if flooding occurs in late winter to early spring. Study results also underscore the importance of nursery management when overwintering containerized seedlings during sub-freezing temperatures. Nurseries should take precautions to keep roots from freezing. Consideration should be taken of as many of these extraordinary conditions as possible that affect survival and growth while planning successful artificial oak regeneration.

LITERATURE CITED

Advanced Hydrologic Prediction Service. [database on the internet]. National Oceanic Atmospheric Administration. u2014. Available from: http://water.weather.gov/ahps/. [Date accessed: July 10, 2014].

Anderson, P.H.; Pezeshki, S.R. 1999. The effects of intermittent flooding on seedlings of three forest species. Photosynthetica. 37: 543-552.

Krinard, R.M.; Johnson, R.L. 1981. Flooding, beaver, and hardwood seedling survival. Gen. Tech. Rep. RN-270. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 6p.

Schreiber, L.A.; Swihart, R.K. 2009. Selective feeding of pine voles on roots of tree seedlings. Canadian Journal of Zoology. 87: 183-187.

Self, A.B.; Ezell, A.W.; Londo, A.J.; Hodges, J.D. 2010. Evaluation of Nuttall oak and cherrybark oak survival by planting stock and site preparation treatment type in a WRP planting on retired agricultural site. In: Stanturf, J., ed. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-121. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 159-163.

Tyler, C.M.; Davis, F.W.; Mahall, B.E. 2008. The relative importance of factors affecting age-specific seedling survival of two co-occurring oak species in southern California. Forest Ecology and Management. 255: 3063-3074.

Walker, L.C; Oswald, B.P. 2000. The southern forest: geography, ecology, and silviculture. Boco Raton, FL: CRC Press LLC. 83 p.

Web Soil Survey. [database on the internet]. Version 3.1. U.S. Department of Agriculture Natural Resources Conservation Service. u2013-Available from: http://www.websoilsurvey.sc.egov.usda.gov/APP/WebSoilSurvey.aspx. [Date accessed: February 3, 2015].

SEVEN YEAR EFFECTS OF MEADOW VOLE HERBIVORY ON OAK SURVIVAL

Andrew B. Self[1]

Abstract—Seedling mortality due to meadow vole herbivory is often thought to be small scale in nature in hardwood afforestation efforts. However, in some instances, this source of mortality may play a more important role than typically realized. A total of 1,440 bare-root Nuttall oak (*Quercus texana* Buckley), Shumard oak (*Quercus shumardii* Buckley), and swamp chestnut oak (*Quercus michauxii* Nutt.) seedlings were planted in February 2008 on a northwest Mississippi site. Plots received treatment with one of four mechanical site preparation methods. Also, all plots received an initial post planting application of Oust XP®. In year two, one half of plots were selected for treatment with a second application of Oust XP®. Overall, seedling survival was excellent throughout the duration of the study. However, cumulative annual vole induced seedling mortality was significant and did not stabilize until the seventh year after planting. Overall seventh-year survival was 83.7 percent, with vole damage accounting for approximately 79.1 percent of all seedling/sapling mortality to date. Analyses did not detect treatment interaction with vole induced mortality in the first three years of this study. However, seventh year analyses detected significant main effect differences for both mechanical treatment and species related to vole herbivory. Vole herbivory was greater in areas receiving less intensive mechanical treatments, and Nuttall oak was preferentially selected for herbivory. While seedling survival approaching 84 percent is not normally considered poor, at 12.9 percent mortality, the level of vole damage observed in this study is of serious magnitude and may warrant consideration in planting efforts.

INTRODUCTION

Meadow vole or pine vole (*Microtus pinetorum* LeConte) herbivory is one mortality factor sometimes associated with forest plantings across the eastern United States (Ostfeld and Canham, 1993). Meadow voles (also known as pine voles) are semifossorial, arvicoline rodents found in woodlands and other habitats across the eastern United States. Meadow vole herbivory is typically subterranean, resulting in seedling root systems being eaten below the root collar (Schreiber and Swihart 2009). Several studies have observed that meadow voles may selectively feed on roots of oak seedlings (Ostfeld and Canham 1993, Rathfon and others 2008, Schreiber and Swihart 2009). Mortality levels as high as 19 percent were noted by Rathfon and others (2008) in southern Indiana for white oak, northern red oak, and black oak seedlings under mature, closed-canopy, oak-dominated forests. The highest levels of meadow vole-induced seedling mortality were found in areas that had undergone midstory removal. Self and others (2015a) detected no survival differences due to vole herbivory among four mechanical site preparation treatments or three oak species in three-year-old oak plantations.

Other studies have shown increased frequency of meadow voles in areas with greater levels of herbaceous vegetation due to midstory and overstory removal (Perry and Thill 2005, Schreiber and Swihart 2009). Increased ground cover provides better habitat and serves to aid in increased meadow vole numbers in these settings (Birney and others 1976). Afforestation attempts on retired agriculture fields may be hindered due to protection of voles from predation on these sites resulting from the greater levels of herbaceous vegetation that these sites typically provide (Buell and others 1971, Gill and Marks 1991, Ostfeld and Canham 1993, Self and others 2015a). While typically not a major concern in afforestation attempts, vole herbivory can reach levels with substantial impact to planting success.

MATERIALS AND METHODS

Site Description

This study site is located approximately 5 miles northwest of Coldwater, MS in Desoto County on the Arkabutla Lake Project owned by the U.S. Army Corps of Engineers. The site was in soybean [*Glycine max* (L.) Merr.] production until September 2007. Soils were silt loams, and 40-year average precipitation was 56.1 inches (NOAA 2015). Soil tests indicate that the site had an average pH of 6.2. Dominant herbaceous species on site at study initiation were Brazilian vervain (*Verbena brasiliensis* Vell.), poorjoe (*Diodia teres* Walt.), and thorny amaranth (*Amaranthus spinosus* L.). Twenty-

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one other herbaceous species were observed in small quantities across the site. Cumulative herbaceous coverage of all species was approximately 5 percent.

Experimental Design

This experiment utilized a split-split-plot design with whole plot factors in a randomized complete block design and sub-plot factors completely randomized within whole plot factors and sub-sub-plot factors completely randomized within sub-plot factors. The whole plot factor was site preparation treatment. The sub-plot factor was species. The sub-sub-plot factor was herbaceous weed control (HWC). The experimental unit was a plot with its unique combination of site preparation treatment, species, and HWC. The response variable was seedling mortality resulting as a direct result of vole herbivory.

Three blocks containing all possible site preparation treatment/species/HWC combinations were established. Each block consisted of 12 planting rows split horizontally resulting in the creation of 24 plots. The experimental unit was the plot which was approximately 190 by 10 feet and contained 20 seedlings. For logistical reasons, site preparation treatments were applied singularly as a group; however, these groups were randomized within each block. Individual species were planted by row for each site preparation treatment. Species were randomized by site preparation treatment. Each row was divided into two plots with each plot receiving a different HWC treatment assigned randomly.

Mechanical Site Preparation and Herbaceous Weed Control

Four mechanical site preparation treatments were employed: control (no site preparation), subsoiling, bedding, and combination plowing. Site preparation treatments were applied on 10 foot centers. Subsoil trenches were cut to a depth of 15 inches. 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.

HWC treatments included a one year application and a two year application of Oust XP®. Both treatments were applied in 5 foot wide bands using a rate of 2 ounces of product per sprayed acre and were applied over the top of seedlings prior to bud break. The one year Oust XP® application was applied during March 2008. The two year Oust XP® application was applied during March 2008 and March 2009. A Solo® backpack sprayer was used for herbicide application with total spray volume of 10 gallons per acre.

Seedling Establishment

Nuttall oak, Shumard oak, and swamp chestnut oak were chosen for use in this study. Seedlings were lifted mid-January 2008, and seedling specifications required 1-0 seedlings of overall vigorous appearance with relatively intact root systems. Specified seedling parameters dictated that stems be 18 to 20 inches tall and possess root systems 8 to 10 inches long with a minimum of eight first-order lateral roots (FOLRs). Seedlings not meeting these specifications were culled before planting. A total of 480 seedlings of each species were planted at root collar depth during February 2008 by university personnel using a 10 foot spacing.

Survival Measurements

Seedling/tree survival and cause of death was determined and recorded annually by ocular evaluation during winter from 2008 to 2014. Seedling/tree bases were examined to determine if vole herbivory was the causal agent for mortality. Seedlings dying from vole herbivory were recorded as such.

Data Analysis

All statistical analyses were performed using a mixed procedure Statistical Analysis System version 9.3.1 (Cary, NC). A mixed model analyses of variance (ANOVA) was used to test for effects and interactions. Data were analyzed using least square means (LSMEANS). Survival percentages were arcsine square root transformed for normalization purposes. This transformation was necessary to convert the binomial distribution of the data to one that is nearly normal. No differences were detected between HWC treatments, however analysis detected significant main effect differences among mechanical treatments and species. While transformed survival data were used for analyses, actual means are presented for interpretation. Means were considered significant if $P < 0.05$.

RESULTS AND DISCUSSION

On this same study site, analysis by Self and others (2015a) did not detect survival differences linked to vole herbivory among species, mechanical site preparation, HWC treatments, or their interactions at year three. Overall tree survival at this point was 95.6 percent with vole herbivory accounting for approximately 75 percent of oak mortality. However, tree survival continued to drop in years four through seven, with 3.9 percent survival reductions in 2011 and 2012, and a 4 percent in 2013. In 2014, overall, mortality seemingly plateaued at 83.7 percent with a 0.1 percent increase in mortality from 2013 observations. Approximately 79.1 percent of tree mortality observed throughout the duration of this study resulted from meadow vole herbivory. Analysis of year seven survival data did not detect differences among HWC treatments.

Survival by Mechanical Treatment

Analysis detected a significant main effect survival difference among mechanical treatments ($p = 0.0134$). Trees grown in areas receiving more intensive mechanical site preparation exhibited less vole induced mortality compared to those grown in areas receiving less intensive treatments (table 1). Year seven cumulative mortality in bedded or combination plowed areas was lower than that observed in both control and subsoiled areas (2.78, 2.99, 3.47, and 3.68 percent, respectively). This result was somewhat unexpected; the expectation being greater vole herbivory/tree mortality in the more friable disturbed soils of the bedded and combination plowed treatment areas.

A possible explanation for this deviation in expected tree mortality results may lie in increased root production of seedlings/trees growing in more intensively treated areas. In earlier analyses of this same study, Self and others (2015b) found seedlings in bedded and combination plowed areas possessed root systems with approximately 58 percent greater root biomass compared to seedlings growing in control and subsoiled areas. Additionally, overall height and diameter seedling parameters were greater in more intensively treated areas. It is reasonable to conclude that increased seedling/tree size in more intensively treated areas resulted in more vigorous trees, better equipped to survive vole herbivory.

Survival by Species

Analysis detected a significant main effect difference in survival among species ($p = <0.0001$) due to vole herbivory. Of the three species planted, vole induced mortality was greatest in Nuttall oak followed by Shumard and swamp chestnut oaks (9.51, 2.29, and 1.11 percent, respectively) (table 2). The finding of preferential selection of Nuttall oak by meadow voles is surprising as no differences were detected in earlier analyses of years one through three survival data (Self and others 2015a). Nuttall oak is a faster growing species compared to Shumard and swamp chestnut oaks (Burns and Honkala 1990). It is possible that greater inherent overall vigor of the species and associated nutrient rich root growth flushes served to attract meadow voles to Nuttall root systems compared to those of Shumard and swamp chestnut oak. Additionally, it is entirely possible that Nuttall oak possesses some nutrient highly desired by voles.

Table 1–Cumulative seventh year vole induced oak mortality by mechanical site preparation treatment

Mechanical Treatment	Cumulative Mortality
	----------percent----------
Control	3.47a[a]
Subsoiling	3.68a
Bedding	2.78b
Combination Plowing	2.99b

[a]values followed by different letters are different at the $\alpha = 0.05$ level.

Table 2—Cumulative seventh year vole induced oak mortality by oak species

Species	Cumulative Mortality
	----------percent----------
Nuttall oak	9.51a[a]
Shumard oak	2.29b
Swamp chestnut oak	1.11c

[a]values followed by different letters are different at the $\alpha = 0.05$ level.

MANAGEMENT IMPLICATIONS

Meadow vole-induced mortality of planted seedlings is a known variable, but is ordinarily thought to be a minor factor in afforestation attempts. While typically not a major concern in afforestation efforts, vole herbivory can reach levels with substantial impact to planting success. Furthermore, vole herbivory is typically thought to be relatively nonexistent after the third or fourth growing season. Oak mortality due to vole damage reached levels of greater concern for this study in frequency and seedling age. Most land managers evaluate planting success during the first three years after plantation establishment. In some situations it is possible that the greatest levels of vole damage do not manifest until after this point. Additionally, of the three tree species tested, the finding that Nuttall oak seems to be preferred compared to other oaks may be of particular concern in afforestation attempts.

CONCLUSIONS

Assuming that vole-killed seedlings had survived and not died via some other unforeseen factor, overall seventh year tree survival would have been approximately 96.6 percent. Conventional wisdom dictates that upon entering the third or fourth growing season, vole herbivory should be a lessening problem regarding continued seedling/tree mortality. Vole-induced seedling mortality did not follow this traditional trend in this study. Additionally, the preferential selection of Nuttall oak stems by meadow voles could have far reaching impacts. Due to its quick growth and elasticity regarding suitable planting sites, Nuttall oak has been one of the primary oak species planted in hardwood afforestation efforts over the past few decades. Considering the acreages currently planted and scheduled for plantation establishment, preferential selection of this species by meadow voles warrants consideration in planting efforts.

LITERATURE CITED

Birney, E.C.; Grant, W.E.; Baird, D.D. 1976. Importance of vegetative cover to cycles of *Microtus* populations. Ecology. 57: 757-780.

Buell, M.F.; Buell, H.F.; Small, J.A.; Siccama, T.G. 1971. Invasion of trees in secondary succession on the New Jersey Piedmont. Bulletin of the Torrey Botanical Club. 98: 67-84.

Burns, R.M.; Honkala, B.H. 1990. Silvics of North America. Vol. 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture. 877 p.

Gill, D.S.; Marks, P.L. 1991. Tree and shrub seedling colonization of old fields in central New York. Ecological Monographs. 61(2): 183-205.

NOAA [National Oceanic and Atmospheric Administration]. 2015. Precipitation totals. Washington, DC: U.S. Department of Commerce National Climatic Data Center. Available at http://www.noaa.gov. [Date accessed: May 5, 2015].

Ostfeld, R.S.; Canham, C.D. 1993. Effects of meadow vole population density on tree seedling survival in old fields. Ecology. 74(6): 1792-1801.

Perry, R.W.; Thill, R.E. 2005. Small-mammal responses to pine regeneration treatments in the Ouachita Mountains of Arkansas and Oklahoma, USA. Forest Ecology and Management. 219: 81-94.

Rathfon, R.A.; Lichti, N.I.; Swihart, R.K. 2008. Disking and mid-and understory removal following an above-average acorn crop in three mature oak forests in southern Indiana. In: Jacobs, D.F.; Michler, C.H., eds. Proceedings of the 16th central hardwood forest conference. Gen. Tech. Rep. NRS-P-24. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 59-69.

Schreiber, L.A.; Swihart, R.K. 2009. Selective feeding of pine voles on roots of tree seedlings. Canadian Journal of Zoology. 87: 183-187.

Self, A.B.; Ezell, A.W.; Rowe, D. [and others]. 2015a. Meadow vole (*Microtus pinetorum*) induced mortality of oak seedlings in a former agricultural field planting. In: Holley, G.A.; Conner, K.F.; Haywood, J.D. Proceedings of the 17th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-203. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 295-300.

Self, A.B., Ezell, A.W., Rowe, D. [and others]. 2015b. Mechanical site preparation and Oust XP effects on stem biomass in three-year-old Nuttall oak seedlings planted on a former agricultural field. In: Holley, G.A.; Conner, K.F.; Haywood, J.D. Proceedings of the 17th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-203. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 57-61.

THE POTENTIAL OF USING COPPICE GROWTH AS TRAINING TREES IN PLANTATIONS FOR THE PRODUCTION OF HIGH-QUALITY OAK BOLES

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Abstract—Oaks (*Quercus* spp.) grown in monoculture plantations often do not develop high-grade boles because all the trees grow at similar rates and crown stratification does not occur resulting in persistent branches and poor bole quality. The use of trainer species in mixed species plantations could promote interspecific competition and crown stratification leading to higher grade oak boles. As part of a larger, long-term study investigating growth, crown interactions, and bole development of cherrybark oak (*Quercus pagoda*) with several species at different planting densities, yellow-poplar (*Liriodendron tulipifera*) and cherrybark oak were planted together. Yellow-poplar has a much faster growth rate and overtops the slower growing cherrybark oak. After four growing seasons at 6- x 6-feet spacing, yellow-poplar averaged 17-feet tall, while cherrybark oak averaged 8-feet tall. Yellow-poplar crowns were overtopping the cherrybark oak. Yellow-poplar was thinned (64 percent of yellow-poplar trees) and allowed to coppice. Two years following the thinning, yellow-poplar coppice growth averaged 4.9-feet tall. This paper reports on yellow-poplar coppice growth and the potential ability of the coppice to have a training effect on the growth, development, and bole form of adjacent cherrybark oaks.

INTRODUCTION

Most hardwood plantations in the southeastern United States have been established and managed as monocultures following the conifer plantation model where growth (volume/weight per unit area) is maximized (Oswalt and Clatterbuck 2011). However, the greatest value of hardwoods is producing high-quality sawtimber (clear wood with minimal defects) rather than just producing fiber (Ashton and Ducey 1996). The conifer plantation model for monocultures promotes uniformity where a single species from similar genetic or geographic sources grow at similar rates with similar sizes. This single species plantation approach does not appear to provide for the development of defect-free, quality hardwood trees similar to those found in mixed species, natural hardwood stands where species stratification occurs. The issue is whether high-quality sawtimber can be produced in plantations of mixed species following natural stand development patterns (Ashton and Ducey 1996, Clatterbuck and Hodges 1988, Larson 1992, Oliver 1982). Many theories, assumptions and suppositions have been suggested such as spatial arrangement, differential growth rates, species tolerance, and varying site productivities to simulate natural stand dynamics in plantations (Kelty 2006). However, few long-term examples of mixed species stand dynamics for hardwood plantations are available in the southern United States to substantiate stand conditions that influence development of a greater volume of clear, knot-free wood (higher quality logs) in plantation settings (Lockhart and others 2012, Oswalt and others 2011).

Afforestation of degraded stands and former agricultural areas has generated increased interest in mixed species plantings as being more environmentally preferable to monocultures, especially in hardwoods (Bristow and others 2006). Mixed species plantings could better simulate development of mixed species natural stands, provide more variety of flora and fauna, and create greater structural forest diversity (Oliver and Larson 1996). Most natural hardwood stands are composed of and developed with mixed species rather than single species monocultures.

The results in this paper are a subset of a much larger research study focused on the growth relationships and species interactions of three species (yellow-poplar (*Liriodendron tulipifera*), black cherry (*Prunus serotina*), and sweetgum (*Liquidambar styraciflua*)) each grown with cherrybark oak (*Quercus pagoda*) at three spacings (6 x 6 feet, 8 x 8 feet, and 10 x 10 feet). Each species mixture by spacing combination was replicated three times in 0.25-acre units (0.20-acre units for the 6- x 6-feet spacing) to evaluate stand development patterns as well as the growth, bole form and grade of cherrybark oak as influenced by these different species and spacings. Preliminary results of cherrybark oak –

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yellow-poplar mixtures at the 6- x 6-feet spacing are presented where 64 percent of the yellow-poplar trees were thinned and allowed to coppice or sprout once the yellow-poplar was overtopping the cherrybark oak. On most productive sites for these species, yellow-poplar will outgrow and overtop slower growing, adjacent cherrybark oak within a few years (Clatterbuck 2011). The purpose is to evaluate the coppice growth of yellow-poplar and its present influence on bole and grade development of cherrybark oak. Will the coppice growth provide a complementary training effect to the cherrybark oak trees or will the coppice growth be great enough to impact the crown expansion and perhaps result in another overtopping of cherrybark oak? Two year coppice data after thinning are presented.

METHODS

Study Site

The study was conducted on the Cumberland Forest, a unit of the University of Tennessee's Forest Resources Research and Education Center (FRREC) in Morgan County, TN. The Cumberland Forest is located in the Wartburg Basin sub-region of the Cumberland Mountains (Smalley 1984). The landscape encompasses high, rugged mountains consisting of narrow crests and long steep slopes often broken with narrow benches and short sandstone cliffs. Small areas of relatively level, smooth land occur on mountain tops, benches, footslopes and stream valleys. The study area is in a narrow, winding valley on one of these footslopes. Elevation averages 1800 feet in the Wartburg Basin. Elevation at the study site is 1350 feet. The soils are Lonewood silt loams, which are fine-loamy, siliceous, semiactive, mesic, Typic Hapludults (National Cooperative Soil Survey 2007). Slopes range from 5 to 12 percent. Site indices range from 70 feet at base age 50 years for shortleaf pine (*Pinus echinata*), white oak (*Quercus alba*), and northern red oak (*Q. rubra*) to 90 feet for yellow-poplar (U.S. Department of Agriculture Natural Resources Conservation Service 2012). The climate is characterized by long, moderately hot summers and short, mild winters (Thornthwaite 1948) Average annual temperature ranges from 34°F in January to 73°F in July with annual average precipitation of 55 to 60 inches that is fairly evenly distributed throughout the year (Smalley 1982).

Planting Design and Establishment

The site was planted during March 2009. The planting design (randomized complete block) was 27 units (3 x 3 x 3) consisting of three species (yellow-poplar, black cherry, and sweetgum) planted with cherrybark oak, three spacings (6 x 6 feet, 8 x 8 feet, and 10 x 10 feet), and three replicates. Each block contained only one spacing such that a block contained three replicates of each species for that spacing (9 units). The 6- x 6-feet units were 0.20 acres each, while the 8- x 8-feet and 10- x 10-feet units were 0.25 acres each. The entire research area was approximately 6.3 acres of planted trees.

Each unit had an odd number of rows. The odd-numbered rows were planted with the competitor species (either yellow-poplar, black cherry, or sweetgum) while the even-numbered rows were planted with alternating competitor species and cherrybark oak with the competitor species always being on the edge of the unit row. Thus eight competitor trees always surrounded a cherrybark oak (fig. 1). An example of the number of trees and rows planted for a 6- x 6-unit (0.20 acre) was 15 rows of 15 trees consisting of 225 total trees, 49 of those trees were cherrybark oak and 176 were the competitor species.

Before Thinning

Row 1	Row 2	Row 3	Row 4	Row 5	Row 6	Row 7
O	O	O	O	O	O	O
O	X	O	X	O	X	O
O	O	O	O	O	O	O
O	X	O	X	O	X	O
O	O	O	O	O	O	O
O	X	O	X	O	X	O
O	O	O	O	O	O	O

After Thinning

Row 1	Row 2	Row 3	Row 4	Row 5	Row 6	Row 7
O		O		O		O
	X		X		X	
O		O		O		O
	X		X		X	
O		O		O		O
	X		X		X	
O		O		O		O

O = competitor species

X = cherrybark oak

Empty cell = thinned competitor stem

Figure 1—Sample planting and thinning design for mixed species units of cherrybark oak and yellow-poplar (competitor species) before and after thinning of yellow-poplar for the mixed hardwood species stand development research at Morgan County, TN, 2014.

The study site was a former mature eastern white pine (*Pinus strobus*) plantation that was completely harvested in 2001 in anticipation of a southern pine beetle (*Dendroctonus frontalis*) epidemic. The area was maintained through the use of periodic burning and bushhogging for seven years. Little woody vegetation was present. Site preparation consisted of disking during the late fall of 2008 prior to planting. After planting, glyphosate was applied twice during each of the first two growing seasons (2009 and 2010) via a backpack sprayer between the seedling rows to control grasses and other herbaceous vegetation.

After four growing seasons in the 6- x 6-feet yellow-poplar units, the planted yellow-poplar were completely overtopping the planted cherrybark oak. The overhead canopy was closed. We observed that after leaf out during the fifth growing season that yellow-poplar needed to be thinned to allow survival and further growth of the cherrybark oak. Thus, 64 percent of the yellow-poplar trees were thinned manually with a chainsaw on June 5-6, 2013. Every yellow-poplar in the even-numbered rows and every second yellow-poplar in the odd-numbered rows were cut (fig. 1).

Measurements

Survival, total height, and diameter at 4.5 feet (dbh) were measured for every planted tree in the three cherrybark oak – yellow-poplar 6- x 6-feet units (N = 675 trees) after four growing seasons. Two years after the yellow-poplar thinning and six growing seasons after the planting, the same measurements were collected for the unthinned residual trees. Number of sprouts and total height were measured for the 2-year yellow-poplar coppice. Means and standard deviations are presented to give an indication of tree size relationships between species and the variability associated with outplanted seedling regeneration and coppice regeneration.

RESULTS

Yellow-poplar Coppice

Two growing seasons following thinning, yellow-poplar coppice averaged 4.9-feet tall with 4.2 sprouts per stump (table 1). The range of coppice heights was 2 to 7.5 feet. Seven percent of the thinned stumps did not sprout or coppice.

Cherrybark Oak

Four growing seasons after planting, cherrybark oak survival, mean total height and mean dbh were 94 percent, 7.8 feet, and 0.7 inches, respectively. Survival, mean total height, and mean dbh after 6 growing seasons including the last two years after yellow-poplar thinning were 92 percent, 11.7 feet, and 1.1 inches, respectively (table 1).

Yellow-poplar

Survival, mean total height, and mean dbh, of all planted yellow-poplar (491 stems) after four growing seasons were 93 percent, 19.5 feet, and 2.7 inches, respectively (table 1). Two years following thinning, the remaining residual yellow-poplar trees (not thinned –

Table 1—Mean total height and diameters (dbh) with standard deviations (std dev) of planted yellow-poplar and cherrybark oak in mixed species plantings at 6- x 6-feet spacing and mean heights and sprout numbers with standard deviations of coppice yellow-poplar following thinning in 2014, Morgan County, TN

	N	Mean total height (std dev)	Mean dbh[a] (std dev)	Mean sprout number (std dev)
		-------feet-------	-------inches-------	-------#-------
After 4 growing seasons (2009-2012)				
Yellow-poplar	491	19.5 (4.4)	2.7 (1.0)	—
Cherrybark oak	138	7.8 (2.2)	0.7 (0.2)	—
After 6 growing seasons (2009-2014) and 2 growing seasons following thinning				
Yellow-poplar residuals	168	26.0 (3.2)	3.9 (0.5)	—
Cherrybark oak	135	11.7 (2.6)	1.1 (0.3)	—
Yellow-poplar coppice (2013-2014)	294	4.9 (1.4)	—	4.2 (2.0)

— = not applicable

[a] dbh = diameter breast height or 4.5 feet

168 stems) after six growing seasons averaged 26-feet tall and 3.9 inches in diameter. Six residual yellow-poplar (3 percent) trees died in the two years following the thinning. These trees had stem breakage from a windstorm during the summer of 2013.

On average for one 6- x 6-feet yellow-poplar - cherrybark oak mixed planting unit (0.20 acres), the following trees and mean total heights were present after six growing seasons: 56 yellow-poplar trees that were 19.5-feet tall, 45 cherrybark oak that were 11.7-feet tall, and 98 two-year old yellow-poplar coppice stumps with heights of 4.9 feet (table 1).

DISCUSSION

Yellow-poplar with its rapid growth will outgrow most commercial species on well-drained, productive sites in the southeast United States (Beck and Della-Bianca 1981). Height growth rates of 4 feet per year and diameter growth rates of 0.5 inches per year are not uncommon (Brooks 2013, Clatterbuck 2004). Similar growth rates were observed in this study (table 1). O'Hara (1986) reported that yellow-poplar dominated oaks in the Piedmont even when oaks were older, residual trees originating before the disturbance that initiated the stand. Yellow-poplar was chosen as one of the competitor species in the larger study to assess the influence of a faster-growing species on the growth, form, and development of cherrybark oak. Yellow-poplar was assumed to have a greater growth rate and would need to be thinned at some time to allow cherrybark oak to continue to grow and develop. By planting at different spacings, the times until crown closure and overtopping of the cherrybark oak by yellow-poplar could be ascertained. At the 6- x 6-feet narrow spacing (1,210 stems per acre) on the sites of this study, canopy closure occurred after 4 growing seasons. The total average height of yellow-poplar was more than twice of that of cherrybark oak (19.5 feet vs. 7.8 feet, respectively – table 1).

Limited information is available about the capability and growth of yellow-poplar coppice. Most of the present research is for short rotation woody crops for bioenergy or biomass (Rousseau 2015). However, research information is not available on how yellow-poplar might be managed initially to encourage straight, branch-free lower boles of preferred crop trees by regulating crown expansion, then the subsequent influence of coppice after thinning for the continued development (form and pruning) of grade crop trees of cherrybark oak. Although yellow-poplar is known to be a copious seeder that regenerates well naturally (Beck and Della-Bianca 1981), the species also is a dependable, prolific sprouter that outgrows most any species originating from seed or sprouts in natural stands (Beck 1977, McGee and Clark 1975, Vickers and others 2011). In this study, coppice sprouts after two growing seasons from 4-year-old planted yellow-poplars averaged 5-feet tall with 93 percent survival. If the thinning had been conducted during the dormant season prior to leaf out rather than in June during the growing season after leaf out when trees were elongating, growth of yellow-poplar coppice may have been even greater (Smith and others 1997).

Preliminary results suggest that the vigorous coppice of yellow-poplar will influence the development of adjacent oaks. The degree is unknown since these are only two-year growth data since thinning or coppicing. At present growth rates, the yellow-poplar coppice could conceivably become taller (table 1) than the oaks even though the oaks are four years older. However, coppice growth rates usually diminish with age (Smith and others 1997) and interspecific competition with taller and older oaks and yellow-poplar is also expected to influence coppice growth rates.

Meanwhile, only four of the eight competitor yellow-poplar trees surrounding each cherrybark oak were thinned after four growing seasons. These remaining residual trees are growing rapidly and expanding their crowns in response to increased growing space. Although the cherrybark oak is continuing to increase in size, the greater growth of the yellow-poplar residuals is projected to overtop the oak after 7 to 9 growing seasons. The first thinning of the 8- by 8-feet spacing of yellow-poplar and cherrybark oak (not reported in this paper) is also expected within the same time frame. The yellow-poplar thinning of the remaining residuals in the 6- x 6-feet planting and in the 8- x 8-feet units will consist of trees that will average 6 inches dbh, allowing some potential income after 7 to 9 years of growth if small-diameter biomass markets are available. The remaining stand will consist of the cherrybark oak trees and two sets (ages) of yellow-poplar coppice following the coppice-with-standards system described by Smith and others (1997). Without the thinning, oaks would succumb to the faster-growing yellow-poplar. Without the influence of the yellow-poplar coppice, resulting oak form, pruning, and subsequent grade would be diminished.

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LITERATURE CITED

Ashton, P.M.S.; Ducey, M.J. 1996. The development of mixed species plantations as successional analogues to natural forests. In: Landis, T.D.; South, D.B., tech. coords. National proceedings, forest and conservation nursery associations. Gen. Tech. Rep. PNW-GTR-389. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station: 113-126.

Beck, D.E. 1977. Growth and development of thinned versus unthinned yellow-poplar sprout clumps. Res. Pap. SE-173. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 11 p.

Beck, D.E.; Della-Bianca, L. 1981. Yellow-poplar: characteristics and management. Agric. Handb. 583. Washington, DC: U.S. Department of Agriculture. 91 p.

Bristow, M.; Nichols, J.D.; Vanclay, J.K. 2006. Improving productivity in mixed-species plantations. Forest Ecology and Management. 223: 193-194.

Brooks, J. R. 2013. Growth and yield for a 7-year-old yellow-poplar plantation in northern West Virginia. In: Miller, G.W.; Schuler, T.M.; Gottschalk, K.W. [and others], eds. Proceedings of the 18th central hardwood forest conference. Gen. Tech. Rep. NRS-P-117. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 22-27.

Clatterbuck, W.K. 2004. Growth and development of yellow-poplar plantations on three sites ranging from 9 to 18 years. In: Connor, K.F., ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 184-186.

Clatterbuck, W.K. 2011. Dynamics of planted cherrybark oak seedlings and yellow-poplar from seed following a regeneration cut. In: Fei, S.; Lhotka, J.M.; Stringer, J.W. [and others], eds. Proceedings of the 17th central hardwood forest conference. Gen. Tech. Rep. NRS-P-78. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 287-293.

Clatterbuck. W.K.; Hodges, J.D. 1988. Development of cherrybark oak and sweetgum in mixed, even-aged bottomland stands in cental Mississippi, U.S.A. Canadian Journal of Forest Research. 18: 12-18.

Kelty, M.J. 2006. The role of species mixtures in plantation forestry. Forest Ecology and Management. 233: 195-206.

Larson, B.C. 1992. Pathways of development in mixed-species stands. In: Kelty, M.J.; Larson, B.C.; Oliver, C.D.; eds. The ecology and silviculture of mixed-species stands. Dordrecht, Netherlands: Kluwer Academic Publishers: 3-10.

Lockhart, B. R.; Ezell, A.W.; Hodges, J.D.; Clatterbuck, W.K. 2012. Twenty-nine years of development in planted cherrybark oak-sweetgum mixtures: implications for future mixed-species hardwood plantations. In: Butnor, J. R., ed. Proceedings of the 16th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 189-194.

McGee, C.E.; Clark, F.B. 1975. Harvesting and reproducing yellow-poplar. In: Impact of inflation on the management and utilization of hardwoods, proceedings of the 3rd annual hardwood symposium. Cashiers, NC: Hardwood Research Council: 26-31.

National Cooperative Soil Survey. 2007. Lonewood Series. https://soilseries.sc.egov.usda.gov/OSD_Docs/L/LONEWOOD.html [Date accessed: January 26, 2014].

O'Hara, K.L. 1986. Developmental patterns of residual oaks and oak and yellow-poplar regeneration after release in upland hardwood stands. Southern Journal of Applied Forestry. 10: 244-248.

Oliver, C.D. 1982. Stand development – its uses and methods of study. In: Means, J.E., ed. Forest development and stand development research in the Northwest. Corvallis, OR: Oregon State University, Forest Research Laboratory: 110-112.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics, update edition. New York: John Wiley. 520 p.

Oswalt, C.M.; Clatterbuck, W.K. 2011. Growing high quality hardwoods: Plantation trials of mixed hardwood species in Tennessee. In: Fei, S.; Lhotka, J.M.; Stringer, J.W. [and others], eds. Proceedings of the 17th central hardwood forest conference. Gen. Tech. Rep. NRS-P-78. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 664.

Oswalt, C.M.; Clatterbuck, W.K.; Larsen, D.R. 2011. Growing high quality hardwoods: Plantation trials of mixed hardwood species in Tennessee. In: Fei, S.; Lhotka, J.M.; Stringer, J.W. [and others], eds. Proceedings of the 17th central hardwood forest conference. Gen. Tech. Rep. NRS-P-78. Newtown Square, PA: U.S. Department of Agriculture Forest Service Northern Research Station: 33-40.

Rousseau, R. 2015. Short rotation woody crops. Publication 2611. Starkville, MS: Mississippi State University Extension Service. 2 p.

Smalley, G.W. 1982. Classification and evaluation of forest sites on the Mid-Cumberland Plateau. Gen. Tech. Rep. SO-38. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 58 p.

Smalley, G.W. 1984. Classification and evaluation of forest sites in the Cumberland Mountains. Gen. Tech. Rep. SO-50. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 84 p.

Smith, D.M.; Larson, B.C.; Kelty, M.J.; Ashton, P.M.S. 1997. The practice of silviculture: applied forest ecology. 9th ed. New York: John Wiley. 537 p.

Thornthwaite, C.W. 1948. An approach toward rational classification of climate. American Geographical Society. 38(1): 55-94.

USDA Natural Resources Conservation Service. 2012. Web Soil Survey. http://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx [Date accessed: March 21, 2012].

Vickers, L.A.; Fox, T.R.; Loftis, D.L.; Boucugnani, D.A. 2011. Predicting forest regeneration in the central Appalachians using the REGEN expert system. Journal of Sustainable Forestry. 30(8): 790-822.

EFFECTS OF LIGHT REGIME AND IBA CONCENTRATION ON ADVENTITIOUS ROOTING OF AN EASTERN COTTONWOOD *(POPULUS DELTOIDES)* CLONE

Alexander P. Hoffman, Joshua P. Adams, and Andrew Nelson[1]

Abstract—Eastern cottonwood (*Populus deltoides*) has received a substantial amount of interest from *in-vitro* studies within the past decade. The ability to efficiently multiply the stock of established clones such as clone 110412 is a valuable asset for forest endeavors. However, a common problem encountered is initiating adventitious rooting in new micropropagation protocols. Stem segments were collected from bud-broken 1 year old clone 110412 cuttings, sterilized, and stimulated to initiate shoots. Developed shoots (~2 cm in height) were excised and placed into one of three rooting media that included indole-3-butyric acid (IBA) concentrations (0.5 mg/L, 1 mg/L, or 2mg/L) in full strength DKW Medium, full strength Gamborg B5 vitamins, 2 percent sucrose, 0.6 percent agar, 10 mg/L AMP, 0.2 ml/L of Fungigone. In addition to IBA concentration, cuttings were randomly assigned to light rack positions to test the effects of wide spectrum fluorescent light (100 µmol m^{-2} s^{-1} photosynthetically active radiation (PAR), 16/8 hour photoperiod) and light emitting diode light (LED; 4:1 red-to-blue diodes, 250 µmol m^{-2} s^{-1} PAR, 16/8 hour photoperiod). After a month of exposure, there was limited rooting exhibited across treatments. However, fluorescents (3.58 ± 1.02) produced significantly better performing microcuttings (judged on morphology, visual vigor, and survival) than LEDs (2.7±0.86) ($p<0.005$). The high light intensity of the LEDs may be prompting weaker performance through unfavorably high transpiration-induced auxin uptake. While LEDs may play a role in future micropropagation protocols, results suggest that wide spectrum florescent lights produce better performing eastern cottonwood 110412 microcuttings.

INTRODUCTION

Eastern cottonwood (*Populus deltoides*), in addition to other *Populus* species and their hybrids, has become an important constituent in woody species biotechnology, *in-vitro* culture, and genetic engineering (Confalonieri and others 2003). The species has been widely cultivated due to its adaptability, growth rate, woody biomass production, wood industry uses (e.g., paper and pulpwood), and has become established as a model system because of its small genome size, short rotation cycle, rapid growth rate, and ease of vegetative propagation (Confalonieri and other 2003, Jansson and Douglas 2007). Over the past 50 years or more, the identification and screening efforts of tree improvement programs have produced a collection of superior clones which could benefit from the application of vegetative multiplication techniques such as micropropagation (Thorpe and others 1991). While each micropropagation process encompasses complex genetic, molecular, and physiological relationships, one step of the process that has received a substantial amount of interest in recent years is the induction and formation of adventitious roots (Costa and others 2013, De Klerk 2002). Two *in-vitro* microenvironmental factors that have been investigated for stimulating favorable adventitious rooting characteristics are auxin concentration and light regime (i.e., quality, intensity, and photoperiod).

While auxin has been well documented as one of the main phytohormones in a number of physiological processes such as apical dominance it has also been documented in a number of adventitious rooting processes such as regulating cell division, cell elongation, and the initiation of root apical meristems (Mironova and others 2010). Recent success of eastern cottonwood micropropagation and *Agrobacterium* transformation studies has been documented with two commonly used auxins, indole-3-acetic acid (IAA) or Indole-3-butyric acid (IBA) (Cavusoglu and others 2011, Chaturvedi and others 2004, Yadav and others 2009). However, a recurrent trend in these experiments is clone-specific responses to identical auxin concentrations, which has prolonged the process of developing new protocols (John and others 2014, Yadav

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and others 2009). In addition, there are still a number of established clones for which a micropropagation, and therefore adventitious rooting, protocol has not been derived. Further research with these clones, such as the 110412, can help determine what auxin concentrations illicit the most favorable adventitious rooting characteristics and how clone-specific responses can be implemented in future protocols. Another benefit to modern micropropagation research is the investigation of new technological advancements such as light-emitting diodes (LEDs) and how they may affect adventitious root formation.

Light quality (i.e., color), intensity, and photoperiod can each be manipulated within an *in-vitro* system to produce a unique light regime with the intent of stimulating specific physiological and morphological changes, such as adventitious rooting, in microcuttings. Studies performed within the past 20 years have documented adequate *in-vitro Populus* adventitious rooting using low intensity (~50-100 µmol $m^{-2}s^{-1}$ PAR) white fluorescents on a 16 hour light / 8 hour dark (i.e., long-day) photoperiod (Cavusoglu and others 2011, Gozukirmizi and others 1998, Kang and others 2009, Noël and others 2002, Phan and others 2004, Thakur and Srivastava 2006, Yadav and others 2009). However, there have been a number of studies, conducted on both woody species and other crop species, that have documented the potential role of far-red light (Kraepiel and others 2001), red light (Li and others 2010), blue light (Pinker and others 1989), and the implications of potential auxin-light quality interactions (Britz and Sager 1990, Chée 1986). In addition, recent technological advancements such as LEDs have provided researchers new resources to implement within *in-vitro* light quality research.

LED's compact engineering, solid-state construction, and especially their specific light spectrum capabilities has caught the attention of a number of plant tissue culture and horticultural researchers (Bula and others 1991, Morrow 2008). While there has been documented success, in terms of favorable adventitious rooting characteristics, in a number of annual crops such as Akihime strawberry (*Fragria* x *ananassa* Akihime) (Nhut and others 2003) and upland cotton (*Gossypium hirsutum* L.) (Li and others 2010) there is currently a lack of woody species, and specifically eastern cottonwood, *in-vitro* research utilizing LEDs. Given the lack of information on micropropagation adventitious rooting protocols for eastern cottonwood clone 110412, especially in response to auxin concentration and light regime, the objective of this study was to investigate the effects of traditional fluorescent and LED light regimes and three IBA concentrations (0.5 mg/L, 1 mg/L, 2mg/L) on adventitious rooting.

METHODS

Plant Material

The eastern cottonwood 110412 clone was originally selected for propagation by the USDA Forest Service Center for Bottomland Hardwood Research due to its excellent rooting, good form, and above average growth on various sites during initial screening trials (Personal communication, Dr. Randy Rousseau). The 110412 clone has also been cited for its phytoremediation abilities (Cardellino 2001, Minogue and others 2012, Rockwood and others 2006).Approximately 61 cm long hardwood cuttings of clone 110412 were collected from 1-year old, coppiced trees located on the campus of Mississippi State University (Starkville, MS) during late winter 2013. Cuttings were taken out of cold storage (4°C) during late fall 2014, cut to 20 cm in length, and soaked in room-temperature water for one week. After soaking, the base of each cutting was dipped in GreenLight© Rooting Hormone (San Antonio, TX) and placed (dipped end down) into 950 ml Mini-Treepots (Stuewe & Sons, Inc., Tangent, OR) containing EarthGro topsoil until only ~25 percent of the cutting remained aboveground. The Mini-Treepots were then placed into 3.28 L growing trays, six pots per tray. The growing trays were placed underneath four Sylvania T12 40W wide spectrum fluorescent tube light bulbs emitting 102.3 ± 6.8 µmol $m^{-2}s^{-1}$ photosynthetically active radiation (PAR) on a 16/8 (i.e., long day) photoperiod until initial bud break. All hardwood cutting establishment took place in a lab at the School of Forest Resources, University of Arkansas at Monticello campus.

Stem Collection and Shoot Initiation

Internode and node segments were collected from shoots developed from hardwood cuttings in early January, 2015. These were cut into 3 cm segments and then surface sterilized using a 25 minute agitated wash in 15mL of 30 percent bleach-ddH_20 solution supplemented with ~1 mL Tween® 20 and overhead ultraviolent light exposure (fig. 1). After the initial wash, stem segments were gently rocked ten times with sterilized ddH_20, drained, and gently rocked an additional ten times with fresh sterilized ddH_20. Each stem segment was then pressed into a BioWorld™ Extra Deep Mono Petri Dishes containing ~36 mL of shoot initiation media (SIM) to stimulate shoot development. The SIM consisted of full strength McCown Woody Plant Medium (McCown and Lloyd 1981), 3 percent sucrose, full strength Gamborg B5 vitamins, 1 percent agar, 0.7 mg/L 6-Benzylaminopurine (BAP), 0.01 mg/L Indole-3-butyric acid (IBA), 0.3 mg/mL Ampicillin Trihydrate (AMP), and 0.5 ml/L of 100x stock solution Fungigone. SIM pH was adjusted to 5.8 using 1 M NaOH. All tissue transfers took place under a sterile hood and all plates were wrapped with Parafilm before removal from the hood. SIM plates were then randomly

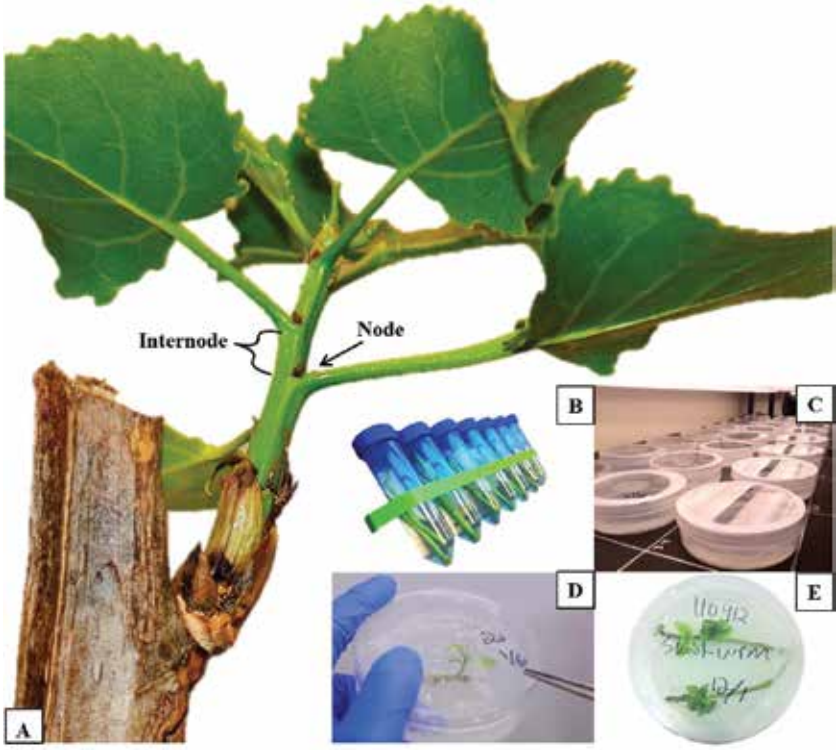


Figure 1—Stem Collection and Sterilization Process. The aboveground biomass, including node and internode segments, of eastern cottonwood (Populus deltoides) clone 110412 cuttings was excised in early January, 2015 (A). All non-stem material (i.e., leaf material) was excised from the stem segments before they were washed in 15 mL of a 30% bleach, ddH20 solution supplemented with approximately 1mL of Tween 20 for 25 minutes (B). The sterilized stem segments were cut into 3 cm segments, placed in SIM plates, and randomly distributed within the SIM light rack (C). After 2-3 weeks shoots were adequately initiated (D, E).

distributed across a SIM light rack (table 1). Stem segments were immediately transferred to a fresh SIM plate upon any sign of contamination.

Shoot Excision, Microcutting Collection, and Experimental Rooting Conditions

After a shoot reached approximately 2 cm in height in the SIM plates, the shoot was excised from its respective stem segment at the lowest feasible point of connection (e.g., microcutting). Each microcutting was vertically positioned in a PhytoTechnology Laboratories® 177 mL culture vessel enclosed by vented PhytoCaps™ containing ~42 mL of rooting media (RM). Three RM were tested, all identical except for different concentrations of IBA. Each RM consisted of full strength DKW medium with vitamins (Driver and Kuniyuki 1984, McGranahan and others 1987), full strength Gamborg B5 vitamins, 2 percent sucrose, 0.6 percent agar, 10 mg/L AMP, 0.2 ml/L of 100x stock solution Fungigone, and either 0.5 mg/L, 1 mg/L , or 2mg/L of IBA. Each RMs' pH was adjusted to 5.6 using 1 M NaOH.

Once microcuttings were randomly assigned to one of the three IBA concentrations, they were randomly placed on light racks where they were exposed to either fluorescent or LED light: 4 culture vessels for each RM per light rack. The fluorescent light regime consisted of four wide spectrum white fluorescent lights producing ~100 µmol m^{-2} s^{-1} PAR on a 16/8 light/dark hours (i.e., long day) photoperiod. The LED light regime consisted of two Tesler 120 watt rectangular indoor grow lights (44 three watt red (630nm) diodes and 11 three watt blue (460nm) diodes) producing ~250 µmol m^{-2} s^{-1} PAR on a long day photoperiod. Light racks were designed so that two of each light regime was tested at the same time. Light conditions varied between light racks, but light conditions were homogenous within racks (table 1). Light rack intensities measurements were taken at the top of each microcutting vessel with a quantum flux meter (Apogee Instruments, Inc.) and the long day photoperiod was achieved by producing 16 hours of continuous light (12:30am-4:30pm) followed by 8 hours of continuous dark (4:20pm-12:30am), as regulated by 24 hour timers. In order to ensure uninterrupted light exposure the RM vessels were placed upon the light

Table 1—Light Rack Specifications. Three light racks were utilized within the experiment 1) Shoot Initiation (SIM) 2) Light Rack 1 (LR1), and 3) Light Rack 2 (LR2). Each SIM contained 24 randomly distributed SIM plates and each LR1 and LR2 contained 12 randomly distributed culture vessels. P-values of a Welch's Two Sample t-test (n=48) and *Welch's Two Sample t-test (n=48) with unequal variances are presented

Light Rack	Light Source	Light Photoperiod (light hours / dark hours)	Trial	Light Intensity ($\mu mol\ m^{-2}\ s^{-1}$ PAR ± SD)	p-value
SIM	Four Sylvania T12 40 W wide spectrum fluorescent tubes	16/8	1	105.88 ± 5.02	0.26*
			2	108.08 ± 8.04	
1	Four Sylvania T12 40 W wide spectrum fluorescent tubes	16/8	1	93.25 ± 6.62	0.35
			2	95.75 ± 6.09	
2	Two Tesler 120 W 4:1 red (630nm) to blue (430nm) diode ratio rectangular indoor LED grow lights	16/8	1	256.75 ± 12.14	0.99
			2	256.75 ± 12.75	

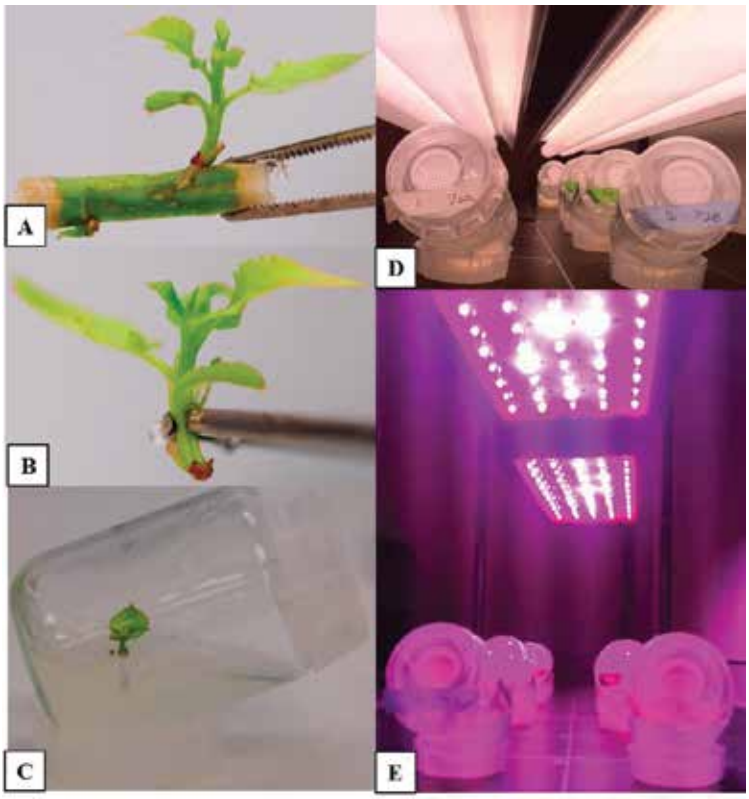


Figure 2—Shoot Excision and Light Regime Assignment. After a shoot reach approximately 2 cm in height it was removed from its SIM plate (A). The shoot was then excised at its base from the stem segment and was thereon termed a microcutting (B). Each microcutting was then randomly assigned a rooting media (RM) and placed into a RM vessel containing that RM (C). The RM vessel was then randomly assigned to, and within, either a wide spectrum fluorescent light rack (D) or a LED light rack (E) where they were exposed to the light regimes for a month.

racks at a 45° angle. Upon any sign of contamination, microcuttings were immediately transferred to a fresh RM vessel containing the same RM, and placed back into its respective position (fig. 2).

Measurements

Microcuttings were exposed to the light regimes for one month. During the trial, weekly performance ratings were conducted using a modified Kang and others (2009) scale that provided a gradient of poor to excellent performance (table 2). Microcuttings were examined daily for any signs of adventitious rooting. If rooting was observed it was documented so that the number of days to visible adventitious root for each microcutting was known. At the conclusion of the month-long exposure period, each microcutting was checked for any visible signs of rooting to determine rooting success, the number of adventitious roots per microcutting (rooting density), and the average length of each adventitious root. A Nikon D7100 with an attached 105 mm Nikon macro lens was used to document any noteworthy morphological changes such as leaf color variation or leaf necrosis that may have affected the rooting attributes measured.

Statistical Analysis

The study used a completely randomized split-plot design with light regime as the main factor and RM as the sub factor within each light regime. Only one shoot produced any signs of rooting which limited the ability to analyze any rooting attribute. Therefore, the only variable analyzed was the weekly performance ranking. Due to the ordinal scale of the rankings and preliminary analysis that found the data were non-normally distributed with heteroscedastic residuals, the nonparametric aligned rank transformation test was performed using the "art" function within the ARTOOL library (Wobbrock and others 2011) in R (version 3.1.2.) to examine the effects of light regime, RM (i.e., IBA concentration), and their interaction on microcutting performance rating (n=48).

RESULTS AND DISCUSSION

Although there have been several significant *in-vitro Populus* breakthroughs documented within the past 20 years, developing new micropropagation protocols are still susceptible to an array of difficulties due to the numerous microenvironmental factors that can be manipulated and their interactions. Rooting was observed in a sole microcutting that was subjected to the LED light regime and 2mg/L IBA. While the microcutting seemed to be consistent in terms of visual vigor (average performance rating of 4 throughout entire exposure period) the microcutting was produced from a stem segment after only 15 days. In comparison, the mean SIM phase duration across all microcuttings was 21.7 ± 5.45 days. The accelerated SIM phase could represent a superior (e.g., physiologically and genetically) stem segment whose superiority continued throughout subsequent processes such as adventitious root induction and formation.

While confident conclusions cannot be drawn from the minimal rooting, the weekly performance ratings revealed several interesting results. The aligned rank test showed that the average weekly performance rating through two weeks of exposure did not differ by light regime, RM, or their interaction ($p \geq 0.13$). However, after three weeks a significant difference in light regime was found ($p < 0.05$), which continued through the fourth week ($p < 0.01$) (table 3). The statistically significant results suggests that after three weeks of exposure, microcuttings in a wide spectrum fluorescent light regime had a higher ranking (3.96 ± 0.91) than microcuttings in a LED regime (3.26 ± 0.81) by an average of 0.70 (table 4). After an additional week, the average difference between light regimes grew to 0.88. This study's result of a wide spectrum white fluorescent

Table 2—Microcutting Performance Rating Scale. A modified Kang and others (2009) performance scale constructed to visually quantify microcutting performance on a weekly basis through the month long experiment. The scale represents a gradient from poor performance (1) to excellent performance (5)

Performance Rating	Description
1	Microcutting completely died
2	Microcutting nearly dead, leaves exhibiting severe necrosis and/or abnormal morphological changes
3	Microcutting has exhibited no growth since last rating and/or exhibiting moderate sign of necrosis and abnormal morphological changes
4	Microcutting has exhibited limited growth, only lower leaves (i.e., leaves touching media) are exhibiting necrosis and/or abnormal morphological changes
5	Microcutting actively growing, showing limited signs of leaf necrosis and/or abnormal morphological changes

light regime producing higher performing microcuttings supports both Cavusoglu and others (2011) who used cool white fluorescent light (50 µmol m^{-2} s^{-1}) to achieve 80 percent-100 percent rooting of eastern cottonwood clones 89 M 011, 89 M 044, 89 M 048, and 89 M 066 in two weeks and Yadav and others (2009) who used florescent tubes (45 µmol m^{-2} s^{-1}) to induce rooting of clones WIMCO199 and L34 after 4-5 weeks. While these studies did document adequate rooting, our lack of rooting may be due to the higher light intensity (100 µmol m^{-2} s^{-1}) produced by the fluorescent lighting that could have increased the microcutting transpiration rates (Jarvis and Ali 1984, Jarvis and Saheed 1987), resulting in greater exogenous auxin uptake—a factor that can be inhibitory to rhizogenic activity (Brock and Kaufman 1991, Garrido and others 2002). The same reasoning may also explain the lack of rooting and weaker performance of cuttings exposed to the LED light regime.

While there was not a statistical significant effect of IBA concentration over the exposure period, the lowest performance ratings resulted from microcuttings exposed to either 1 mg/L IBA within the LED light regime or the 2 mg/L IBA within the LED light regime each week (table 5). The LED's high light intensity (250 µmol m^{-2} s^{-1}) could have exacerbated the increased transpiration-induced exogenous auxin uptake over the exposure period. Another interesting result was observed of a continual (~0.6 rating points per week) decline in microcutting performance rating exposed to the LED light regime across the IBA concentrations. The continual decline in microcutting performance supports the documentation of high auxin concentrations being stimulatory within the induction phase (~ first 96 hours) but becoming progressively inhibitory after the initial induction phase (De Klerk 2002). However, there is also another light regime factor that was manipulated within the LED light regime, light quality. While the role of red and blue light qualities has been well documented in their regulation of phytochromes and cryptochromes, their role (and potential physiological associations) within *in-vitro* adventitous rooting is still unclear. Li and others (2010) documented a significant decrease

Table 3—Aligned Rank ANOVA. Results from the nonparametric aligned rank ANOVA test used to examine the effects of light regime, rooting media (RM) and their interaction over a four week period. Significance was assessed at the α = 0.05 level

Week	Factor	Sum of Squares	F Value	P Value
1	Light Regime	470.14	2.45	0.13
	RM	184.41	0.46	0.64
	Light Regime x RM	494.50	1.27	0.29
2	Light Regime	345.41	1.73	0.20
	RM	188.36	0.46	0.63
	Light Regime x RM	519.85	1.33	0.28
3	Light Regime	1441.35	8.24	<0.05
	RM	533.50	1.36	0.27
	Light Regime x RM	384.93	0.96	0.39
4	Light Regime	1735.02	10.66	<0.005
	RM	70.09	0.18	0.84
	Light Regime x RM	96.14	0.24	0.79

Table 4—Mean Performance Ratings by Light Regime. Mean (± 1 standard deviation) weekly performance ratings by light regime using a modified Kang and others (2009) scale

Light Regime	Week 1 Rating (±SD)	Week 2 Rating (±SD)	Week 3 Rating (±SD)	Week 4 Rating (±SD)
Wide Spectrum Fluorescent	4 ± 0.98	4.17 ± 0.87	3.96 ± 0.91*	3.58 ± 1.02**
Tesler 4:1 LEDs	4.43 ± 0.59	3.91 ± 0.79	3.26 ± 0.81*	2.7 ± 0.86**

* statistical significance at the 0.05 level
** statistical significance at the 0.005 level

Table 5—Average Weekly Performance Rating by Light Regime and IBA Concentration. Microcuttings were randomly assigned one of three IBA concentrations, light regime, and position within light regime to investigate their effects on adventitious rooting stimulation over a month long exposure period. Weekly performance ratings were taken using a modified Kang and other (2009) scale

Light	IBA (mg/L)	Week 1 Rating (±SD)	Week 2 Rating (±SD)	Week 3 Rating (±SD)	Week 4 Rating (±SD)
Wide Spectrum Fluorescent	0.5	3.88 ± 0.83	4.25 ± 0.71	4.00 ± 0.76	3.75 ± 0.89
	1	3.88 ± 0.99	3.75 ± 1.04	3.63 ± 1.10	3.38 ± 1.06
	2	4.31 ± 1.16	4.50 ± 0.76	4.25 ± 0.89	3.63 ± 1.19
Tesler 4:1 LEDs	0.5	4.63 ± 0.52	4.13 ± 0.83	3.63 ± 0.92	2.63 ± 0.92
	1	4.29 ± 0.76	4.00 ± 0.82	3.00 ± 0.58	2.57 ± 0.53
	2	4.38 ± 0.52	3.63 ± 0.74	3.13 ± 0.83	2.88 ± 1.13

in root activity of upland cotton (*Gossypium hirsutum* L.) microcuttings cultured under a 3:1 red (660 nm) to blue (460 nm) LED, standard white flourescent, and blue LED light regimes in comparison to microcuttings cultivated under a red light regime. Our study's 4:1 red (630 nm) to blue (430 nm) LED seems to be an unfavorable light quality and may be made more effective by decreasing the blue diode component, a light quality that has also been documented to be inhibitory to birch (*Betula pendula*) microcutting rooting (Pinker and others 1989) and loblolly pine (*Pinus taeda*) hypocotyl explant adventitous root elongation (Amerson and others 1988). Merkle and others (2005) also discussed the implications of red and blue LEDs when they documented a positive effect on germination, conversion, and early growth of *Pinus taeda, Pinus elliottii* Engelm.,and *Pinus palustris* Mill. somatic embryos with red wavelengths. However, the researchers were hesitate to designate the cause of the documented effects to a stimulatory effect of red wavelengths because they also documented a consistent poorer result from blue wavelengths in comparison to both red wavelengths and standard white flourescents. Those results could imply an inhibtory effect of blue wavelengths instead of a stimulatory effect of red wavelengths. In consideration of potential light quality and auxin interaction during *in-vitro* adventitious rooting, Rossi and others (1993) documented no significant differences in root induction or formation of damson plum (*Prunus insititia*) clone GF655-2 microcuttings exposed to blue, red, and white light qualties supplemented with either an absence or application of exogenus auxin.

Our study's results suggest that a wide spectrum flourescent light regime will produce better performing eastern cottonwood 110412 microcuttings regardless of IBA concentration. However, in developing future micropropagation protocols for the 110412 clone, other potential microenvironmental manipulations may include a lower light intensity (e.g., 50 µmol m^{-2} s^{-1}), relatively unstable auxin (e.g., indole-3-acetic acid) that can be relatively quickly photo-oxidized (De Klerk 2002, Fett-Neto and others 2001), and use of either white or red light qualities.

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LITERATURE CITED

Amerson, H.V.; Frampton, L.J.; Mott, R.L., Spaine, P.C. 1988. Tissue culture of conifers using loblolly pine as a model. In: Hanover, J.W. and Keathley, D.E., eds. Genetic manipulation of woody plants. Basic life sciences. New York: Plenum Press: 117-137.

Brock, T.G.; Kaufman, P.B. 1991. Growth regulators: an account of hormones and growth regulation. In: Steward, F.C.; Bidwell, R.G.S., eds. Plant physiology a treatise, Vol. 10: growth and development. San Diego: Academic Press: 277-340.

Britz, S.J., Sager, J.C. 1990. Photomorphogenesis and photoassimilation in soybean and sorghum grown under broad spectrum or blue-deficient light sources. Plant Physiology. 94(2): 448-454.

Bula, R.J.; Morrow, R.C.; Tibbitts, T.W.; Barta, D.J. 1991. Light-emitting diodes as a radiation source for plants. HortScience 26(2): 203-205

Cardellino, R.W. 2001. Phytoremediation of arsenic contaminated soils by fast growing eastern cottonwood *Populus deltoides* (Batr.) clones. Gainesville, FL: University of Florida. 92 p. M.S. thesis.

Cavusoglu, A.; Ipekci-altas, Z.; Bajrovic, K. [and others]. 2011. Direct and indirect plant regeneration from various explants of eastern cottonwood clones (*Populus deltoides* Bartram ex Marsh.) with tissue culture. African Journal of Biotechnology. 10(16): 3216-3221.

Chaturvedi, H.C.; Sharma, A.K..; Agha, B.Q. [and others]. 2004. Production of cloned trees of *Populus deltoides* through in vitro regeneration of shoots from leaf, stem and root explants and their field cultivation. Indian Journal of Biotechnology. 3(2): 203-208.

Chée, R. 1986. In vitro culture of *Vitis*: the effect of light spectrum, manganese sulfate and potassium iodide on morphogenesis. Plant Cell, Tissue and Organ Culture. 7(2): 121-134.

Confalonieri, M.; Balestrazzi, A.; Bisoffi, S.; Carbonera, D. 2003. In vitro culture and genetic engineering of *Populus* spp.: Synergy for forest tree improvement. Plant Cell, Tissue and Organ Culture. 72(2): 109-138.

Costa, C.T.; Almeida, M.R.; Ruedell, C.M. [and others]. 2013. When stress and development go hand in hand: main hormonal controls of adventitious rooting in cuttings. Frontiers of Plant Science. 4: 1-19.

De Klerk, G.-J. 2002. Rooting of microcuttings: Theory and practice. In Vitro Cellular & Developmental Biology-Plant. 38(5): 415-422.

Driver, J.A.; Kuniyuki, A.H. 1984. In vitro propagation of paradox walnut rootstocks. HortScience. 19(4): 507-509.

Fett-Neto, A.G.; Fett, J.P.; Veira Goulart, L.W. [and others]. 2001. Distinct effects of auxin and light on adventitious root development in *Eucalyptus saligna* and *Eucalyptus globulus*. Tree Physiology. 21(7): 457-464.

Garrido, G.; Guerrero, J.R.; Cano, E.A. [and others]. 2002. Origin and basipetal transport of the IAA responsible for rooting of carnation cuttings. Physiologia Plantarum. 114(2): 303-312.

Gozukirmizi, N.; Bajrovic, K.; Ipekci, Z. [and others]. 1998. Genotype differencies in direct plant regeneration from stem explants of *Populus tremula* in Turkey. Journal of Forest Research. 3(2): 123-126.

Jansson, S.; Douglas, C.J. 2007. *Populus*: a model system for plant biology. Annual Review of Plant Biology. 58: 435-458.

Jarvis, B.C.; Ali, A.N. 1984. Irradiance and adventitious root formation in stem cuttings of *Phaseolus aureus* Roxb. New Physiologist. 97(1): 31-36.

Jarvis, B.C.; Saheed, A.E. 1987. Adventitious root formation in relation to irradiance and auxin supply. Plant Biology. 29(5): 321-333.

John, E.; Maqbool, A.; Malik, K.A. 2014. Optimization of Agrobackterium Tumefaciens mediated transformation in *Populus Deltoides*. Pakistan Journal of Botany. 46(3): 1079-1086.

Kang, B.; Osburn, L.; Kopsell, D. [and others]. 2009. Micropropagation of *Populus trichocarpa* "Nisqually-1": the genotype deriving the *Populus* regerence genome. Plant Cell, Tissue and Organ Culture. 9(3): 251-257.

Kraepiel, Y.; Agnes, C.; Thiery, L. [and others]. 2001. The growth of tomato (*Lycopersicon esculentum* Mill.) hypocotyls in the light an in darkness differentially involves auxin. Plant Science. 161(6): 1067-1074.

Li, H.; Xu, Z.; Tang, C. 2010. Effect of light-emitting diodes on growth and morphogenesis of upland cotton (*Gossypium hirsutum* L.) plantlets *in vitro*. Plant Cell, Tissue and Organ Culture. 103(2): 155-163.

McCown, B.H.; Lloyd, G. 1981. Woody plant medium (WPM) a mineral nutrient formulation for microculture of woody plant species. HortScience. 16(3): 453.

McGranahan, G.H.; Driver, J.A.; Tulecke, W. 1987. Tissue culture of Juglans. In: Cell and tissue culture in forestry. Netherlands: Springer: 261-271.

Merkle, S.A.; Montello, P.A.; Xia, X. [and others]. 2005. Light quality treatments enhance somatic seedling production in three southern pine species. Tree Physiology. 26(2): 187-194.

Minogue, P.J.; Miwa, M.; Rockwood, D.L.; Mackowiak, C. L. 2012. Removal of nitrogen and phosphorus by Eucalyptus and Populus at a tertiary treated municipal wastewater sprayfield. International Journal of Phytoremediation. 14(10): 1010-1023.

Mironova, V. V.; Omelyanchuk, N. A.; Yosiphon, G. [and others]. 2010. A plausible mechanism for auxin patterning along the developing root. BMC Systems Biololgy. 4(1): 98.

Morrow, R.C. 2008. LED Lighting in Horticulture. HortScience. 43(7): 1947-1950.

Nhut, D.T.; Takamura, T.; Watanabe, H. [and others]. 2003. Responses of strawberry plantlets cultured in vitro under superbright red and blue light-emitting diodes (LEDs). Plant Cell, Tissue and Organ Culture. 73(1): 43-52.

Noël, N.; Lepé, J.C.; Pilate, G. 2002. Optimization of in vitro micropropagation and regeneration for *Populus* x *interamericana* and *Populus* x *euramericana* hybrids (P. *deltoides*, P. *trichocarpa*, and P. *nigra*). Plant Cell Reports. 20(12): 1150-1155.

Phan, C.T.; Jorgensen, J.; Jouve, L. [and others]. 2004. Micropropagation of *Populus euphratica* Oliver. Belgian Journal of Botany. 137(2): 175-180.

Pinker, I.; Zoglauer, K.; Göring, H. 1989. Influence of light on adventitious root formation in birch shoot cultures *in vitro*. Journal of Plant Biology. 31(4): 254-260.

Rockwood, D.L.; Carter, D.; Stricker, J. 2006. Commercial tree crops for phosphate mined lands. Final Report. Publication No. 03-141-225. Bartow, FL: Florida Institute of Phosphate Research. 5 p.

Rossi, F.; Baraldi, R.; Facini, O.; Lereari, B. 1993. Photomorphogenic effects on in vitro rooting of *Prunus* rootstock GF 655-2. Plant Cell, Tissue and Organ Culture. 32(2): 145-151.

Thakur, A. K.; Srivastava, D.K. 2006. High-efficiency plant regeneration from leaf explants of male himalayan poplar (*Populus ciliata* wall.). In Vitro Cellular & Developmental Biology - Plant. 42(2): 144-147.

Thorpe, T.A.; Harvy, I.S.; Kumar, P.P. 1991. Application of micropropagation in forestry. In: Debergh, P.; Zimmerman, R.H., eds. Micropropagation, technology and application. Dordrecht, Netherlands: Kluwer Academic Publishers: 311-336

Wobbrock, J.O.; Findlater, L.; Gergle, D.; Higgins, J.J. 2011. The aligned rank transform for nonparametric factorial analyses using only ANOVA procedures. In: Proceedings of the ACM conference on human factors in computing systems. Vancouver, British Columbia: New York: ACM Press: 143-146.

Yadav, R.; Arora, P.; Kumar, D. [and others]. 2009. High frequency direct plant regeneration from leaf, internode, and root segments of Eastern Cottonwood (*Populus deltoides*). Plant Biotechnology Report. 3(3): 175-182.

SURVIVAL AND GROWTH PERFORMANCE OF TWO OAK SPECIES AND THREE PLANTING STOCKS ON LANDS DISTURBED BY HURRICANE KATRINA

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Abstract—Regeneration of oaks is a priority for most landowners in the south given their inherent wildlife benefits, economic return, ascetics, and providing habitat for endangered species. In the case of natural disasters such as Hurricane Katrina artificial regeneration of these stands may be the only viable option to reestablish an overall oak component in a future stands overstory. This study evaluated growth of two oak species, water oak *(Quercus nigra*) and swamp chestnut oak (*Quercus michauxii*), and three planting stocks: 1-0 bareroot, conventional containerized, and EKOgrown™ seedlings were compared for two growing seasons. Conventional containerized planting stock exhibited greater groundline diameter (GLD) growth for both species at the end of the first growing season compared to bareroot and EKO™ planting stock. Bareroot seedlings had similar GLD growth to EKO™ seedlings for both years. Conventional containerized seedlings height differed in water oak but did not differ in swamp chestnut oak at the end of year two compared to bareroot seedlings. EKO™ seedlings exhibited severe dieback at the end of both growing seasons and the least amount of height growth.

INTRODUCTION

The southern states of North America have some of the most productive bottomland soil in the United States. These fertile deposits from rivers coupled with a warmer climate and longer growing season make these sites the primary area for bottomland hardwoods in the region. Hardwoods such as oaks (*Quercus* spp.) are valuable resources to the southern forest for timber production, flood storage, and nutrient charge (Hall and Lambou 1989, Ezell and others 2007, Moree and others 2010). Bottomland hardwoods provide many benefits to both non-industrial private landowners (NIPL) and industry landowners including aesthetics, excellent wildlife habitat, recreational activities, endangered species refuges, and opportunities to generate money from harvesting. Bottomland hardwood forests in Mississippi were damaged by Hurricane Katrina in 2005 when it made landfall along the Gulf Coast. A major problem resulting from the damage is lack of oak regeneration, which can be problematic without advance planning. Thus, the preferred practice of natural regeneration, which requires a well-structured plan and timing to be successful, may not be an option (Coder 1994, Belli 1999, Dey and others 2008). Typically, oaks will be a much smaller percentage of the new overall stand component when compared to the parent stand (Beck and Hooper 1986, Loftis 1988). Some studies suggest that this can be a result of oaks being much slower to grow when compared to light-seeded competitors (Smith 1993, Thompson and Nix 1995). In the South, oaks are not strong competitors with rapid and aggressively growing herbaceous vegetation. Herbaceous competition is the main cause for seedling mortality during the establishment period of 1-2 years (Smith and others 1997). With these circumstances, artificial regeneration of desirable hardwoods is a viable option and may be the only one available in areas disturbed by a major storm.

Regeneration efforts on bottomland sites are mainly focused to enhance wildlife habitat, produce timber, and increase/protect water quality which is a main concern with managers in the South (Witter 1991). Artificial regeneration has become an important forest management option when a stand lacks natural regeneration to restore the oak component of a stand. However, this practice has potential to be problematic on mesic sites (Lorimer 1993, Johnson and others 2002). Seasonal flooding on poorly drained sites causes more problems for oak re-establishment on a floodplain site and favors undesirable species that are more tolerant of wet conditions.

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Planting a high quality and vigorous seedling is an essential element of any artificial regeneration prescription (Dey and Parker 1997). These seedlings will have taller stems, more fibrous roots, ideal shoot to root ratio, and larger diameters. These characteristics lead to better survival and growth rates. According to Dey and Parker (1997), larger seedlings perform better against competing vegetation. Couple a high quality seedling with proper chemical and mechanical methods and survival and growth has shown to increase (Ezell and others 2007).

Proper planting is also an essential component. Improper planting such as J-rooting, excessive root pruning or shallow planting can increase the chances of mortality and seedling stress. Using a high quality seedling is negated if improper planting and handling techniques are used. Many research studies have correlated mortality of seedlings with improper planting. These research studies mainly focused on one planting stock, but little has been done to compare survival and growth of various oak planting stocks. This study will help fill the void of information to help landowners make a more well-informed and more cost-effective decision.

OBJECTIVES

The objectives were (1) compare the two year height growth and groundline diameter growth of water oak (*Q. nigra*) and swamp chestnut oak (*Q. michauxii*); and (2) compare the two year height growth and groundline diameter growth of three planting stocks (high-quality 1-0, bareroot, conventional containerized, and large containerized seedlings).

MATERIALS AND METHODS

Study Sites

Research was conducted on two privately owned sites located in southeast Mississippi. One site is located 16 km northwest of Hattiesburg and the other is 8 km southeast of Lucedale. The soil series represented in this study were Freest-Susquehanna-Prentiss and Lenoir silt loam, respectively. According to the Natural Sources Conservation Service (2012), the Hattiesburg area receives an average of 57 inches of annual rainfall. The Lucedale area receives an average of 61inches of annual rainfall.

The Malone site (3123’47.93N”, -8928’33.24”W) in Lamar County, has a Freest-Suspuehanna-Prentis soil series. Prior to Katrina, timber on this site was a mixed stand of loblolly pine (*Pinus taeda*), sweetgum (*Liquidambar styraciflua*) and water oak. After Katrina, a salvage operation was performed. The site has been kept open by periodic mowing. The few remaining woody stems were injected with a 20 percent aqueous solution of Arsenal® AC (*Imazapyr*) to prepare for planting. A restrictive layer approximately 10 in. below ground was reported by the planting crews that may be a restrictive layer for the first years of growth.

The second site, the Welford site (3049’27.27”N, -8827’13.86”W) in George County, has a Lenoir silt loam soil series. Prior to Katrina the site also had a mixed stand of loblolly pine and hardwoods. A small drain in the center of the site contained several stems of pond cypress (*Taxodium ascendens*). According to the landowner, this site does flood during, wet winters and springs due to its close proximity to the Escatawapa River. After Katrina, a salvage operation was performed and remaining debris was piled. The site was then root-raked and has been mowed and cultivated every year for a wildlife food plot. Remaining stems were injected with a 20 percent aqueous solution of Aresenal® AC to prepare for planting.

Seedlings

Two oak species, water oak and swamp chestnut oak, and three planting stocks: high quality 1-0 bareroot, 240cm^3 conventional containerized, and EKOgrown™ seedlings grown in a Rootmaker® container were used for evaluation. Bareroot seedlings were purchased from the Rayonier nursery in Elberta, Alabama. Conventional containerized seedlings were purchased from Mossy Oak Native Nurseries™ in West Point, Mississippi. EKO™ seedlings were produced and purchased from RES Native Tree Nursery in Montegut, Louisiana.

Planting

A total of 3,600 seedlings were planted for this study. Each site had 1,800 seedlings planted representing 300 seedlings per species and planting stock combination. Mississippi State personnel planted bareroot seedlings and conventional containerized seedlings with planting shovels on the first weekend of February 2013. A commercial planting crew planted the EKO™ seedlings with planting shovels in late October 2012. Seedlings were planted next to a pre-marked pin flag to insure proper spacing for uniformity. Each planting job was monitored by a graduate research assistant to ensure planting quality.

Study Design

The Malone study area was established with 1,800 seedlings planted on a 3.05m by 3.05m spacing. The Welford study area was established with 1,800 seedlings planted on a 2.74m by 2.74m spacing. Spacing was altered for the Welford site due to limited land area. A compass and two 300 ft. surveryor’s tapes were used to ensure row straightness and uniform tree spacing. Each study area was divided into three blocked replicates. Six plots containing 100 planting locations were randomly assigned to groups of adjacent rows within each replicate to represent each of the six species and planting stock combinations. Pin flags

of different colors were used to distinguish species and planting stock combinations and mark planting locations. A piece of 1.2m rebar was placed at the beginning and ending of each planting row with an aluminum tag attached denoting the replicate, row number, species, and planting stock. Corners of each study area were marked with 3.0m pieces of polyvinyl chloride (PVC) pipe placed over a 1.2m piece of rebar to ensure no disturbance to the study area.

Herbicide Application

Bareroot and conventional containerized seedlings were treated with a post-plant, pre-bud break application of Oust® XP (140g/sprayed ha) in March of 2013 and 2014. An 11.4L Solo® diaphragm-pump backpack sprayer equipped with a TeeJet 8003 Visiflo® nozzle, specially designed to minimize wind drift, was used to apply the herbicide as a 1.5m band over the top of seedlings.

Seedling Evaluation

Initial groundline diameter (GLD) and height measurements were recorded February 2, 2013. Height of bareroot, conventional containerized and EKO™ seedlings were measured to the nearest centimeter using a meter stick. GLD measurements were measured to the nearest tenth of a millimeter using Mititoyo® digital calipers. First year GLD and height measurements were recorded on November 2, 2013 for both sites. Final measurements were recorded on November 8 and 9, 2014. Only living portions of the dominant stem were measured in height and GLD measurements in the case that a seedling exhibited dieback completely and re-sprouted.

RESULTS AND DISCUSSION

Malone Site

Height Growth—Water oak conventional containerized planting stock had the highest overall total growth (table 1). This may be attributed to the ability of these seedlings being able to mitigate planting shock with the inherent fibrous root system typical to this planting stock. Bareroot seedlings typically need a year to reestablish a root system before allocating resources to height growth. This is evident when comparing year one growth with year two growth. Some water oaks also exhibited high levels of deer browse. These seedlings appeared to be physiologically active when transplanted, which may explain the herbivory and could certainly lead to increased transplant shock, stress, dieback, and mortality. Once bareroot seedlings reestablished roots in the second growing season they added a greater amount of growth. EKO™ seedlings were planted in October which is typically one of the driest months for Mississippi. This factor, when compounded with a shoot to root ratio imbalance and lack of seedling dormancy when planted appeared to result in extreme dieback. The prescribed "plant and walk away" approach for these seedlings which precludes any additional treatment such as herbaceous weed control, may have also contributed to poor performance of these seedlings.

Swamp chestnut oak bareroot seedlings had greater average height growth compared to conventional containerized or EKO™ seedlings. Conventional containerized seedlings had significantly greater height growth in the first growing season than bareroot. Once bareroot seedlings had their roots established, height growth was almost doubled compared to conventional containerized seedling height growth in year two. After two growing seasons, there was no significant difference between bareroot and conventional containerized seedlings regarding average total height growth. EKO™ swamp chestnut oak seedlings appeared to suffer from the same problems as water oak EKO™ seedlings.

GLD Growth—Water oak bareroot and conventional containerized seedlings exhibited positive growth during both growing seasons (table 2). Conventional containerized seedlings had the greatest amount of GLD growth for both growing seasons, with this growth being significantly greater than the other two planting stocks. There was a significant difference between bareroot and EKO™ seedlings at the end of the second growing season. However, bareroot seedlings did not exhibit a significant difference for GLD total growth at the end of year two compared to EKO™ seedlings. EKO™ seedlings exhibited a negative change in GLD the end of the second growing season. The same factors that affected height growth and survival may have contributed to the lesser performance in bareroot and EKO™ seedlings.

Swamp chestnut oak had similar GLD growth to that observed in water oak. At the end of each growing season bareroot and conventional containerized seedlings exhibited positive growth. Conventional containerized seedlings had the greatest GLD growth for each growing season and the greatest amount of overall growth. This growth was significantly different from the other two planting stocks. Bareroot seedlings outperformed EKO™ seedlings during the first growing season. However, bareroot and EKO™ growth did not differ significantly at the end of the second growing season. For overall growth, the amount the bareroot seedlings grew during the first growing season was enough to make its total GLD growth significantly different from EKO™ seedlings. Based on direct observation, the commercial planting crew hired to plant these larger seedlings did not utilize proper

Table 1—Average total height growth for species/planting stock combination for both sites

Height Growth				
Species/Planting Stock	Malone Site		Welford Site	
	cm			
WAO/BR	11.31	B	-0.53	B
WAO/CON	23.39	A	17.85	A
WAO/EKO™	-29.22	C	-12.85	C
SCO/BR	14.18	A	3.12	A
SCO/CON	11.72	A	3.39	A
SCO/EKO™	-29.22	B	-32.10	B

WAO= water oak
SCO= swamp chestnut oak
BR= bareroot
CON= conventional containerized
EKO™= Large containerized

Table 2—Average total groundline diameter growth for species/planting stock combination for both sites

GLD Growth				
Species/Planting Stock	Malone Site		Welford Site	
	mm			
WAO/BR	1.72	B	2.14	B
WAO/CON	5.33	A	5.76	A
WAO/EKO™	0.22	B	2.37	B
SCO/BR	1.97	B	1.88	B
SCO/CON	3.33	A	2.44	A
SCO/EKO™	-0.59	C	1.42	B

WAO= water oak
SCO= swamp chestnut oak
BR= bareroot
CON= conventional containerized
EKO™= Large containerized

seedling care and planting practices. Thus, EKO™ seedlings were possibly adversely affected.

Welford Site

Height Growth—Water oak conventional containerized planting stock had the greatest average total height growth (table 1). It is surmised that conventional containerized seedlings were better able to mitigate planting shock due to the fibrous root system inherent to containerized seedlings. Bareroot seedlings need a year to reestablish a root system after planting before resources are allocated to height growth. This is evident when comparing year one and year two growth. During the second growing season, bareroot seedlings exhibited positive height growth. Deer browse may have also contributed to poor height growth in bareroot seedlings. These factors resulted in bareroot seedlings being placed under increased stress which resulted in bareroot seedlings exhibiting dieback during the first growing season. EKO™ seedlings were planted in October, which is typically one of the driest months in Mississippi. Problems resulting from this early planting time were likely compounded with a shoot to root ratio imbalance ending with increased dieback. The "plant and walk away" approach, advertised for use with EKO™ stock, may have contributed to poor performance.

Swamp chestnut oak conventional containerized seedlings exhibited greater overall height growth compared to bareroot seedlings or EKO™ seedlings. Conventional containerized seedling growth was not significantly different from bareroot seedlings total height growth. Both conventional containerized and bareroot seedlings had similar height growth for both growing seasons. EKO™ swamp chestnut oak seedlings suffered from the same problems as water oak EKO™ seedlings. They were planted outside of the normal planting season, root to shoot ration imbalance and not being dormant when planted may have led to the onset of extreme dieback.

GLD Growth—Water oak seedlings had the same growth pattern for both growing seasons. The conventional containerized seedlings had a significantly greater average GLD growth than the other two planting stocks (table 2). Bareroot and EKO™ seedlings had positive growth at the end of the first growing season but had negative change at the end of the second growing season. There was not a significant difference for GLD growth among these planting stocks at the end of each growing season or for average total growth. The pattern of growth was most similar to that of observed at the Malone site.

Swamp chestnut oak seedlings performed differently than water oak. At the end of the first growing season there was not a significant difference among the three planting stocks. However, at the end of the second growing season conventional containerized seedlings GLD growth was significantly different from the other two planting stocks. Swamp chestnut oak bareroot and EKO™ seedlings had negative GLD growth at the end of the second growing season. Conventional containerized seedlings had the greatest total GLD growth and differed significantly from the other two planting stocks.

CONCLUSIONS

Despite EKO™ seedlings being twice as large in both height and GLD compared to bareroot and conventional containerized seedlings they did not exhibit a clear advantage in height growth or GLD growth. Results of this study show that with proper site preparation, seedling care, quality planting job, and first year herbaceous weed control a land manager can properly reforest lands with a high quality 1-0, bareroot seedling at a much lower cost compared to using containerized planting stock. This study is applicable to land managers trying to regenerate these two oak species on marginal agricultural lands and areas where typical site preparation has been applied on similar soils and expected growing conditions.

LITERATURE CITED

Beck, D.E.; Hooper, R.M. 1986. Development of a southern Appalachian hardwood stand after clearcutting. Southern Journal of Applied Forestry. 10(3): 168-172.

Belli, K.L. 1999. Assessment of the regeneration potential of red oaks and ash on minor bottoms of Mississippi. Southern Journal of Applied Forestry. 23(3): 133-138.

Coder, K. 1994. Southern hardwood management: make every acre work. R8-MB 67. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region: 1-2.

Dey, D.C.; Jacobs, D.; McNabb, K. [and others]. 2008. Artificial regeneration of major oak (*Quercus*) species in the Eastern United States-a review of the literature. Forest Science. 54(1): 77-106.

Dey, D.C.; Parker, W.C. 1997. Morphological indicators of stock quality and field performance of red oak (*Quercus rubra*) seedlings under-planted in a Central Ontario shelterwood. New Forests. 14(2): 145-156.

Ezell, A.W.; Yeiser, J.L.; Nelson, L.R. 2007. Survival of planted oak seedlings is improved by herbaceous weed control. Weed Technology. 21: 175-178.

Hall, D.H.; Lambou, V.W. 1989. Determining the value of bottomland hardwood riverine wetlands to fisheries. In: Wetlands and river corridor management: proceedings of the international wetlands symposium. Charleston, SC: 132-137.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2002. The ecology and silviculture of oaks. Wallingford, Oxon, UK: CABI Publishing, CAB International. 503 p.

Loftis, D.L. 1988. Regenerating oaks on high-quality sites, an update. In: Proceedings of the workshop on guidelines for regenerating Appalachian hardwood stands. Society of American Foresters publ. 88-03: 199-209.

Lorimer, G.L. 1993. Causes of the oak regeneration problem. In: Loftis, D.L.; McGee, C.E., eds. Oak regeneration: serious problems, practical recommendations (symposium proceedings). Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 14-39.

Loftis, D.L.; McGee, C.E., eds. 1993. Oak regeneration: serious problems practical recommendations (symposium proceedings). Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 319 p.

Moree, J.L., Ezell, A.W.; Hodges, J.D. [and others]. 2010. Evaluating the use of enhanced oak seedlings for increased survival and growth: first-year survival. In: Stanturf, J.A., ed. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-20 Asheville, N.C., U.S. Department of Agriculture Forest Service, Southern Research Station: 165-169.

Smith, D.M., Larson, B.C.; Kelty, M.J.; Ashton, P.M.S. 1997. The practice of silviculture: Applied Forest Ecology. 9th ed. New York: John Wiley & Sons, Inc. 537 pp.

Smith, D.W. 1993. Oak regeneration: the scope of the problem. In: Loftis, D.L.; McGee, C.E., eds. 1993. Oak regeneration: serious problems practical recommendations (symposium proceedings). Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 40-52.

Thompson, J.F.; Nix, L.E. 1995. Two-year results of herbicide released, naturally regenerated bottomland cherrybark and Shumard oak seedlings. In: Gottschalk, K.W.; Fosbroke, S.L.C., eds. Proceedings of the 10th central hardwood forest conference. Gen. Tech. Rep. NE-197. Radnor, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station: 341-348.

U.S. Department of Agriculture Natural Resources Conservation Service. Web soil survey. Last modified: 12/06/2013. Available at: http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm. [Date accessed: July 22, 2015].

Witter, R.F. 1991. Direct seeding of bottomland oaks in Oklahoma. Southern Journal of Applied Forestry. 15(1): 17-22.

ASSESSING THE REGENERATION POTENTIAL OF PRODUCTIVE MIXED-HARDWOOD STANDS FOLLOWING SINGLE AND REPEATED PRESCRIBED FIRE

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Abstract—Management efforts on public lands across the southern Appalachian Mountains are increasingly focused on the creation, maintenance, and/or restoration of resilient structures and species compositions, with prescribed burning being the primary tool by which many of these restoration efforts are conducted. In this study, we use regeneration data from a study designed to examine the ecosystem response of upland hardwood forests to prescribed burning in western North Carolina. Four 5-ha productive mixed-hardwood stands were burned between 2009 and 2010. Five years following the first burn (2014), two of the four stands received a second prescribed burn. Regeneration data were collected prior to burning as well as one and five years following the first burn and, for two of the stands, one year following the second burn. Using the REGEN model, we modeled species composition using data obtained from the regeneration inventories to examine how site-preparation burns designed to promote oak seedling development and reduce the abundance of oak competitors may influence post-harvest species composition. Although our results are model forecasts, the underlying data were obtained from actual regeneration inventories. In general, one and two site-preparation burns were forecasted to have little effect on species composition following harvest. The stands used in this study are scheduled to be burned a total of three times prior to implementing a regeneration harvest. Monitoring of the regeneration pool and subsequent success of the various species groups following harvest will be tracked over the long term.

INTRODUCTION

In upland oak-hickory forests of the Central Hardwood Region (CHR), prescribed burning is increasingly used to manage species composition in the forest understory. Objectives associated with burning often include promoting the establishment and development of advance oak reproduction and reducing the abundance of shade-tolerant competitors such as red maple (see app. A for list of scientific names associated with tree species) (Dey and Hartman 2005). The oak regeneration process on intermediate- to high-quality sites, in particular, is complex (Larsen and Johnson 1998), with successful regeneration and recruitment of oak species following overstory disturbance(s) dependent upon the presence of large oak seedlings in the understory prior to disturbance (Sander 1971; Sander 1972; Loftis 1990). Results regarding the efficacy of prescribed fire to promote conditions conducive to the development of competitive advance oak reproduction as well as the ability of prescribed fire to control undesirable, shade-tolerant species are variable (Brose and others 2013; Brose and others 2014). In general, a single burn in closed-canopied, undisturbed oak-hickory forests does little to promote, and in some cases negatively affects, the establishment, growth, and/or abundance of oak reproduction (Brose and others 2013). This has led many to suggest that repeated burning may be necessary to develop competitive oak reproduction and effectively decrease the abundance of shade-tolerant competitors in the forest understory (Albrecht and McCarthy 2006).

Recent reviews of the fire and oak hypothesis (e.g., Brose and others 2014; Brose and others 2013; Arthur and others 2012) suggest the role of fire in sustaining oak forests is complicated, with fire interacting with multiple disturbance agents to affect the oak regeneration process (McEwan and others 2011, Hutchinson and others 2012). Despite growing recognition that prescribed burning alone may not promote the development of competitive oak regeneration across the heterogeneous landscape of CHR (e.g., Iverson and others 2008), burning—either without coupled silvicultural activity (e.g., harvesting) or as a site-preparation tool prior to harvest—continues to be a management tool utilized in mixed-hardwood/

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mixed-oak forests. In this study, we examined the effects of single and repeated prescribed burns on the regeneration potential in productive mixed-hardwood stands in the southern Appalachian Mountains. Specifically, we inventoried the regeneration layer at regular intervals following site-preparation burning and, in conjunction with the regeneration model, REGEN (Loftis 1989), forecasted the effects of single and repeated site-preparation burns on post-harvest species composition.

METHODS

Study site

This study was conducted on the North Carolina Wildlife Resource Commission's Cold Mountain Game Lands (CMGL) in Haywood County, western North Carolina. The CMGL encompasses ~1300 ha and is located in the Blue Ridge physiographic province of the southern Appalachian Mountains. Elevations within the study area range from 975 to 1280 m. Terrain is mountainous with steep slopes. The climate is characterized by warm summers and cool winters. Average monthly temperature ranges from 3 °C in January to 24 °C in July (McNab and Avers 1994). Average annual precipitation approximates 1200 mm and is evenly distributed throughout the year (McNab 2011). In our study, upland oak site index (base age 50) ranged from 23.0 to 30.4 m. Oak (red, white, chestnut, and black oak) and hickory species were the predominant overstory species, whereas sourwood, blackgum, silverbell, flowering dogwood, and red maple dominated the subcanopy positions.

Experimental design and data collection

During the summer of 2008 (prefire), we located four 5-ha mature stands (i.e., the experimental unit) of mixed species composition throughout the CMGL. Within each stand, two transects that ran parallel to the contour were established, with initial transect locations randomly located along the boundary of each stand. The two transects were separated by at least 30 m. Along each of the two transects, three 0.05-ha permanent plots were established at approximately 50, 112, and 175 m along the transects so that each stand contained six 0.05-ha plots—two plots located on each of the lower, middle, and upper slope positions. Within each 0.05-ha permanent plot, we sampled tree regeneration on two 0.004-ha subplots originating 8 m from plot center at bearings of 45° and 225°. Within each subplot, arborescent regeneration sources were tallied by species in the following size classes: (1) <0.6 m tall, (2) 0.6 to <1.2 m tall, (3) ≥1.2 m tall but <3.8 cm diameter at breast height (dbh), and (4) ≥3.8 cm dbh.

On April 1, 2010, two stands (stands 7 and 11) each received a single prescribed burn (hereafter referred to as the 1x burn treatment). Two separate stands (stands 15 and 16) each received two prescribed burns, the first occurring on February 25, 2009, and the second on April 1, 2010 (hereafter referred to as the 2x burn treatment). For all four stands, postfire inventories of the regeneration layer were conducted one and five growing seasons following the first prescribed burn. For stands in the 2x burn treatment, regeneration inventories were also conducted one growing season following the second prescribed burn. All fires were of low intensity and were considered dormant-season fires, as leaf-out had not yet occurred.

Modeling

Using the data collected from the regeneration inventories, we used the REGEN model to examine how site-preparation burns conducted prior to a regeneration harvest may affect post-harvest species composition. In this study, species composition was defined as the percentage of dominant/codominant stems by species group occurring at the time of crown closure following a simulated (i.e., modeled) regeneration harvest. For the 1x burn treatment, we forecasted the effects of site-preparation burning on species composition using the regeneration inventory collected during three time periods: (1) preburn, (2) one year postburn, and (3) five years postburn. For the 2x burn treatment, the effects of site-prep burning prior to harvest on species composition was forecasted using the regeneration data collected during four time periods: (1) preburn, (2) one year following the first burn, (3) five years following the first burn, and (4) one year following the second burn.

The REGEN model is described in depth by Vickers and others (2011). Briefly, REGEN predicts species composition following a stand-replacing disturbance, such as a regeneration harvest, using the following parameters: (1) rankings that quantify the relative competitiveness of various species and possible regeneration sources (i.e, advance reproduction, stump sprouts, root suckers, and new germinants) (table 1); (2) stump sprout probabilities (Keyser and Loftis 2015); and (3) the probabilities of new germinants establishing after harvest. REGEN forecasts species composition by picking six "winners," or species with the highest ranking (i.e., lowest numerical value) on each plot. When a winner is of stump sprout origin, the number of possible winners decreases due to the amount of growing space occupied/required by stump sprouts. Procedures and rules embedded within the model are implemented during situations when regeneration sources of the same rank are chosen as winners. REGEN is operated at the plot level (i.e., winners are selected on each plot) and aggregates plot-level outcomes to produce stand-level estimates of species composition. Low replication (n=2 per burn treatment) prevented meaningful statistical analyses. Consequently, we present simple mean values related to post-harvest species composition per treatment.

Table 1—Competitive rankings of regeneration sources found in Appalachian hardwood forests that are submesic in moisture availability and intermediate in elevation and fertility (Loftis 1989)

Rank	Regeneration source
1	Black cherry-SP; Black locust-SP; Sweet birch-SP; Yellow-poplar-SP; Silverbell-SP; Basswood-SP
2	Eastern white pine-L, White ash-SP; Black cherry-L; Red maple-SP; Sugar maple-SP; Sweet birch-L; Yellow-poplar-L; Cucumber tree-SP; Fraser magnolia-SP
3	Black cherry-M; Black locust-L; Sweet birch-M; Yellow-poplar-M; Basswood-L
4	Eastern white pine-M; Hickory-SP; Southern red oak-SP,L; White ash-L; American beech-SP; Black cherry-M; Black locust-M; Chestnut oak-L,SP; Northern red oak-L,SP; Red maple-L; Scarlet oak-L, SP; Sugar maple-L; Sweet birch-S; Blackgum-SP; Yellow-poplar-S; Black oak-SP,L; Sourwood-SP; Silverbell-L; Cucumbertree-L; Hemlock-L; Fraser magnolia-L; Buckeye-SP; Serviceberry-SP; Sassafras-L,SP
5	Hickory-L; Southern red oak-M; White ash-M; American beech-L,RS; Black cherry-S; Black locust-S,RS; Chestnut oak-M; Northern red oak-M; Red maple-M; Scarlet oak-M; Sugar maple-M; Sweet birch-G; White oak-L, SP; Blackgum-L; Dogwood-SP,L; Yellow-poplar-G; Black oak-M; Sourwood-L; Silverbell-M; Cucumbertree-M; Striped maple-SP; Fraser magnolia-M; Buckeye-L; Serviceberry-L; Basswood-M; Ironwood-SP; Musclewood-SP
6	Eastern white pine-S; Hickory-M; American beech-M; Black cherry-G; White oak-M; Blackgum-M; Sourwood-M; Hemlock-M; Holly-L,SP; Striped maple-L; Buckeye-M; Sassafras-M,RS; Ironwood-L; Musclewood-L
7	Southern red oak-S; White ash-S; Chestnut oak-S; Northern red oak-S; Red maple-L; Scarlet oak-S; Sugar maple-S; Dogwood-M; Black oak-L; Silverbell-S; Cucumbertree-S; Holly-M; Striped maple-M; Fraser magnolia-S; Serviceberry-M; Basswood-S; Sassafras-S; Ironwood-M; Musclewood-M
8	Hickory-S; American beech; White oak-S; Blackgum-S; Dogwood-S; Sourwood-S; Hemlock-S; Holly-S; Striped maple-S; Buckeye-S; Serviceberry-S; Ironwood-S; Musclewood-S

SP=Stump sprout (stems ≥3.8 cm diameter at breast height)

L=Large advance reproduction (seedlings ≥1.2 m and <3.8 cm diameter at breast height)

M=Medium advance reproduction (seedlings ≥0.6 m and <1.2 m)

S=Small advance reproduction (seedlings <0.6 m)

G=Germinants that establish after harvest

RS=Root suckers that establish after harvest

RESULTS AND DISCUSSION

The regeneration layer was diverse and abundant prior to and following the burns. Regardless of sampling period and species group, small seedlings, which are less competitive following harvest than larger seedlings and new germinants that establish post-harvest (Loftis 1990) (table 1), dominated the advance reproduction pool in both the 1x (fig. 1) and 2x (fig. 2) burn treatment stands. We observed a lack of any sizeable advance oak reproduction in our study sites; this is characteristic of undisturbed stands throughout the CHR (e.g., Loftis 1983, Loftis and McGee 1993, Iverson and others 2008).

For the 1x burn treatment, regardless of year since burning, canopy species (table 2) were forecasted to dominate species composition following harvest, constituting, on average, 64 percent of the dominant/codominant canopy stems (table 3). Of the canopy species, black cherry—a species capable of regenerating via new seedling establishment—accounted for 52 percent of the post-harvest dominant/codominant stems. In contrast, yellow-poplar, another species capable of regenerating from new seedlings following harvest, remained a minor component of the stand, ranging from 2 to 7 percent of post-harvest

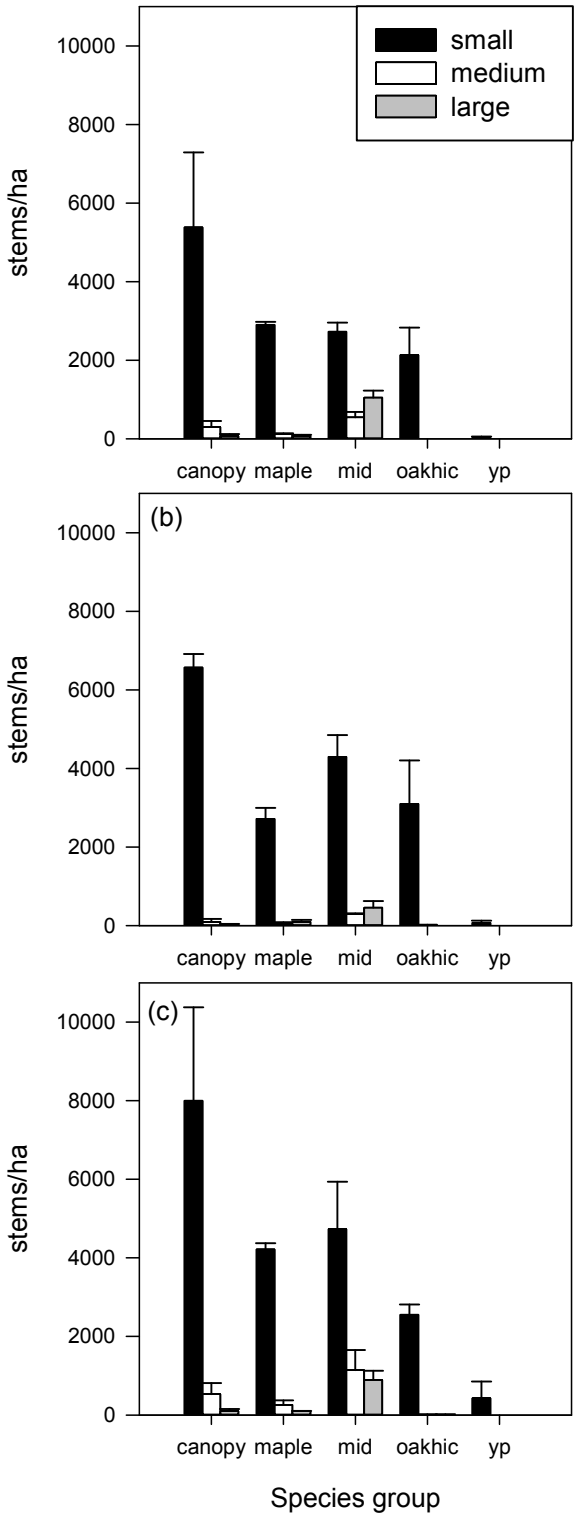


Figure 1—Density of the regeneration layer by species group (table 2) from the 1x burn treatment stands (a) preburn, (b) one year postburn, and (c) five years postburn. Small = advance reproduction <0.6 m, medium = advance reproduction ≥0.6 m and <1.2 m, large = advance reproduction ≥1.2 m and <3.8 cm diameter at breast height. Mid = midstory, oakhic = oak-hickory, yp = yellow-poplar.

species composition. The limited role of yellow-poplar in the post-harvest stand was likely due to these particular stands being located at higher elevations (~1100 m) where forest vegetation begins to transition from upland hardwood forest types to species compositions characteristic of northern hardwood forest types. Regardless of when the regeneration harvest was simulated in relation to time since burn, the contribution of the oak-hickory species group to species composition was predicted to be low (≤2 percent) despite the presence of >2000 seedlings per ha in all regeneration inventories (fig. 1). The lack of the ability of species in the oak-hickory group to compete successfully following harvest was not unexpected, as 99 percent of the oak seedlings were small (<0.6 m) and, consequently, in a noncompetitive position (Sander 1971; Loftis 1990). For maple and midstory species, a single site-preparation burn had little impact on their contribution to species composition following harvest, with maple and midstory species contributing an average of 9 and 20 percent of post-harvest composition, respectively. Although abundant, small maple seedlings, like oak species, were predicted to be unable to compete successfully with faster growing, shade-intolerant species such as black cherry (table 1).

For the 2x burn treatment, the effect of site-preparation burning on the regeneration pool was predicted to increase the amount of yellow-poplar in dominant/codominant positions following harvest (relative to prefire conditions) from 9 to 25 percent (table 4). Unlike the 1x burn treatment, these stands are located at lower elevations where yellow-poplar is abundant and extremely competitive. Relative to other time periods, we observed a 13-percent decrease in the proportion of dominant/codominant oak-hickory stems after harvest when the simulation was conducted one year following the first burn. This is likely due to the 84-percent reduction in large oak seedlings after the first burn (fig. 2). Although the medium and large oak seedling pool experienced a slight recovery, the oak-hickory group was never forecasted to dominate species composition following harvest. Similar to the 1x treatment, little effect of the site-preparation burns on the abundance of species in the midstory group following harvest was observed. For maple, the regeneration pool present one year following the second fire resulted in the lowest proportion of seedlings following harvest, with maple composing 12 percent of the dominant/codominant stems compared to an average of 21 percent over the other three time periods.

Although our results are model forecasts, the data used to conduct model simulations were obtained from regeneration inventories from stands treated with site-preparation burns. Regardless of fire frequency, preharvest site-preparation burning was forecasted to have only minor effects on post-harvest species

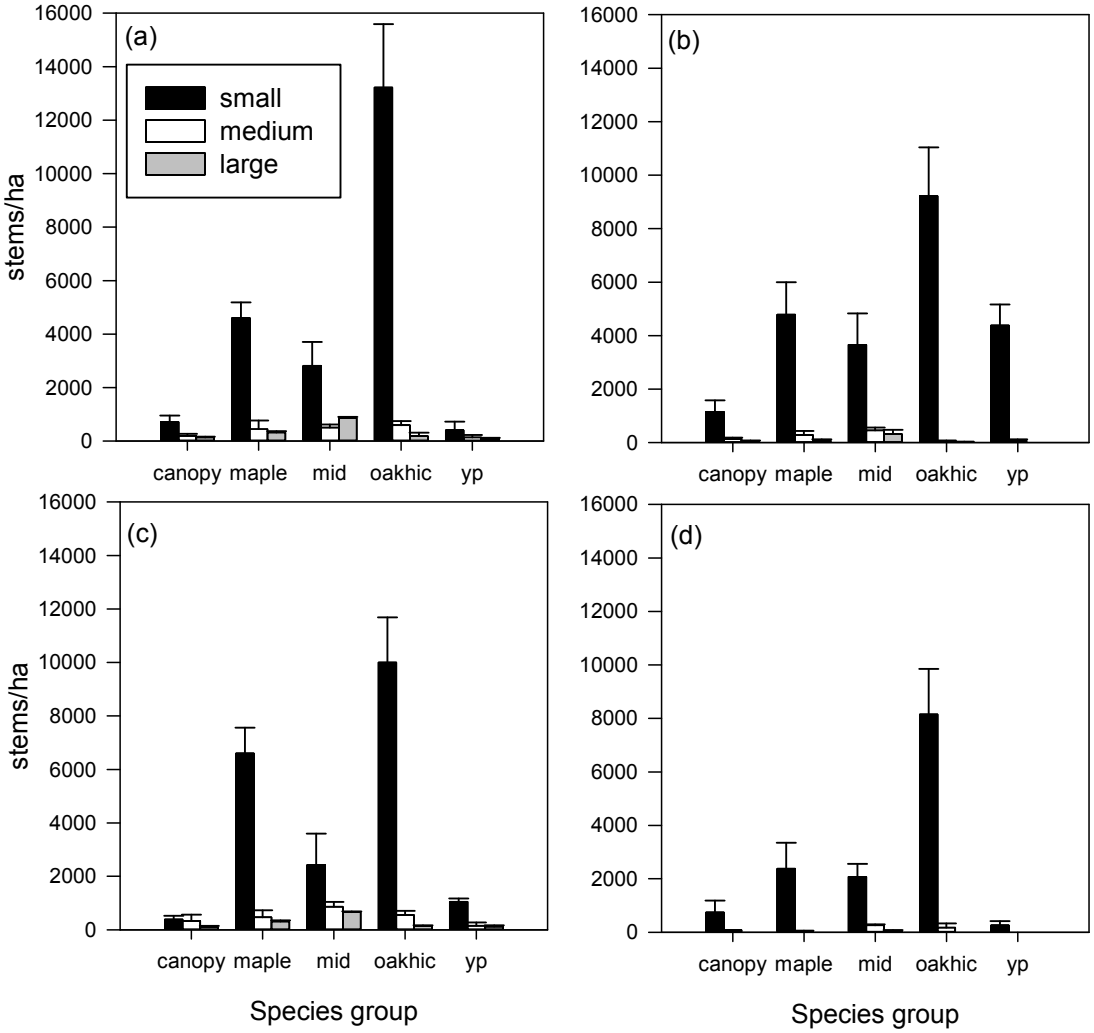


Figure 2—Density of the regeneration layer by species group (table 2) from the 2x burn treatment stands (a) preburn, (b) one year following the first burn, (c) five years following the first burn, and (d) one year following the second burn. Small = advance reproduction <0.6 m, medium = advance reproduction ≥0.6 m and <1.2 m, large = advance reproduction ≥1.2 m and <3.8 cm diameter at breast height. Mid = midstory, oakhic = oak-hickory, yp = yellow-poplar.

composition (table 3, 4). Depending on when the harvest was simulated, site-preparation burns negatively or positively affected oak-hickory abundance in the post-harvest stands; however, overall effects on oak-hickory were minor and likely of little biological significance. From the regeneration inventory data, site-preparation burning was ineffective at recruiting small advance oak-hickory reproduction into larger, more competitive size classes (figs. 1 and 2). In Kentucky, although repeated burns significantly reduced the survival of small red maple seedlings, the competitive status of oak stems remained unchanged due to vegetative resprouting of fire-killed stems and rapid recovery of canopy cover (Alexander and others 2008, Blankenship and Arthur 2006). The limited response of oak-hickory species to burning in this study and elsewhere suggests that burning coupled with some type of canopy removal [and continued competition control via preharvest removal of stump sprout potential (Loftis 1985), continued burning, and/or chemical release treatments] will be required to develop competitive advance oak reproduction and facilitate overstory recruitment (e.g., Hutchinson and others 2012, Arthur and others 2012, Brose 2010; Iverson and others 2008).

MANAGEMENT IMPLICATIONS

Securing oak regeneration is problematic on productive southern Appalachian hardwood forests (Loftis 1990). Therefore, prescribed burning is increasingly used to promote the abundance and competitiveness of advance oak reproduction. Brose and others (2013) suggest site-preparation burning can be used to

Table 2—Species composing species groups

Species group	Species
Canopy	Sweet birch, black cherry, yellow birch, white ash, black locust, black walnut, buckeye, white basswood, Fraser magnolia, Cucumbertree
Maple	Red maple, sugar maple
Midstory	Striped maple, flowering dogwood, alternate dogwood, sourwood, holly, ironwood, musclewood, American beech, silverbell, blackgum, serviceberry, sassafras
Oak-hickory	Hickory species, northern red oak, black oak , scarlet oak, chestnut oak, white oak
Yellow poplar	Yellow poplar
Other	Eastern white pine, shortleaf pine, Virginia pine, pitch pine, eastern hemlock

Table 3—The proportion of the dominant/codominant stems by species group (table 2) forecasted to occur at the time of crown closure by the REGEN model using regeneration inventory data from the 1x burn treatment prior to the fire, one year postburn, and five years postburn. Rounding resulted in some values that sum to <100 percent

Time period	Percentage of dominant/codominant stems					
	Canopy	Maple	Midstory	Oak-hickory	YP	Other
Prefire	65	8	18	2	7	0
One year postfire	64	9	23	2	2	0
Five years postfire	63	10	18	1	7	0

YP = yellow poplar

Table 4—The proportion of the dominant/codominant stems by species group (table 2) forecasted to occur at the time of crown closure by REGEN using regeneration inventory data from the 2x burn treatment prior to the fire, one year following the first fire, five years following the first fire, and one year following the second fire. Rounding resulted in some values that sum to <100 percent

Time period	Percentage of dominant/codominant stems					
	Canopy	Maple	Midstory	Oak-hickory	YP	Other
Prefire	32	25	2	14	25	2
One year after 1st fire	30	18	<1	1	50	0
Five years after 1st fire	29	20	2	10	38	0
One year after 2nd fire	35	12	2	18	34	0

YP = yellow poplar

promote the establishment and competitiveness of oak species prior to harvest. In addition to developing the existing advance reproduction pool, Schuler and others (2010) suggest that site-preparation burning can be used to deplete the seed bank of competitive mesophytic species such as yellow-poplar, sweet birch, and black cherry, a result that contrasts findings specific to our study sites (Keyser and others 2012). Although we report the effects of site-preparation burning on post-harvest species composition after one and two burns, Brose and others (2013) recommend site-preparation burning be conducted up to 10 years prior to harvest if it is to positively affect the abundance of oak and overall species composition in the post-harvest stand. Furthermore, for the most effective competition control, Brose (2010) recommends that burning be conducted during the early growing season, at a time when oak competitors (e.g., maple species) are breaking bud. Burn windows are limited in the southern Appalachians, where precipitation is abundant and evenly distributed throughout the year. Consequently, limiting site-preparation burns to a specific period during the burn season would further reduce the likelihood of accomplishing burns. The stands used in this study are scheduled to be burned a total of three times prior to implementing a regeneration harvest. Monitoring of the regeneration pool and subsequent success of the various species groups following harvest will be tracked over the long term.

LITERATURE CITED

Albrecht, M.A.; McCarthy, B.C. 2006. Effects of prescribed fire and thinning on tree recruitment patterns in central hardwood forests. Forest Ecology and Management. 226: 88-103.

Alexander, H.D.; Arthur, M.A.; Loftis, D.L.; Green, S.R. 2008. Survival and growth of upland oak and co-occurring competitor seedlings following single and repeated prescribed fires. Forest Ecology and Management. 256: 1021-1030.

Arthur, M.A.; Alexander, H.D.; Dey, D.C. [and others]. 2012. Refining the oak-fire hypothesis for management of oak-dominated forests of the eastern United States. Journal of Forestry. 110: 257-266.

Blankenship, B.A.; Arthur, M.A. 2006. Stand structure over 9 years in burned and fire-excluded oak stands on the Cumberland Plateau, Kentucky. Forest Ecology and Management. 225: 134-145.

Brose, P.H. 2010. Long-term effects of single prescribed fires on hardwood regeneration in oak shelterwood stands. Forest Ecology and Management. 260: 1516-1524.

Brose, P.H.; Dey, D.C.; Phillips, R.J.; Waldrop, T.A. 2013. A meta-analysis of the fire-oak hypothesis: Does prescribed burning promote oak reproduction in eastern North America. Forest Science. 59: 322-334.

Brose, P.H.; Dey, D.C.; Waldrop, T.A. 2014. The fire-oak literature of eastern North America: Synthesis and guidelines. Gen. Tech. Rep. NRS-135. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 98 p.

Dey, D.C.; Hartman, G. 2005. Returning fire to Ozark Highland forest ecosystems: Effects on advance regeneration. Forest Ecology and Management. 217:37-53.

Hutchinson, T.F.; Long, R.P.; Rebbeck, J. [and others]. 2012. Repeated prescribed fires alter gap-phase regeneration in mixed-oak forests. Canadian Journal of Forest Research. 42: 303-314.

Iverson, L.R.; Hutchinson, T. F.; Prasad, A.M.; Peters, M.P. 2008. Thinning, fire, and oak regeneration across a heterogeneous landscape in the eastern U.S.: 7-year results. Forest Ecology and Management. 255: 3035-3050.

Keyser, T.L.; Loftis, D.L. 2015. Stump sprouting of 19 upland hardwood species 1 year following in initiation of a shelterwood with reserves silvicultural system in the southern Appalachian Mountains, USA. New Forests. 46: 449-464.

Keyser, T. L.; Roof, T.; Adams, J.L. [and others]. 2012. Effects of prescribed fire on the buried seed bank in mixed-hardwood forests of the southern Appalachian Mountains. Southeastern Naturalist. 11: 669-688.

Larsen, D.R.; Johnson, P.S. 1998. Linking the ecology of natural oak regeneration to silviculture. Forest Ecology and Management. 106: 1-7.

Loftis, D.L. 1983. Regenerating southern Appalachian mixed-hardwood stands with the shelterwood method. Southern Journal of Applied Forestry. 7: 212-217.

Loftis, D.L. 1985. Preharvest herbicide treatment improves regeneration in southern Appalachian hardwoods. Southern Journal of Applied Forestry. 9: 177-180.

Loftis, D.L. 1989. Species composition of regeneration after clearcutting Southern Appalachian hardwoods. In: Proceedings of the 5th biennial southern silvicultural research conference. Miller, J.H., comp. Gen. Tech. Rep. SO-74. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experimental Station. 618 p.

Loftis, D.L. 1990. Predicting postharvest performance of advance red oak reproduction in the southern Appalachian. Forest Science. 36: 908-916.

Loftis, D.L., McGee, C.E. (Eds.). 1993. Oak regeneration: serious problems, practical recommendations. Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 319 p.

McEwan, R.W.; Dyer, J.M.; Pederson, N. 2011. Multiple interacting ecosystem drivers: toward and encompassing hypothesis of oak forest dynamics across eastern North America. Ecography. 34: 244-256.

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

McNab, W.H. 2011. Subregional variation in upland hardwood forest composition and disturbance regimes of the Central Hardwood Region. In: Greenberg, C.H.; Collins, B.; Thompson, F.R.,III, eds. Sustaining young forest communities: Ecology and management of early successional habitats in the Central Hardwood Region, US. Dordrecht, Netherlands: Springer: 11-26.

Sander, I.L. 1971. Height growth of new oak sprouts depends on size of advance reproduction. Journal of Forestry. 69: 809-811.

Sander, I.L. 1972. Size of advance reproduction key to growth following harvest cutting. Research Paper NC-79. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 6 p.

Schuler, T.M.; Thomas Van-Gundy, M.; Adams, M.B.; Ford, W.M. 2010. Seed bank response to prescribed fire in the central Appalachians. Research Paper NRS-9. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 9 p.

Vickers, L.A.; Fox, T.R.; Loftis, D.L.; Boucugnai, D.A. 2011. Predicting forest regeneration in the central Appalachians using the REGEN expert system. Journal of Sustainable Forestry. 30: 790-822.

Biomass

Moderator:

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IMPACT OF THINNING ON SOIL PROPERTIES AND BIOMASS IN APALACHICOLA NATIONAL FOREST, FLORIDA

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Abstract—The effect of a silvicultural operation, row thinning at two intensities (single row, SR, and double row, DR, thinning), on soil properties and biomass were investigated in selected 28 year-old slash pine (*Pinus elliotti*) plantations in the Apalachicola National Forest. Stands were thinned in May 2011 and burn regimes were executed during dormant seasons. Indicators of changes in soil physical properties and biomass were evaluated in this work. Response variables included soil bulk density, stand basal area, and biomass. Stand basal area was consistently highest across locations in the control treatments ranging from 21 to 31 m^2 ha^{-1}. In two locations, SR treatment had the greatest DBH and DGL values and was significantly different ($p > 0.05$) in location 2. The average soil bulk density at two depths, 0 and 15 cm were 1.51 g cm^{-3}, 1.49 g cm^{-3}, and 1.44 g cm^{-3}, and 1.62 g cm^{-3}, 1.64 g cm^{-3}, and 1.62 g cm^{-3} for the SR, DR and control treatments, respectively with no significant treatment effects observed. Varying results in understory biomass was detected with as much as 30 percent and 24 percent reduction for SR and DR respectively at location 2 while there was a 3 percent and 15 percent increase for SR and DR respectively at location 1. Aboveground biomass had significant reduction ($p > 0.05$) due to treatment effects with a range of 12 percent to 36 percent for SR and 42 percent to 51 percent for DR across locations.

INTRODUCTION

Southern forests have an increased timber productivity in recent years that is attributed to increased utilization of intensive management (Grace and others 2006), but broad concern has been expressed related to the potential productivity decline and long term adverse impacts in intensively managed forests (Eisenbies 2006, Haywood and Tiarks 1990, Miwa and others 2004, Powers and others 2010). Intensive forest management practices include site preparation, thinning, harvesting, and fertilization with the majority of these management practices performed by heavy machinery (Grace and others 2006). Previous studies have shown that intensive management operations can cause a myriad of effects ranging from alteration of soil physical properties and lowering of site productivity (Eisenbies 2006, Gent and others 1984, Powers and others 2004), increase in the risk of erosion (Carter and others 2006), change in the rate of transpiration and water table dynamics (Bliss and Comerford 2002; Skaggs and others, 2008), and change in soil moisture (Gent and others 1984; Grace and others 2006; Kozlowski, 1999).

The present management plan for the Apalachicola National Forest (ANF) is the regeneration of long-leaf (*Pinus palustris*) pine as the forest's dominant species as was in pre European settlement times and in the early 1950's (US Forest Service Management Plan 1999). Thinning and prescribed burning are two management techniques used by the US Forest Service in actualizing the pre-specified management goal for ANF (US Forest Service Management Plan 1999). Thinning, a silvicultural practice that selectively removes poorly performing trees and leaves a healthy vigorous stand, increases growth of residual trees, develops structural characteristics needed for wildlife habitats, and generally produces healthy, vigorous forests with less risk of insect infestation, destructive fire, and wind damage (Bradford and Palik 2009, Demers and others 2013, Harrington 2001). Whereas, prescribed burning is the deliberate application of fire to forest fuels to achieve site preparation, reduction of woody understory competition, and restoration, or maintenance of fire-dependent ecosystems (Carter and Foster 2004, Fernandes and Botelho 2003).

The objective of this study is to quantify the impact of two levels of thinning (single row (SR) and double row (DR) thinning) in conjunction with prescribed burning on soil bulk density in forest watersheds in the panhandle region of Florida, specifically, the Apalachicola National Forest (ANF). Additionally, the study also investigates the combined influence of these two silvicultural treatments (i.e. thinning with prescribed burning) on

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stand characteristics, and, aboveground and understory biomass.

MATERIALS AND METHODS

Study Site

The study area was located in compartment 246 in the Northeastern portion of the Apalachicola National Forest approximately 30° 18' latitude and 84° 27' longitude in Leon County, FL (fig. 1). The site is poorly drained and has a shallow water table with a depth to water table between 0 to 46 cm. Soils are primarily Leon hydric sands and the vegetation in the study area predominantly consists of a 28 year-old slash pine (*Pinus elliotti*) plantation with small cleared patches planted with longleaf pines (*Pinus palustris*) in 2012.

Experimental Design

The experimental design superimposed on this study was a split-plot design with locations as the main plots and management practices as the sub-plots. Three experimental locations were randomly selected to give an unbiased representation of the forest stand properties and watershed characteristics of compartment 246. Nine '7.3 m radius' sub-plots within each location were randomly created using a Microsoft® Excel© program listing directions and bearings from a physical landmark. The random program was designed so that treated plots and corresponding control plots within a location had similar soil types so as to limit the influence of soil variability on treatment effects.

The subplots consisted of two treatments namely; double row thinning with prescribed fire (DR), single row thinning with prescribed fire (SR) and, a no thinning with prescribed fire (C). The DR treatment consisted of thinning two rows for every two rows of pine trees, whereas SR treatment consisted of thinning one row for every two rows, and C treatment had no thinning. All treatments had controlled burning during the dormant season of 2014. A split-plot design was chosen for this study because the treatments and locations had already been implemented according to existing Forest Service management plans for that compartment. Locations

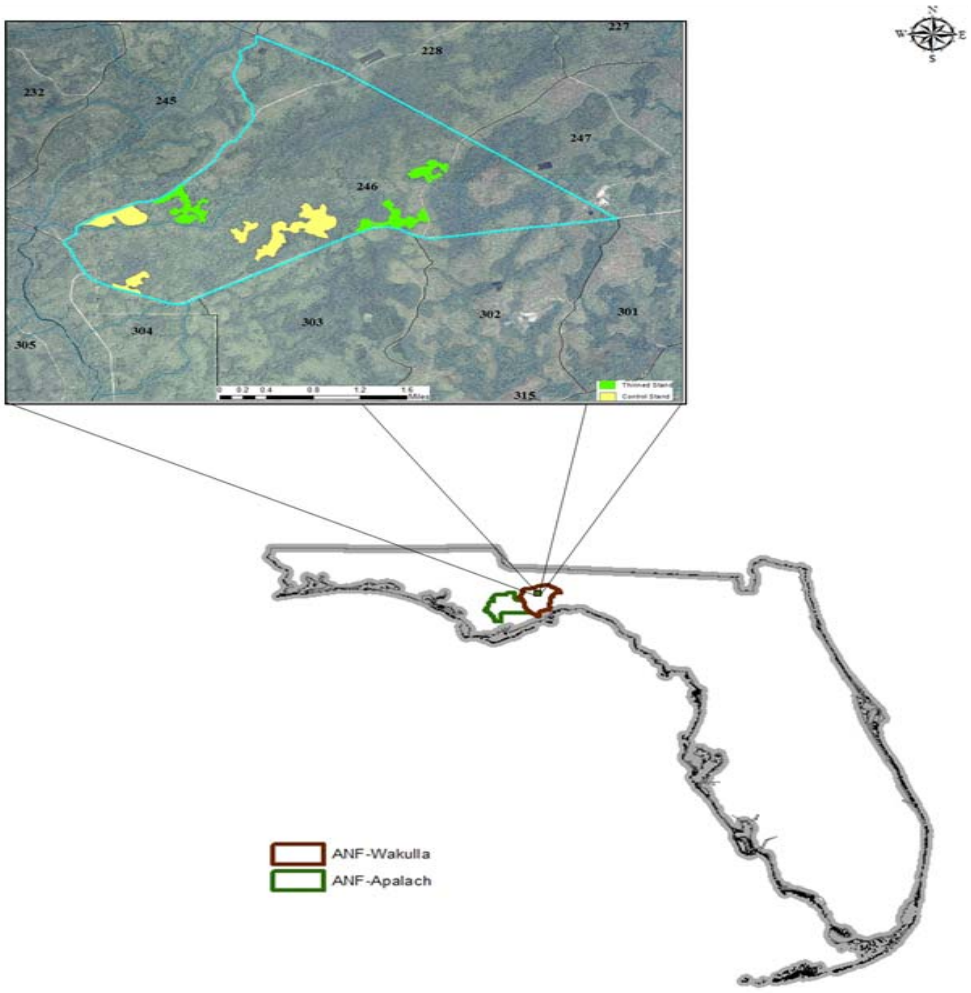


Figure 1—Location of the study area in compartment 246 of the Wakulla District of the ANF within the panhandle region of Florida. Thinned stands are indicated in green and control stands are indicated in yellow.

1 and 3 consisted of nine '7.3 m radius' sub-plots whereas location 2 had six sub-plots with a randomly located "1 m radius" destructive sampling plot within each sub-plot. The disparity in plot number in location 2 was due to the fact that only two management practices had been implemented on that location and this difference would serve as another means of comparing the different treatments. Plot boundaries were surveyed and measured with a compass and logger's tape, and corners or plot centers were marked with flags.

Bulk Density

Soil cores were collected using an AMS 1000 core sampler (5.2 cm diameter by 6 cm length) during two extensive field expeditions in July to September and November to December, 2014. Three core samples were randomly collected from each sub-plot at two depths (0 - 15, and 15 – 30 cm). Collected cores were trimmed in the field, sealed in plastic bags, labelled, and transported to the lab for subsequent bulk density determination. A total of 108 soil cores were collected from the twenty-four sub-plots to describe baseline and post - treatment soil conditions.

Biomass

Overstory biomass (consisting of all standing pine trees) and understory biomass (all living understory biomass) were quantified in all twenty-four sub-plots. Variables used to estimate overstory biomass included: Diameter at Ground Level (DGL), Diameter at Breast Height (DBH), Total Height (TH) and Total Height to first Living Branch (THLB) of the slash pine plantation. The data were used to calculate stand basal area, and, estimate the biomass of slash pine trees within each plot using allometric equations. Understory biomass was quantified in each of the randomly located "1 m radius" destructive sampling plots within each sub-plot. All living understory biomass was severed in the destructive sampling plots, weighed in the field, and ground into a homogenous mixture of all biomass components. Subsequently, a representative sub-sample of the mixture was collected, labelled, and transported to the lab for further nutrient analysis in a separate future study.

Data Analysis

The general model for each treatment response variable was:

$$Y_{ijk} = \mu\ldots + \alpha_i + \beta_j + (\alpha\beta)_{ij} + \gamma k(i) + \varepsilon_{ijk}$$

where: Y_{ijk} is the dependent variable; μ is a constant; α_i is the treatment effects; β_j is the sub-plot effects; $(\alpha\beta)_{ij}$ is the treatment-subplot interaction; $\gamma k(i)$ is the treatment errors; ε_{ijk} is treatment errors.

Response variables in the investigation; bulk density, total biomass, and stand basal area, were tested by the ANOVA. The null hypothesis is that thinning treatment will have no effect on response variables. Tukey multiple range tests, alpha = 0.05, were used to separate means where ANOVA detected significant differences.

RESULTS AND DISCUSSION

Stand Characteristics

The slash pine density in locations 1, 2 and 3 were 2,518, 1,666, and 2,360 trees ha $^{-1}$, respectively. The average tree density for SR, DR and C treatments, respectively were 720, 654 and, 1018 trees ha $^{-1}$. The estimated aboveground biomass for the plantation pine was 97, 74, and 123 tonnes ha $^{-1}$ for the SR, DR, and C conditions, respectively. The estimated biomass contained in the understory vegetation was 2.7, 3.3, and 3.3 tonnes ha $^{-1}$ for the SR, DR, and C treatments, respectively. The similarity in the distribution of the various locations' slash pine trees shows that although the locations and sub-plots were randomly located, the sites gave a fairly unbiased accurate representation of compartment 246 (fig. 2).

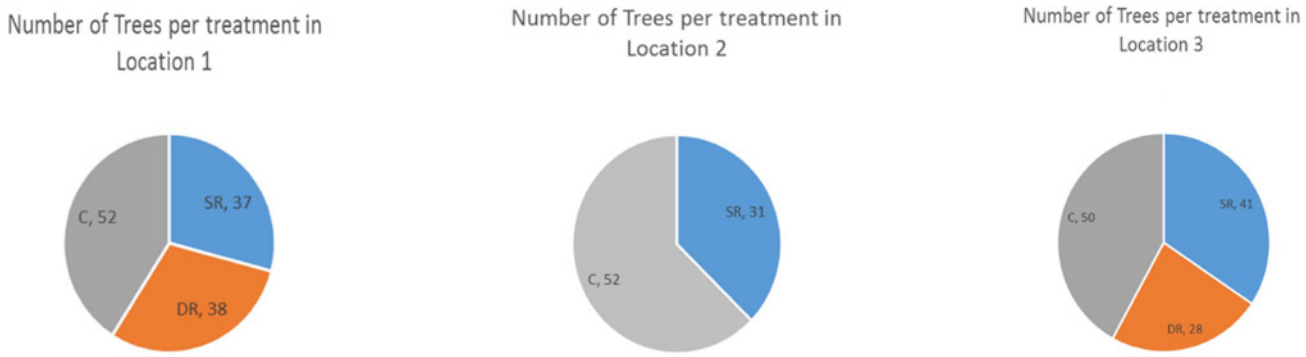


Figure 2—Pie-chart showing distribution of trees amongst the treatments in each location (SR – Single Row thinning; DR – Double Row thinning; C – Control).

Stand Basal Area, Diameter at Breast Height and Diameter at Ground Level

Average stand basal area (SBA) was calculated from measurements of all slash pine trees in each of the three locations using the standard equation;

Basal area = (pi / (4 * 144)) × (DBH) ^{2}

* where pi = 3.14, DBH = diameter breast height (Barlow and Elledge 2012).

The range of average SBA ranged from 12 to 31 m^2 ha^{-1} with location 1 having the largest average SBA (table 1). The control had the greatest stand basal area followed by the single row and the double row thinning in that order. Although statistically significant differences were only observed in location 1, the other locations had evident differences in SBA with the double row treatment always having the lowest. This can be explained by the fact that more trees were removed in the DR treatment (two rows) as compared to the SR treatment (one row). The level of differences in similar treatments among locations could probably be attributed to intrinsic location characteristics and variations.

DBH at 1.4 m was measured for all plantation slash pine within inventory plots at each location. With the exception of location 1, DR had the lowest average DBH amongst the treatments while SR always had the greatest DBH ranging from 17.4 to 18.7 cm. Significant differences ($p > 0.05$) were observed between SR and C with SR having an average DBH of 18.7 cm compared to C having an average of 15.2 cm in location 2. SR, DR, and C, respectively, had an average DBH of 17.5, 16.7. and 18.6 cm in location 1, and 17.4, 15.6 and 16.6 cm in location 3. Presumably, SR having the greatest DBH with the exception of location 1 was a result attributed to it having the closest stand basal area to the intended management basal area of 16 m^2 ha^{-1} coupled with the fact that diameter growth is significantly affected by stand density (Bradford and Palik, 2009; US Forest Service Management Plan, 1999). It can also be as a result of reduced disturbance in the form of compaction, or less competition in the SR plots in comparison to the DR plots which had more mechanical disturbances per similar unit area.

An identical trend was also observed in the DGL data. SR had the greatest average DGL in location 1 and 2 but was slightly lower than DR in location three. Similar to the case in the DBH data significant differences between the two treatments, SR and C were detected in location 2.

Bulk Density

Soil bulk densities (Db) in the 0 - 15 cm averaged 1.48 g cm^{-3}, 1.48 g cm^{-3}, and 1.49 g cm^{-3} for the SR, DR, and C treatments, respectively in location 1 (table 2). Whereas in location 3, Db increased from 1. 39 g cm^{-3} for C to 1.49 g cm^{-3} for DR and 1.54 g cm^{-3} for SR. At the 15 – 30 cm depth, Db averages were 1.63 g cm^{-3}, 1.66 g cm^{-3} and 1.64 g cm^{-3} for the SR, DR, and C treatments, respectively in location 1. Db averages at that same depth were 1.60 g cm^{-3}, 1.61 g cm^{-3}, and 1.59 g cm^{-3} respectively for the SR, DR, and C treatments in location 3.

There were no statistical significant differences in Db ($p > 0.05$) among the treatments in the different locations at both depths but at the 0 – 15 cm depth in location 3, SR>DR>C. A plausible explanation for the observation of no treatment effects on Db may be ascribed to the inherent soil properties of the sites, primarily consisting of sandy soils, which has the property of being less compactable than other soil types (Gent and others 1984). Additionally, considering thinning was implemented in 2011 the detection of no treatment effects may be as a result of natural rejuvenation of the soils from specific site activities such as increase in organic matter from thinning residues, root growth, micro-organism activities and environmental conditions (Da-Lun and others 2010, Powers and others, 2004).

Table 1—Calculated stand basal area (m^2 ha^{-1}), average diameter at breast height (cm), and average diameter at ground level (cm) across treatments per location

	SBA (m^2 ha^{-1})			DBH (cm)			DGL (cm)		
	SR	DR	C	SR	DR	C	SR	DR	C
Location 1	20a	18a	31b	17.5a	16.7a	18.6a	22.8a	20.9a	22.8a
Location 2	18a		21a	18.7b		15.2a	24.8b		19.8a
Location 3	21a	12a	24a	17.4a	15.6a	16.6a	21.6a	21.7a	21.5a

*[Means in same row followed by different letters are significantly different at $P < 0.05$, using Tukey's]

Understory Biomass

Understory biomass ranged from 2.1 to 3.7 tonnes ha^{-1} varying on the treatment and location (table 3). The control treatment had the most biomass with 3 tonnes ha^{-1} and 3.7 tonnes ha^{-1} in location 2 and 3, respectively. While in location 1, the control had the least biomass with 3.2 tonnes ha^{-1} whereas DR had the greatest with 3.7 tonnes ha^{-1}. This goes contrary to the expectation of thinning opening up treated stands to more effective burn regimes (Carter and Foster 2004) generally resulting in unthinned (control) forest stands having the most understory biomass. Probably, this deviation can likely be attributed to a less severe or intense burn regime in location 1, environmental conditions during burn schedule or native site characteristics (e.g. high water table, ephemeral ponds).

Aboveground Biomass Estimation

An allometric equation from Gonzalez-Benecke and others (2014) was used to estimate standing whole tree biomass (including trunk, stems, barks, limbs and foliage) for each plot in this study. The equation in its simplified form is:

$[T = b_1. (DBH^{b2}). (AGE^{b3})]$

* where T = total above stump biomass; DBH = diameter at breast height; AGE = age of tree; b1, b2, and b3 = empirical coefficients dependent on tree species and geographical location.

The biomass estimates (table 4) show that control as hypothesized always had the most aboveground biomass. DR always had the lowest biomass estimates with as much as a 42 percent and 51 percent decrease in biomass compared to the control treatments in location 1 and location 3. Significant treatment effects were observed in location 1 although noticeable differences were also observed in locations 2 and 3. In location 3, there was a huge difference in the biomass estimates between SR and DR treatments which may be due to two sites in the DR treatments having relatively smaller number of trees, 6 and 9, compared to the average number of 13 trees per DR plot.

Table 2—Average soil bulk density (Db) at 0 cm and 15 cm for the three treatments at the different locations

	Db at 0cm (g cm^{-3})			Db at 15cm (g cm^{-3})		
	SR	DR	C	SR	DR	C
Location 1	1.48a	1.49a	1.49a	1.63a	1.66a	1.64a
Location 3	1.54a	1.49a	1.39a	1.60a	1.61a	1.59a

*[Means in same row followed by different letters are significantly different at $P < 0.05$, using Tukey's]

Table 3—Understory biomass (i.e. all living herbaceous plants) for SR and DR treatments showing percentage change with respect to the control

	Understory biomass (tonnes ha^{-1}) (% change w.r.t Control)		
	SR	DR	C
Location 1	3.3a (3%)	3.7a (15%)	3.2a
Location 2	2.1a (-30%)		3a
Location 3	2.6a (-30%)	2.8a (-24%)	3.7a

*[Means in same row followed by different letters are significantly different at $P < 0.05$, using Tukey's]

Table 4—Aboveground biomass estimates for SR and DR treatments showing percentage change with respect to the control in all locations

Aboveground biomass (tonnes ha^{-1}) (% change w.r.t Control)			
	SR	DR	C
Location 1	98.5a (-36%)	89.3a (15%)	154.4b
Location 2	87.9a (-9%)		96.8a
Location 3	104.3a (-12%)	58.2a (-51%)	118.9a

*[Means in same row followed by different letters are significantly different at P < 0.05, using Tukey's]

CONCLUSION

The effect of thinning combined with prescribed burning was investigated on stand characteristics, soil bulk density, and biomass. Management practices resulted in decrease in SBA and stand density in the SR and DR treatments with significant differences (p > 0.05) in location 1. No significant differences were detected in soil bulk density across the treatment. This observation may be attributed to the sandy nature of the soils in majority of the sites. The opening of forest stands due to thinning and the application of prescribed burns resulted in an effective understory control with the no thinning (control) treatment always having the greatest understory biomass except in location 1. The unthinned treatment, C, had the most aboveground biomass with SR and DR having significant reduction (p > 0.05) as much as 36 percent and 51percent reduction, respectively.

LITERATURE CITED

Barlow, B.; Elledge, J. 2012. Basal area: a measure made for management. ANR-1371. Auburn, AL: Alabama Cooperative Extension System. 4 p.

Bliss, C.; Comerford, N.B. 2002. Influence of forest harvesting on water table dynamics in a Florida flatwoods landscape. Soil Science Society of America Journal. 66: 1344-1349.

Bradford, J.B.; Palik, B.J. 2009. A comparison of thinning methods in red pine: consequences for stand-level growth and tree diameter. Canadian Journal of Forest Research. 39(3): 489-496.

Carter, E.A.; Rummer, R.B.; Stokes, B.J. 2006. Evaluation of site impacts associated with three silvicultural prescriptions in an upland hardwood stand in northern Alabama, USA. Biomass and Bioenergy. 30: 1025-1034.

Carter, M.C.; Foster, C.D. 2004. Prescribed burning and productivity in southern pine forests: a review. Forest Ecology and Management. 191: 93-109.

Da-Lun, T.; Yuan-Ying, P.; Wen-De, Y., [and others]. 2010. Effects of thinning and litter fall removal on fine root production and soil organic carbon content in Masson pine plantations. Pedosphere. 20(4): 486-493.

Demers, C.; Andreu, M.; McGowan, B. [and others]. 2013. Thinning southern pines - a key to greater returns. Series of the School of Forest Resources and Conservation, SS FOR 24. Florida Cooperative Extension Service, Institute of Food and Agricultural Sciences, Tallahassee, FL: University of Florida. 6 p.

Eisenbies, M. H. 2006. Long-term timber productivity research on intensively managed pine forests on the South. In: Ireland, L. C.; Camp, A. E.; Brissette, J. C.; Donohew, Z.R., eds. Long-term silvicultural and ecological studies: results for science and management. GISF Res. Pap. 005. New Haven, CT: Yale University: 139-153.

Fernandes, P.M.; Botelho, H.S. 2003. A review of prescribed burning effectiveness in fire hazard reduction. International Journal of Wildland Fire. 12: 117-128.

Gent, J. A., Jr.; Ballard, R.; Hassan, A.E.; Cassel, D.K. 1984. Impact of harvesting and site preparation on physical properties of Piedmont forest soils. Soil Science Society of America Journal. 48: 173-177.

Gonzalez-Benecke, C.A.; Gezan, S.A.; Albaugh, T.J. [and others]. 2014. Local and general above-stump biomass functions for loblolly pine and slash pine trees. Forest Ecology and Management. 334: 254-276.

Grace, J.M. III; Skaggs, R.W.; Cassel, D.K. 2006. Soil physical changes associated with forest harvesting operations on an organic soil. Soil Science Society of America Journal. 70(2): 503–509.

Harrington, T.B. 2001. Silvicultural basis for thinning southern pines: concepts and expected responses. Report No: FSP001. Athens, GA: Warnell School of Forest Resources, Georgia Forestry Commission. 13 p.

Haywood, J.D., and Tiarks, A.E. 1990. Eleventh-year results of fertilization, herbaceous, and woody plant control in a loblolly pine plantation. Southern Journal of Applied Forestry. 14(4): 173-177.

Kozlowski, T.T. 1999. Soil compaction and growth of woody plants. Scandinavian Journal Forest Research. 14: 596-619.

Miwa, M.; Aust, W.M.; Burger, J. A. [and others]. 2004. Wet-weather timber harvesting and site preparation effects on Coastal Plain sites: a review. Southern Journal of Applied Forestry. 28 (3): 137-151.

Powers, M.D.; Palik, B.J.; Bradford, J.B. [and others]. 2010. Thinning method and intensity influence long-term mortality trends in a red pine forest. Forest Ecology and Management. 260(7): 1138-1148.

Powers, R. F.; Sanchez, F.G.; Scott, D. A.; Page-Dumroese, D. 2004. The North American long-term soil productivity experiment: coast-to-coast findings from the first decade. In: Shepperd, W.D.; Eskew, L.G., comps. Silviculture in special places: proceedings of the national silviculture workshop. RMRS-P-34. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 191-206.

Skaggs, R.W.; Chescheir, G.M.; Amatya, D.M.; Diggs, J.D. 2008. Effects of drainage and forest management practices on hydraulic conductivity of wetland soils. In: Farrell, C.; Feehan, J., eds. Proceedings of the 13th international peat congress: after wise use- the future of peatlands. Jyvaskyla, Finland: International Peat Society Volume 1: 452-456.

United States Department of Agriculture, Forest Service Southern Region. 1999. Record of decision for the revised land and resource management plan for National Forests in Florida. Management Bulletin R8-MB-83C. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region. 37 p.

OPTIMAL MANAGEMENT AND PRODUCTIVITY OF EUCALYPTUS GRANDIS ON FORMER PHOSPHATE MINED AND CITRUS LANDS IN CENTRAL AND SOUTHERN FLORIDA: INFLUENCE OF GENETICS AND SPACING

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Abstract—*Eucalyptus* short rotation woody crops (SRWC) with superior genotypes are promising in central and south Florida due to their fast growth, freeze resilience, coppicing ability, and site tolerance. Four *Eucalyptus grandis* cultivars, E.nergy™ G1, G2, G3, and/or G5, were established in 2009 at varying planting densities on a reclaimed clay settling area (CSA) in phosphate mined land in central Florida and a bedded former citrus site in southern Florida. Planting densities were 1025, 2050, and 3416 trees/acre on the CSA, and 581, 869, 1162, 1452, and 1742 trees/acre on the citrus site. Modified land expectation values (LEV) for coppicing species are reported for G2, G3, and/or G5 SRWCs on CSAs and citrus land. Optimal coppice stage and cycle lengths to the nearest 1/10th year were estimated for each cultivar × spacing × land scenario, assuming a range of coppice yields, cultural treatments (weed control and fertilization), plantation establishment and maintenance costs, stumpage prices, and real discount rates of 6, 8, and 10 percent. For example, at a 10 percent discount rate, stumpage price of $14 green/ton, costs of $250, 50, 974, 55, 90, and 10/acre for land preparation, bedding, planting, pre– and post–establishment weeding, fertilization, and annual management, respectively, and expected coppice yields, the LEV of CSAs under G3 at 1025 trees/acre was $561/acre or an equal annual equivalent of ~$56/acre/year. Currently, *Eucalyptus* is primarily harvested for landscape mulch, but markets are likely to expand into bioenergy and pulpwood applications.

INTRODUCTION

SRWC production is a potential land-use alternative for undeveloped CSAs and citrus lands affected by citrus greening in central and southern Florida. CSAs and overburden sites in central Florida are a potential SRWC land base of ~200,000 acres (Rockwood and others 2006), and citrus greening has been confirmed in all citrus-growing counties in Florida. *Eucalyptus* species are ideal SRWCs because of their fast growth, site tolerance, coppicing ability, and uses ranging from landscape mulch to biofuel production (Rockwood and others 2008). Furthermore, high-density eucalypt plantations can provide ecosystem services on CSAs by shortening the dewatering time, increasing soil organic matter, and excluding invasive, non-native vegetation, such as cogongrass (*Imperata cylindrica*), and facilitating the reintroduction of native understory vegetation (Tamang and others 2005). *E. grandis* is now grown commercially in southern Florida for mulchwood and can be deployed in central Florida if freeze resilient stock is used. Demand for *E. grandis* mulchwood is likely to increase if cypress availability decreases (Rockwood 2012). Though not a native species, *E. grandis* is non–invasive in Florida and has been planted commercially in south central Florida since the 1960s without spreading (Rockwood 1996).

Much SRWC emphasis in Florida has been on *Eucalyptus* tree improvement for adaptability to damaging freezes, infertile soils, and pest resistance [notably the blue gum chalcid (*Leptocybe invasa*)]. Advanced-generation breeding and the ongoing development of seedling and clonal seed orchards can increase genetic gains in these traits, but significant improvements are possible with clonal forestry. Produced through four generations of *E. grandis* genetic improvement for Florida conditions, including three "1-in-100 year" freezes, cultivars E.nergy™ G1, G2, G3, G4, and G5 were released by the University of Florida in 2009 for commercial planting (Rockwood 2012). Selected based on 18 tests throughout Florida on widely representative site/soil types including flatwoods spodosols, sandhills, dredged bay soil, muck soil, and CSAs, these cultivars have fast growth, excellent stem form, tolerance to various site conditions, coppicing ability, freeze resilience, and ease of propagation compared to 4th generation *E. grandis* seedlings (Rockwood and others 2012).

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Langholtz and others (2007) reported the value of phosphate mined land in central Florida under *E. amplifolia* SRWCs, but our analysis describes 1) the field performance since 2009 of *E. grandis* cultivars G1, G2, G3, and/or G5 on CSAs and former citrus lands and 2) their associated economics under current operational costs and stumpage prices.

MATERIALS AND METHODS

Experimental Design and Inventory

Cultivars G1, G2, and G3 were planted in September 2009 on a bedded CSA near Ft. Meade, FL, in 15 rows spaced 8.5 feet apart with 126 trees/row at three spacings: 66, 40, and 20 trees at 1.5, 2.5, and 5 feet, respectively (3416, 2050, and 1025 trees/acre, respectively). Single cultivar blocks of three or six rows (including a three row buffer) were systematically assigned within spacing blocks. The interior measurement plots for each cultivar contained 99 and 60 trees within 0.029 acres under planting densities 3416 and 2050 trees/acre, respectively, and 33 trees within 0.032 acres at 1025 trees/acre. Measurements of tree height, diameter at breast height (DBH), form, and survival were recorded at ages 7, 15, 23, 33, 49, and 60 months. At 41 months in February 2013, all trees in rows 1, 2, 14, and 15 and all trees at 1.5 feet spacing in rows 3-13 were felled to 1) initiate coppicing and 2) provide 20 sample trees to determine wood properties.

On former citrus beds near Ft. Pierce, FL, cultivars G1, G2, G3, and G5 were planted in July 2009 with four replications of two to six row wide, 30 feet long single cultivar block (measurement) plots centered on 60 feet wide citrus beds with 3 feet spacing between trees in a row and 5 feet between rows, resulting in five planting densities (581, 869, 1162, 1452, and 1742 trees/acre). Half (reps 1 and 2) of the 3200 trees received 113.4 g of slow release Osmocote fertilizer near the planting hole at planting. Height and survival were recorded at 9 months, and height, DBH, form, and survival were recorded at 15, 23, 39, 52, and 64 months.

Based on a subset of tree heights, the remaining trees' heights were predicted from site, cultivar, and treatment specific height versus DBH relationships (Equation 1):

$$\log(HT) = a \times \log(DBH) + b \quad (1)$$

where: HT = total tree height, and a and b are estimated parameters. Data from cultivars destructively sampled in October 2014 were used to generate cultivar-specific coefficients for a whole stem green weight regression equation (Equation 2) (R^2 >.98):

$$GW = b_0 + b_1 D^2H \quad (2)$$

where: GW = individual tree whole stem green weight, D = diameter at breast height, H = total tree height, and b_0 and b_1 are estimated parameters. For D^2H values below the lower limits of observed D^2Hs, Equation 3 with previously defined variables and a cultivar-specific dry weight to green weight ratio (R) was used to estimate individual tree whole stem green weight.

$$GW = \frac{-0.364 + 0.0163 \times D^2H}{R} \quad (3)$$

Growth Projections

Yield data were fit to Equation 4 using nonlinear regression (Langholtz and others 2007):

$$B(t) = e^{[b + c \times \ln(t) - d \times t]} \quad (4)$$

where: B(t) = whole stem green weight (green tons/acre), t = stand age (years), and b, c, d are estimated parameters. Regression parameters were generated for each land scenario × cultivar × planting density × fertilization (citrus site) combinatin (table 1).

Economic analyses used the terminology of Smart and Burgess (2000), where stage length is defined as the time between coppice harvests and cycle length is the total number of stages or years between replanting. Improved and expected coppice factors were applied to each growth function (Langholtz and others 2007), with a maximum of five stages per cycle (the original planted stand plus four coppice stages). Improved coppice scenarios assumed coppice factors of 120, 100, 80, and 60 percent of the original growth function for stages 2, 3, 4, and 5 (first, second, third, and fourth coppice, respectively). Expected coppice scenarios assumed coppice factors of 80, 60, 40, and 20 percent of the original growth function for stages 2, 3, 4, and 5, respectively.

Financial Optimization Model

An optimization model utilized Palisade's @RISK® Optimizer solver function to estimate the optimal number of stages and stage lengths. The objective function maximized land expectation value (LEV), which is defined by Equation 5 (Langholtz and others 2007, Medema and Lyon 1985) and occurs at the financial rotation age (t^*) where the marginal revenue product of letting the stand grow an additional year is no longer greater than but equals the opportunity cost of holding the growing stock and the cost of delaying future rotations (known as the marginal input cost) (Chang, 1984):

Table 1—Yield function parameters for G Series cultivars under three planting densities on a CSA and the most productive (fertilized) cultivar × spacing treatments on a former citrus site (R^2 >.99)

Cultivar	Trees Per Acre	Estimated Parameters		
		b	*c*	*d*
		----------*CSA*----------		
G1	1025	1.14	3.82	0.68
	2050	2.39	2.76	0.67
	3416	1.76	4.37	1.08
G2	1025	1.99	3.80	0.66
	2050	2.04	2.46	0.23
	3416	2.70	3.35	0.83
G3	1025	1.21	5.44	1.03
	2050	2.41	3.62	0.62
	3416	2.10	4.57	1.07
		----------Former Citrus Site----------		
G3	581	2.14	3.30	0.54
G2	869	1.38	3.67	0.54
G2	1162	2.67	3.96	0.84
G5	1452	2.21	2.74	0.35
G2	1742	2.83	3.00	0.52

$$LEV = \frac{\sum_{s=1}^{n}\left[V(t_s)e^{\left(-r\sum_{j=1}^{s} t_j\right)} - C_s e^{\left(-r\sum_{j=1}^{s} t_{j-1}\right)}\right]}{1 - e^{\left(-r\sum_{j=1}^{n} t_j\right)}} - \left(\frac{C_a}{1 - e^{-r}}\right) - C_r \quad (5)$$

where: n = number of stages, s; V(t) = stand value at time t within stage s; r = real discount rate; C_s = costs of stage s at the start of the stage; C_a = annual cost; C_r = year zero start–up establishment cost. LEV is the maximum amount that an investor could afford to bid for bare forestland and earn the required interest rate (Davis and others 2001). Harvest schedules are reported to the nearest 1/10th year due to optimum stage lengths that are less than six years; therefore, an annual interest rate was converted to an effective periodic rate.

Because G1 is no longer commercially viable due to its susceptibility to blue gum chalcid, LEVs at 6, 8, and 10 percent real discount rates (net of inflation) were calculated only for G2, G3, and G5.

Summary of Model Inputs

The economic models assumed various timings and costs of management activities and a range of reported mulchwood stumpage prices to assess the sensitivity of LEV to changes in market conditions (table 2). Due to uncertainty associated with *Eucalyptus* markets in central and southern Florida, the operational costs are higher compared to conventional forest plantations in the South. Weed control was assumed to take place at the beginning of each stage to promote rapid growth rates and improve yields. Annual costs were $10 and 35/acre for the CSA and citrus site (includes a $25/acre drainage cost), respectively.

RESULTS AND DISCUSSION

Productivity Estimates

CSA—MAI_{max} for G Series cultivars ranged from 9.2 (G1 at 3416 trees/acre) to 34.9 (G3 at 2050 trees/acre) green tons/acre/year. G3 had the highest yields at all three planting densities (fig. 1), producing 25.9 (4.3 years), 34.9 (4.2 years), and 17.3 (3.4 years) green tons/acre/year at 1025, 2050, and 3416 trees/acre, respectively.

Table 2—Summary of model inputs for CSA and former citrus sites

Activity	Timing	Values
	----------CSA----------	
Land Preparation	One-time start-up cost	$125 and 250/acre
Bedding	Beginning of each cycle	$50/acre (same)
Planting Cost	Beginning of each cycle	$0.10 and 0.25/tree
Planting Densities	N/A	1025, 2050, and 3416 trees/acre
	----------Former Citrus Site----------	
Land Preparation	One-time start-up cost	$400 and 500/acre
Chemical Site Preparation	Beginning of each cycle	$90 and 120/acre
Planting Cost	Beginning of each cycle	$0.25 and 0.40/tree
Planting Densities	N/A	581, 869, 1162, 1452, and 1742 trees/acre
	----------CSA and Former Citrus Site----------	
Fertilization	Beginning of each cycle	$55 (both) and 70 (citrus) or 90/acre (CSA)
Weed Control	Beginning of each stage	$55/acre (same)
Planting Material	Beginning of each cycle	$0.55 and 0.70/propagule
Real Discount Rates	N/A	6, 8, and 10%
Stumpage Prices	N/A	$9, 14, and 19/green ton
Coppice Yields	Duration of each stage	Expected and Improved
Number of Stages	N/A	5 Stages Maximum

G2 achieved a MAI_{max} of 25 (4.2 years), 27 (6.4 years), and 16.4 (2.8 years) green tons/acre/year at 1025, 2050, and 3416 trees/acre, respectively. The pooled average of MAI_{max} at 3416, 2050, and 1025 trees/acre were 14.3, 24.1, and 20.3 green tons/acre/year, respectively. The low productivity at 3416 trees/acre may be due to nutrient limitations (no fertilization at establishment), heavy clay compaction (no subsoiling prior to stand establishment), and stocking levels above the onset of density-dependent mortality, which corresponds to 55 percent of the maximum stand density index of 490 trees/acre for *Eucalyptus* species (Reineke 1933).

Former citrus site—Through 64 months, planting density, cultivar, and fertilization variously influenced productivity, which was as high as 33.6 green tons/acre/year with fertilized G2 in a 3.8-year rotation at 1742 trees/acre (fig. 2). Unfertilized, chalcid susceptible G1 was the least productive with a MAI_{max} of 9.2 green tons/acre/year at 869 trees/acre. Fertilized G3 in a 4.3-year rotation was the most productive at 581 trees/acre with a MAI_{max} of 24.2 green tons/acre/year. Fertilized G2 had the highest yields at 869, 1162, and 1742 trees/acre. The MAI_{max} for fertilized G2 at 869 and 1162 trees/acre were 20.1 and 31.3 green tons/acre/year in 5 and 3.5 years, respectively. Fertilized G5 was superior at 1452 trees/acre with a MAI_{max} of 26.1 green tons/acre/year in a 5-year rotation. The pooled averages of MAI_{max} were 12.8, 14.0, 22.4, 22.5, and 30.9 green tons/acre/year at 581, 869, 1162, 1452, and 1742 trees/acre, respectively. The double-row configuration was established on the apex of a citrus bed, which could explain its low productivity as there may be less plant-available water and residual fertilizer compared to planting rows established on a bed shoulder or back slope (former citrus row locations).

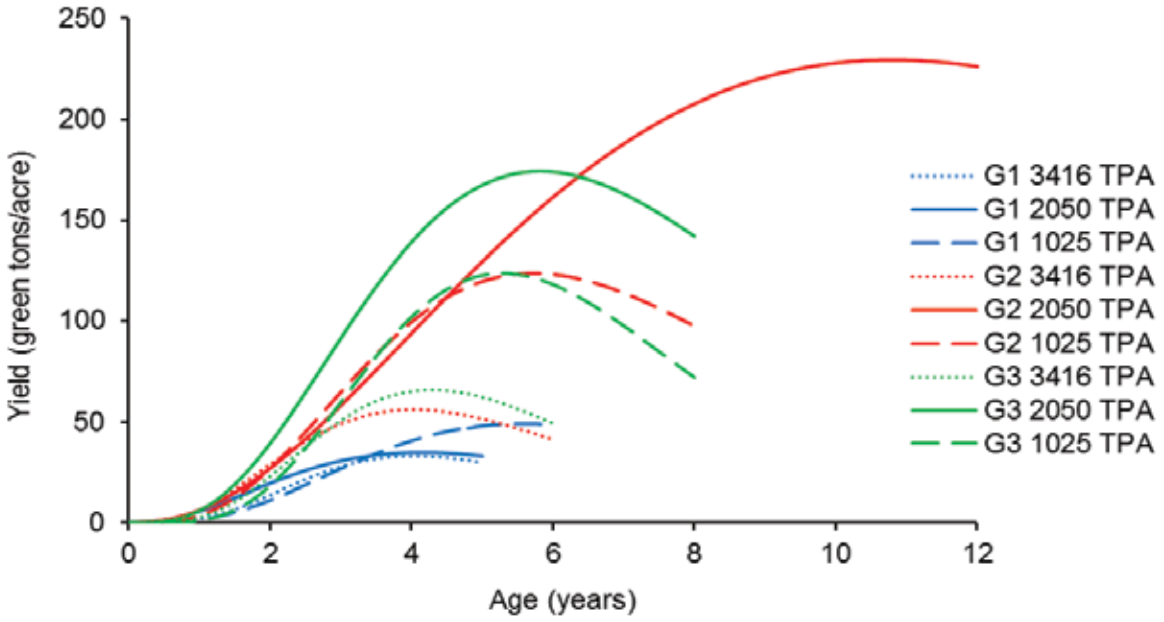


Figure 1—Outside bark yields (green tons/acre) for each cultivar × spacing treatment on a phosphate mined CSA in central Florida.

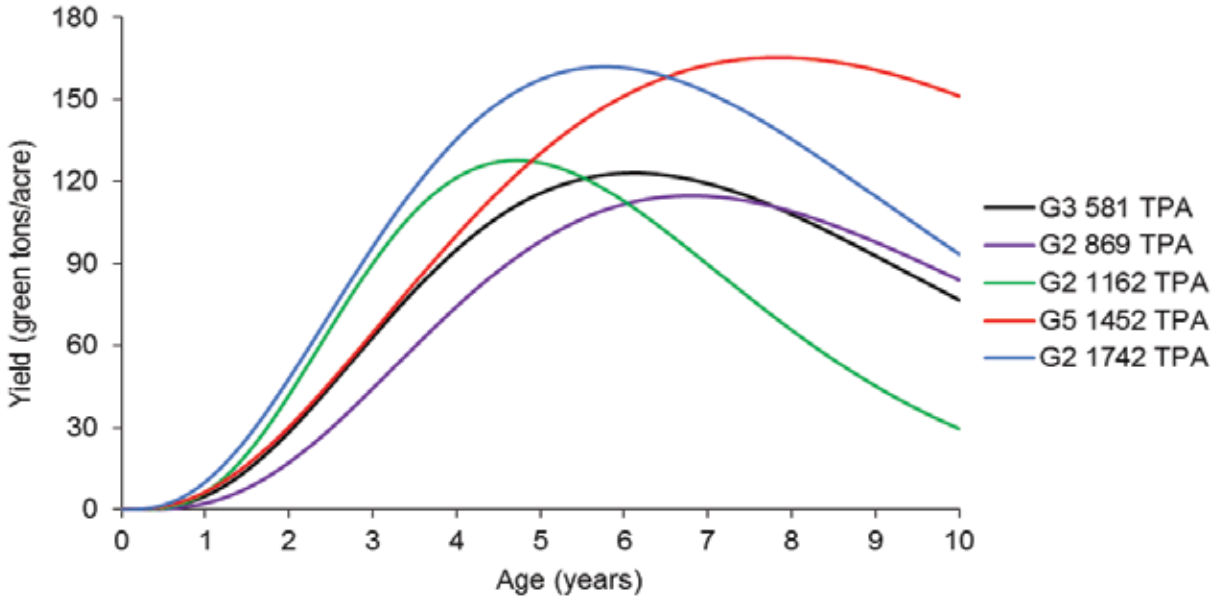


Figure 2—Outside bark yields (green tons/acre) for fertilized cultivar × spacing treatments on a former citrus site in southern Florida.

Financial Results

CSA—Break-even prices were calculated for each genotype at 3416 trees/acre due to low LEV results and the tendency to delay replanting by maximizing the length of the last stage. Since break-even prices exceeded $24/green ton under expected coppice yields and were significantly greater than current *Eucalyptus* mulchwood stumpage prices due to high establishment costs and low productivity, there were no feasible harvest schedules at 3416 trees/acre. For cultivars G2 and G3 at 1025 and 2050 trees/acre, and under all combinations of model inputs, LEVs ranged from -$1168 to 7534/acre. G3 was more profitable than G2 at all three planting densities. Assuming expected coppice yields, G3 generated higher LEVs at 2050 trees/acre compared to 1025 trees/acre under the following inputs: 1) low management costs, $14/green ton, and real discount rates of 6, 8, and 10 percent; 2) low and high management costs, $19/green ton, and real discount rates of 6 and 8 percent; and 3) low management costs, $19/green ton, and 10 percent real discount rate. Table 3 shows the financial results for cultivar G3 at 1025 trees/acre under the least profitable assumptions (high management costs and expected coppice yields) at all three stumpage prices and real discount rates. In addition, assuming improved coppice yields, G3 produced higher LEVs at 2050 trees/acre compared to 1025 trees/acre under the following assumptions: 1) low management costs, $9/green ton, and 6 and 8 percent real discount rates; 2) low and high management costs, $14/green ton, and 6 and 8 percent real discount rates; 3) low management costs, $14/green ton, and 10 percent real discount rate; and 4) low and high management costs, $19/green ton, and 6, 8, and 10 percent real discount rates.

Former citrus site—The most profitable results at each planting density and for all combinations of management costs, stumpage prices, real discount rates, and coppice yields had LEVs ranging from -$1471 to 6193/acre. In general, fertilized stands of G3 at 581 trees/acre and G2 at 1162 trees/acre were the most profitable cultivar × spacing treatments. Assuming high management costs, G3 at 581 trees/acre was the most profitable regime at $9 to 14/green ton, although G2 at 1162 trees/acre generated higher LEVs at $14/green ton and improved coppice yields. At $19/green ton and high management costs, G2 at 1162 trees/acre was the most profitable. However, G3 at 581 trees/acre outperformed G2 at 1162 trees/acre assuming a 10percent real discount rate, $19/green ton, high management costs, and expected coppice yields (table 4). Under low management costs, G2 at 1162 trees/acre was the most profitable for most discount rate, stumpage price, and coppice yield scenarios, but G3 at 581 trees/acre generated higher LEVs under the following assumptions: 1) $9/green ton and expected coppice yields at 6, 8, and 10 percent real discount rates; 2) $9/green ton, improved coppice yields, and 10 percent real discount rate; and 3) $14/green ton, expected coppice yields, and 10 percent real discount rate. Overall, G3 at 581 trees/acre produced higher LEVs under high management costs and low stumpage prices, while G2 at 1162 trees/acre obtained higher LEVs with low management costs and/or high stumpage prices.

The "rank change" in productivity and profitability between cultivars G2 and G3 across planting sites may suggest a genotype × environment interaction (G×E), which could increase as fewer clones are deployed over widely contrasting soil types. With intensive silviculture, there is a greater likelihood of significant G×E interactions with clonal or full-sib family forestry due to less buffering to environmental conditions compared to genetically diverse open-pollinated families, and G×E is caused by both additive and non-additive genetic effects (McKeand and others 2006).

Sensitivity Analysis—In general, increasing the discount rate slightly decreased optimum stage lengths. Additional growth stages were observed at higher discount rates and lower stumpage prices to delay the cost of replanting. This relationship was consistent with results from Smart and Burgess (2000) and Langholtz and others (2007), who observed for coppicing species that increasing the discount rate does not shorten the cycle (rotation) length as it would with non-coppicing species. This is because the opportunity cost of delaying future rotations is proportionally greater than the opportunity cost of holding the growing stock for SRWC systems compared to non-coppicing species. Improved coppice yields may occur if weeding and/or fertilization is applied at the beginning of each stage, which can significantly increase LEV, lengthen coppice cycles, and favor higher planting densities. Furthermore, assuming expected coppice yields, increasing the planting density of G2 to 1162 trees/acre on former citrus lands and G3 to 2050 trees/acre on CSAs were optimal under low management costs and moderate to high ($14-19/green ton) stumpage prices.

CONCLUSION

Under current market conditions in central and southern Florida, which includes high operational costs and low to moderate mulchwood stumpage prices ($9-14/green ton), *Eucalyptus* SRWCs could generate LEVs that are comparable to, or greater than, Florida agricultural land prices. G Series cultivars can be profitable at real rates of 10 percent on CSAs and former citrus lands, even at moderate stumpage prices, high operational costs, and expected coppice yields. Continued progress in *Eucalyptus* genetic improvement for Florida conditions is essential to increasing growth, freeze resilience, and chalcid resistance, and to meet feedstock demands when bioenergy markets develop.

Table 3—Financial results for cultivar G3 at 1025 trees/acre on a CSA, assuming expected coppice yields, high management costs, three stumpage prices, and three real discount rates

Stumpage Price ($/green ton)	Real Discount Rate	LEV ($/acre)	Stage Lengths (years)
9	6%	301	4.7, 4.7, 4.7, 4.6
	8%	-69	4.6, 4.6, 4.7, 4.7, 4.4
	10%	-292	4.5, 4.6, 4.6, 4.7, 4.7
14	6%	1824	4.6, 4.5, 4.4
	8%	1027	4.5, 4.5, 4.5
	10%	561	4.5, 4.5, 4.5, 4.3
19	6%	3443	4.5, 4.5, 4.3
	8%	2215	4.5, 4.4, 4.3
	10%	1472	4.4, 4.4, 4.3

Table 4—Financial results and optimum stage lengths for the most profitable (fertilized) cultivar × spacing treatments on a former citrus site, assuming expected coppice yields, high management costs, three stumpage prices, and three real discount rates

Stumpage Price ($/green ton)	Real Discount Rate	Cultivar	Trees Per Acre	LEV ($/acre)	Stage Lengths (years)
9	6%			-146	4.9, 4.9, 4.8, 4.4
	8%	G3	581	-422	4.7, 4.8, 4.8, 4.6
	10%			-596	4.6, 4.7, 4.7, 4.7
14	6%			1309	4.7, 4.6, 4.3
	8%	G3	581	627	4.6, 4.5, 4.4
	10%			214	4.5, 4.5, 4.4
19	6%	G2	1162	3043	3.9, 3.8, 3.6
	8%	G2	1162	1800	3.8, 3.8, 3.7
	10%	G3	581	1071	4.4, 4.4, 4.1

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LITERATURE CITED

Chang, S.J. 1984. Determination of the optimal rotation age: a theoretical analysis. Forest Ecology and Management. 8(2): 137-147.

Davis, L.S.; Johnson, K.N.; Bettinger, P.; Howard, T.E. 2001. Forest management to sustain ecological, economic, and social values. 4th ed. Long Grove, IL: Waveland Press, Inc. 816 p.

Langholtz, M.; Carter, D.R.; Rockwood, D.L.; Alavalapati, J.R.R. 2007. The economic feasibility of reclaiming phosphate mined lands with short-rotation woody crops in Florida. Journal of Forest Economics. 12: 237-249.

McKeand, S.E.; Jokela, E.J.; Huber, D.A. [and others]. 2006. Performance of improved genotypes of loblolly pine across different soils, climates, and silvicultural inputs. Forest Ecology and Management. 227: 178-184.

Medema, E.L.; Lyon, G.W. 1985. The determination of financial rotation ages for coppicing tree species. Forest Science. 31(2): 398-404.

Reineke, L.H. 1933. Perfecting a stand density index for even–aged forests. Journal of Agricultural Research. 46: 627-638.

Rockwood, D.L. 1996. Using fast-growing hardwoods in Florida. Fact Sheet EES-328. Florida Cooperative Extension Service, Institute of Food and Agricultural Sciences, University of Florida. 8 p. http://ufdcimages.uflib.ufl.edu/IR/00/00/29/07/00001/MG34900.PDF. [Date accessed: July 28, 2015].

Rockwood, D.L. 2012. History and status of *Eucalyptus* improvement in Florida. International Journal of Forestry Research. Volume 2012, Article ID 607879. http://dx.doi.org/10.1155/2012/607879. 10 p. [Date accessed: July 28, 2015].

Rockwood, D.L.; Carter, D.R.; Langholtz, M.H.; Stricker, J.A. 2006. *Eucalyptus* and *Populus* short rotation woody crops for phosphate mined lands in Florida USA. Biomass and Bioenergy. 30: 728-734.

Rockwood, D.L.; Rudie, A.W.; Ralph, S.A. [and others]. 2008. Energy product options for *Eucalyptus* species grown as short rotation woody crops. International Journal of Molecular Sciences. 9: 1361-1378.

Rockwood, D.L.; Tamang, B.; Kirst, M.; Zhu, J.Y. 2012. Field performance and bioenergy characteristics of four commercial *Eucalyptus grandis* cultivars in Florida [Abstract]. In: Butnor, J.R., ed. 2012. Proceedings of the 16th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 267-268.

Smart, J.; Burgess, J. 2000. An environmental economic analysis of willow SRC production. Journal of Forest Economics. 6(3): 193-266.

Tamang, B.; Rockwood, D.L.; Langholtz, M. [and others]. 2005. Vegetation and soil quality changes associated with reclaiming phosphate-mine clay settling areas with fast growing trees. In: Proceedings of the 32nd annual conference on ecosystem restoration and creation, October 27-28, 2005, Plant City, FL. Hillsboro Community College, FL: Curran Associates, Inc.: 126-140.

FEASIBILITY OF UTILIZING SMALL DIAMETER SOUTHERN PINE FOR BIOMASS IN THE VIRGINIA COASTAL PLAIN

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Abstract—New or retrofitted wood-fired energy plants have increased demand for woody biomass in the state of Virginia. Loblolly pine (*Pinus taeda*) commonly serves as a feedstock for these energy plants. Pulpwood conventionally requires a minimum diameter of 4 inches diameter at breast height (DBH) for merchantability, whereas a minimum merchantable diameter for biomass is currently undefined. A harvesting case study was completed during the first thinning of a 15 year-old loblolly pine stand in the Virginia Coastal Plain to determine production rates and costs while harvesting densely stocked small diameter loblolly pine stems (< 4 inches DBH). Pre-harvest stand attributes included a quadratic mean diameter (QMD) of 4.2 inch DBH, an average density of 1,377 stems/acre, and an average volume of 73.7 green tons/acre. Post-harvest 34.0 green tons/acre remained. Harvesting equipment included three Tigercat 718 feller-bunchers, two Caterpillar 535C skidders, and one Peterson 4300 chipper. Elemental time studies conducted on the feller-bunchers, skidders, and chipper determined individual machine productivity rates of 30.8, 23.4, and 83.7 green tons/productive machine hour (PMH), respectively. Use of the Auburn Harvesting Analyzer determined an on-board truck cost of $16.52/green ton and a total cut-and-haul cost of $23.46/green ton. Regional average prices for in-woods chips indicate the harvest was not economically feasible.

INTRODUCTION

Woody biomass energy use has seen a large increase in the southeast US throughout the last decade. It is estimated that wood-consuming bioenergy projects may increase total wood use to 45 million green tons per year in the US south by the year 2023 (Forisk 2015). In Virginia, several wood-fired energy plants have been established and the state ranks 5th in US biomass energy plant total nameplate capacity (Biomass Magazine 2015). Small-diameter stems (i.e., < 4 inches DBH) from pre-commercial thinning (PCT) may be a potential source of feedstock for energy facilities. PCT is a practice used in the southeastern US to mitigate southern pine beetle risk and increase residual tree diameter growth (Burkhart and others 1986, Nowak and others 2008). PCT residues are normally left on-site and do not produce forest products (Perlack and others 2011).

Since PCT treatments normally incur an added management cost to landowners, some states in the southeast provide cost-share programs available to private landowners to help combat the expense of PCT. In Virginia, the Virginia Department of Forestry offers the Pine Bark Beetle Prevention Program, which provides private landowners a 60 percent cost-share for PCT treatment (Watson and others 2013). Previous studies have suggested the potential use of small-diameter PCT residues for woody biomass energy production (Perlack and others 2011, Staudhammer and others 2011), which, if utilized, could potentially reduce the added management cost of PCT.

A variety of harvesting case studies analyzing harvesting productivity and costs of utilizing small-diameter stems have been completed (Bolding and Lanford 2005, Mitchell and Gallagher 2007, Pan and others 2008). However, many of these studies have focused on stands older than 20-years, which would not typically be considered for a “normal” PCT (less than 15 years old). Additionally, some of these studies have also focused on integrated harvest scenarios in which a variety of forest products are produced, whereas a PCT biomass harvest would likely be a biomass-only harvest.

PURPOSE

The purpose of our study was to perform a biomass harvesting case study on a site with “near-PCT” conditions to determine machine productivity and operating costs while harvesting small-diameter stems in a young pine stand. The main goals of our study were to: 1) determine stand density and volume attributes, 2) collect machine productivity information, and 3) calculate harvesting costs and compare to regional delivered prices for biomass.

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METHODS

The site selected for this case study was located in Greensville County, VA, within the southern coastal plain region and near the city of Emporia. The stand comprised 87 acres of 15 year-old planted, non-bedded loblolly pine. The nearest biomass energy plant to which the majority of the chips were taken was roughly 30 miles away from the site.

Pre-harvest and post-harvest inventories of the stand were completed using: 1/10th fixed-acre plots for stems ≥ 2 inches DBH and 1/100th fixed-acre for stems < 2 inches DBH. Inventory data was used to determine initial, residual, and removed stand volumes and densities.

Harvesting equipment used for the operation included three Tigercat 718 feller-bunchers, two Caterpillar 535C skidders, and one Peterson 4300 mobile chipper. Each machine had separate operators. To evaluate productivity, activity and work sampling time studies were used to determine the amount of green tons (gt) produced per productive machine hour (PMH) for each type of the three harvesting components. Cycles times for the feller-bunchers, skidders, and chipper were observed. Stem counts were also observed in each of the feller-buncher and skidder cycles. Based on the inventory data, the average volume of the removed stems (gt/stem) was calculated and used in productivity calculations.

Each cycle time interval was defined by the time from when the previously observed cycle time ended and to the time when the current observed cycle ended. For example, feller-buncher cycle times would start when the previously observed bunch of trees was dropped, and end when the currently observed bunch of trees was dropped. The same principle applied to each skidder cycle, using the time from skid turn to skid turn, and the chipping cycle, using the time between the filling of each chip van. After productivity rates were calculated, the machine rate method (Miyata 1980) was used with the Auburn Harvest Analyzer (AHA) (Tufts and others 1985) to estimate harvesting costs. To assess the effect of machine configuration changes on total harvesting costs, sensitivity analyses were conducted in the AHA by adjusting the number of machines in each harvesting component.

RESULTS

Inventory results show an initial, pre-harvest stand density of 1,377 stems/acre with a residual, post-harvest density of 390 stems/acre (table 1). Initial stem diameter averaged 3.8 inches DBH and the QMD was 4.3 inches DBH. Following harvest, stem diameters averaged 5.1 inches DBH and QMD was 5.3 inches DBH. The initial standing volume was 73.7 gt/acre with a residual 34.0 gt/acre, indicating a removed volume of 39.7 gt/acre.

Feller-buncher cycles (*n* = 398) averaged 1.5 minutes and 19 stems per cycle (table 2). Overall individual feller-buncher productivity was 30.8 gt/PMH. Cycle times varied among the feller-buncher operators; on average, Operator 1 had the longest cycle time followed by Operator 2 and Operator 3. These longer cycle times translated to a higher productive rate for Operator 3 (35.7 gt/PMH) followed by Operator 2 (32.2 gt/PMH) and Operator 1 (29.9 gt/PMH), which was expected considering Operator 3 had more experience followed by Operators 2 and 1.

Skidder cycles (*n* = 145) averaged 10.04 minutes and 97 stems per cycle (table 3). Overall productivity averaged 23.4 gt/PMH. Operator 1 had a lower average cycle time than Operator 2, equating to a higher productive rate for Operator 1 (25.6 gt/PMH) than Operator 2 (21.0 gt/PMH). However, Operator 2 had an average skid distance nearly 600 feet greater than Operator 1, contributing to their lower productive rate. Chipper cycles (*n* = 48) averaged 19.1 minutes per cycle. After chipping, loaded chip vans averaged 26.7 gt/load, equating to a productive rate of 83.7 gt/PMH.

Individual harvesting component costs as a percentage of the total cut-and-haul cost were 33 percent for hauling ($6.94/gt), 24 percent for felling ($5.12/gt), 22 percent for chipping ($4.67/gt), and 21 percent for skidding ($4.34/gt). Hauling represented the largest proportion of total harvesting costs, which was consistent with hauling costs observed in previous

Table 1—Pre- and post-harvest inventory results (stems ≥ 2 inches DBH)

	Pre	Post
Loblolly pine density *stems/acre*	723	267
All stem density *stems/acre*	1,377	390
Mean loblolly pine DBH *inches*	5.2	5.8
Mean all stem DBH *inches*	3.8	5.1
All stem QMD *inches*	4.3	5.3
Loblolly pine volume *green tons/acre*	57.9	26.2
All stem volume *green tons/acre*	73.9	34.0

Table 2—Feller-buncher time study summary statistics

		Time per bunch *minutes*		Stems per bunch		Productivity *gt/PMH*
	n	Mean	SE	Mean	SE	
Overall	398	1.5	0.03	19	0.4	30.8
Operator 1	170	1.8	0.04	22	0.6	29.9
Operator 2	129	1.4	0.06	19	0.6	32.2
Operator 3	99	1.1	0.04	16	0.8	35.7

Table 3—Skidder time study summary statistics

		Time per skid turn *minutes*		Stems per skid turn		Skid distance *feet*		Productivity *gt/PMH*
	n	Mean	SE	Mean	SE	Mean	SE	
Overall	145	10.0	0.5	97	2.8	1,427	41.7	23.5
Operator 1	83	9.0	0.5	95	4.2	1,186	33.4	25.7
Operator 2	62	11.4	0.9	99	3.2	1,751	67.7	21.1

studies (Bolding and Lanford 2005, Mitchell and Gallagher 2007, Pan and others 2008). However, felling costs comprised a larger proportion of total costs than similar studies (Pan and others 2008, Bolding and others 2009), which was likely due to the high harvesting costs of using additional feller-bunchers in this study compared to other studies that used fewer feller-bunchers.

The calculated on-board truck cost, which includes felling, skidding, and chipping, was $16.52/gt (table 4). Including hauling, the total cut-and-haul cost was $23.46/gt. The regional average price for whole tree in-woods pine chips at the time of the case study was $17.35/gt (Timber Mart South 2014), lower than the calculated cut-and-haul cost. Sensitivity analysis conducted in the AHA found that the overall system was "balanced" by reducing the number of feller-bunchers from three to two, reducing the total cut-and-haul cost to $22.28/gt.

DISCUSSION

Individual machine productive rates from this study were compared to similar studies (table 5). The average productive rates of the feller-bunchers and skidders in this study were in-between the rates observed in similar studies (Pan and others 2008, Bolding and others 2009). However, the productive rate of the chipper was much higher than rates observed in other studies (Bolding and Lanford 2005, Mitchell and Gallagher 2007, Bolding and others 2009). This higher productive rate can likely be attributed to the relatively large size of the chipper used in our study in comparison to these other studies. Additionally, considering some of these other studies merchandised stems while our study chipped all stems without merchandising, higher productive rates of the chipper could further be expected as a result of increased chipper use.

Some insight regarding the utilization of PCT stands for woody biomass energy is gained by this study. Although the total-cut-and haul costs of this study are higher than the delivered price for biomass, landowners may still have an opportunity to reduce the cost of conventional PCT treatment by harvesting PCT biomass instead. If the payment to a logger for harvesting PCT biomass is less than the cost of conventional treatment, biomass harvesting would be financially preferable to the landowner.

CONCLUSION

This study analyzed the productivity and costs of a biomass-only operation harvesting small-diameter stems in a 15 year-old loblolly pine stand in the coastal

Table 4—Harvesting costs and biomass prices

	Per green ton
On-board truck (felling, skidding, and chipping)	$16.52
Total cut-and-haul (felling, skidding, chipping, and hauling)	$23.46
Delivered chip price (Timber Mart South 2014 3rd Quarter)	$17.35

Table 5—Machine productivity (gt/PMH) comparison

	Our study	Bolding and others 2009	Pan and others 2008	Mitchell and Gallagher 2007	Bolding and Lanford 2005
			gt/PMH		
Feller-buncher	32.6	38.1	23.8	N/A	N/A
Skidder	23.5	16.2	34.8	N/A	N/A
Chipper	83.7	24.7	N/A	66.7	20.2

plain of Virginia. Individual productive rates for the feller-bunchers, skidders, and chipper were 30.8, 23.4, and 83.7 gt/PMH, respectively. A total cut-and-haul cost of $23.46/gt was determined, and is higher than the regional average delivered price for in-woods chips.

Going forth, production studies are needed in more traditional PCT stands to gain better insight on the harvesting productivities and costs associated with utilizing PCT biomass since the stand used for this case study was at the upper age limit for typical PCT stands. Furthermore, alternative harvesting equipment configurations should be analyzed to investigate potential harvesting cost reductions. Lastly, since this study does not consider effects of stem removal on nutrient availability, soil disturbance, and other site characteristics, further work is needed to help analyze these effects.

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Virginia Department of Forestry

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LITERATURE CITED

Biomass Magazine. 2015. 2015 U.S. Biomass Plants. Available at: http://biomassmagazine.com/plants/listplants/biomass/US/ [Last accessed: September 17, 2015].

Bolding, M.C.; Lanford, B.L. 2005. Wildfire fuel harvesting and resultant biomass utilization using a cut-to-length/small chipper system. Forest Products Journal. 55(12): 181-189.

Bolding, M.C.; Kellogg, L.D.; Davis, C.T. 2009. Productivity and costs of an integrated mechanical forest fuel reduction operation in southwest Oregon. Forest Products Journal. 59(3): 35-46.

Burkhart, H.E.; Haney, H.L.; Newberry, J.D. [and others]. 1986. Evaluation of thinning for reduction of losses from southern pine beetle attack in loblolly pine stands. Southern Journal of Applied Forestry. 10(2): 105-108.

Forisk. 2015. Available at: http://forisk.com/resources/resources-from-forisk-wood-bioenergy-us-free-summary/ [Last accessed: February 2, 2015].

Mitchell, D.; Gallagher, T. 2007. Chipping whole trees for fuel chips: a production study. Southern Journal of Applied Forestry. 31(4): 176-180.

Miyata, E.S. 1980. Determining fixed and variable costs of logging equipment. Gen. Tech. Rep. NC-55. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experimental Station. 20 p.

Nowak, J.; Asaro, C.; Klepzig, K.; Billings, R. 2008. The southern pine beetle prevention initiative: working for healthier forests. Journal of Forestry. 106(5): 261-267.

Pan, F.; Han, H.S.; Johnson, L.R.; Elliot, W.J. 2008. Production and cost of harvesting, processing, and transporting small-diameter (≤ 5 in) trees for energy. Forest Products Journal. 58(5): 47-53.

Perlack, R.D.; Stokes, B.J., lead authors. 2011. US billion-ton update: biomass supply for a bioenergy and bioproducts industry. Oak Ridge, TN: United States Department of Energy, Oak Ridge National Laboratory. 235 p.

Staudhammer, C.; Hermansen, L. A.; Carter, D.; Macie, E.A. 2011. Wood to energy: using southern interface fuels for bioenergy. Gen. Tech. Rep. SRS-132. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 100 p.

Timber Mart South. 2014. Norris Foundation, University of Georgia, Athens, GA. 4th quarter. 44 p.

Tufts, R.A.; Lanford, B.L.; Greene, W.D. Burrows, J.A. 1985. Auburn harvesting analyzer. Compiler. 3(2): 14-15.

Watson, A.C.; Sullivan, J.; Amacher, G.S.; Asaro, C. 2013. Cost sharing for pre-commercial thinning in southern pine plantations: willingness to participate in Virginia's pine bark beetle prevention program. Forest Policy and Economics. 34: 65-72.

STRIP THINNING YOUNG HARDWOOD FORESTS: MULTI-FUNCTIONAL MANAGEMENT FOR WOOD, WILDLIFE, AND BIOENERGY

Jamie Schuler and Ashlee Martin[1]

Abstract—Upland hardwood forests dominate the Appalachian landscape. However, early successional forests are limited. In WV and PA, for example, only 8 percent of the timberland is classified as seedling and sapling-sized. Typically no management occurs in these forests due to the high cost of treatment and the lack of marketable products. If bioenergy markets come to fruition, these young forests can be managed in ways that promote improved forest growth, increased wildlife habitat, and biomass feedstocks. We will demonstrate how strip thinning in young stands can simultaneously provide (1) long-term wood products (sawtimber), (2) the maintenance of early successional habitat that many wildlife species require, and (3) a woody feedstock that can be repeatedly harvested and requires no establishment costs.

INTRODUCTION

Forests continue to dominate the landscape throughout the eastern U.S. However, they are increasingly utilized for multiple uses and functions. While forests are commonly managed for wood and wildlife, their role in providing biomass for energy is expected to increase. The goal of this study was to examine the opportunities to simultaneously manage forests for wood, wildlife and bioenergy.

Deciduous hardwoods are a predominant cover type across the eastern U.S. landscape. In the central Appalachian region, oaks (*Quercus*), maples (*Acer*), hickories (*Carya*), yellow-poplar (*Liriodendron tulipifera*), and birch (*Betula*) are represented in most forested stands. The age class distributions across the region show most of these forests are heavily skewed to older stands. For example, Pennsylvania and West Virginia each have less than 8 percent of their forested landscape as younger forests (i.e., <20 years old, fig. 1) (FIA 2007). As a consequence, early successional habitat and its associated wildlife species are declining.

Recently harvested stands can provide for early successional habitat, providing a mixture of grasses, herbaceous plants, shrubs and tree species (Greenberg and others 2011). However, after a decade or two, trees dominate and shade out most other vegetation. As these forests develop, the high stem densities that occur during this period also constrain tree growth, thereby prolonging suboptimal growth and the time until commercial products are available. While studies have documented positive growth responses resulting from treatments that reduced stem density at young ages (Trimble 1973, Schuler and Robison 2006, Robison and others 2004, Pham 1985; Smith and Lamson 1983), the operational challenge has been that these activities are pre-commercial which produce no immediate return at a considerable cost.

When implemented, pre-commercial thinning treatments in hardwood stands are generally applied as individual tree release treatments, that select the best 50-75 stems/acre and discriminate against poorly formed stems and less desirable species (Miller and others 2007). The drawback of these “crop tree” treatments is that they are expensive, time consuming and labor intensive. Additionally, the crop tree approach does not allow for the collection of harvested material. As an alternative, strip-thinning could be applied in a cost-effective and efficient manner in young stands; however, there is little a priori knowledge to predict the quality and growth rate of stands treated in this fashion. The few published studies generally indicate accelerated stand development and improved growth following strip thinning in young stands, (e.g., Bella and DeFranceschi 1982, McCreary and Perry 1983, Cain 1993, Schuler and Robison 2009), but they offer very little insight as to the magnitude of the response in young Appalachian hardwoods stands.

The feasibility of strip thinning relies on the availability of equipment and markets for small diameter stems. Recently, many prototypes and some commercially available harvesting machines have been marketed for operating in forest conditions characterized as having high-density, small diameter stems (Dykstra 2010, Roise and others 2009). Certainly, without a suitable

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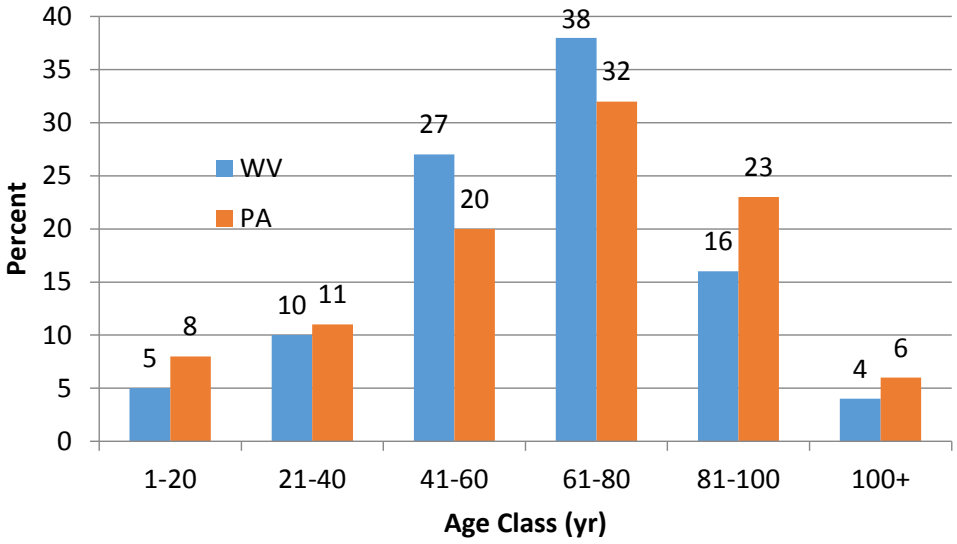


Figure 1—Age class distribution among forest stands in PA and WV (FIA 2007).

market for small diameter biomass, these strip thinning treatments will result in a significant expense to the landowner. The largest potential use for this biomass is for energy production. Interest in utilizing trees as a woody feedstock for bioenergy development remains high as the U.S. attempts to reduce fuel imports and reduce foreign reliance for its energy needs. Utilizing this technology, harvesting previously non-merchantable trees from natural stands for use as a bioenergy feedstock would alleviate the marketing constraints and increase the number of silvicultural management options for stands with high numbers of small diameter stems, poorly formed or defective trees, and non-merchantable species.

Once considered a drawback to managing young stands (Trimble 1973), the significant sprouting potential of Appalachian hardwoods can offer a considerable opportunity when managing for woody feedstocks and wildlife. Coppicing is a management strategy that relies on the development of new stems from stump sprouts or root suckers. Coppicing young stands allows for the repeated harvests on short rotations (e.g., 3-5 years). This sequence of activities can be repeated for a long period of time. The result is a sustainable system of forest management which is capable of producing a large amount of biomass. The coppice system has been widely promoted in the parts of the northeastern US for growing short rotation woody crops biomass using shrub willows grown under a 3 year harvest cycle (Heller and others 2003). Historically, this method has been in practice throughout Europe for centuries, where woodlands are repeatedly harvested for fuelwood.

A variant of the coppice system includes *standards*, which allows for the retention a few non-coppiced stems for other values such as timber production, mass production, and aesthetics. This modified coppice system that includes standards can achieve a suite of desired commodity and non-commodity objectives and diversify the forest landscape in the Appalachians by providing opportunities to manage young stands, generate additional income through woody feedstock production, and enhance wildlife habitat, all while maintaining the traditional sawtimber forest products for which the Appalachian hardwood forests are well known.

The coppice with standards method is untested in natural stands under the scenario envisioned here. This study will address the feasibility of such systems for improving growth rates of trees in residual strips, as well as providing yield estimates for each coppice cycle. Short rotation coppice systems can persist 5-7 cycles until significant reductions in yield occur. In the proposed system, the width of the coppiced strips will affect the long-term production, since canopy encroachment from the uncut residual strips will shade the coppice regrowth. Our objective was to determine the effect of cut strip width and position within the cut strip on the development and quantity of woody biomass produced.

METHODS

Strip thinning occurred on three upland sites located on the West Virginia University Research Forest (Monongalia Co., WV) in March and April 2014. Each

stand was regenerated using shelterwood methods, with the final removal cut occurring 10 years prior to this study. Species composition on this site were representative of other forests in the Allegheny Plateau. The most common species included black cherry (*Prunus serotina*), red oak (*Q. rubra*), red maple (*A. rubrum*), yellow-poplar, and black birch (*B. lenta*) at 33, 13, 12, 12, and 9 percent of stand density, respectively. At the time this study was initiated, the developing stands averaged 3,000 stems/ac and 1.8 inches d.b.h.

For each treatment plot, five pairs of alternating cut and leave strips were established using chainsaws to harvest stems (fig. 2), with strips oriented in a N-S direction. Three different cut strip widths (8, 12, and 16 feet) were examined. Leave strips were 8 feet wide regardless of cut strip width. The measurement plots were restricted to the center three cut strips and two residual strips. Stump height in the cut strips was approximately 4 inches. Each treatment was repeated once on each of the three sites.

The first year re-growth of woody vegetation was estimated on 3 subplots per measurement strip using a 5 feet wide plot with a length equal to the cut strip width. In each subplot, the woody vegetation was clipped and samples were dried at 65 degrees C, and the weight recorded. In addition, samples were separated based on position (west edge, center, east edge) to assess the impact of edge. A 2.6 feet x 5 feet plot was delineated within each subplot to partition biomass growth on the east and west edges of the cut strip as well as for the center of the cut strip. Browse was recorded as present/absent for each stem.

All stems >1 inch d.b.h. in the residual uncut strips were tagged on two, 60 feet long measurement plots per treatment plot. In addition, three 1/100 ac plots were assigned to a uncut area to serve as control plots that were not impacted by the strip thinning. All stems within the control plots were also tagged and measured. All tagged stems were remeasured at the end of the first growing season following release.

Biomass and residual tree diameter growth were compared by cut strip width treatment using ANOVA with subsampling. Overall treatment effects and Tukey mean separation tests were assessed at $P \leq 0.05$. SAS 9.3 (SAS, Inc, Cary, NC) was used for statistical analyses.

RESULTS

Woody biomass was recorded separately for stem sprouts and new seedlings. Sprout biomass varied by cut strip width (P=0.02), with each successively wider strip producing statistically greater biomass. Sprout biomass averaged 1.52, 3.00, and 5.97 oven dried (od) tons per cut ac for 8, 12, and 16 feet strips, respectively. Sprout biomass was greatest in the center of each strip. Biomass on east and west edges were significantly less (56 and 73 percent, respectively) of the biomass present on the center plots (P=0.001). New seedlings represented a very small fraction of the overall biomass. On cut strips, new seedling biomass was 0.019 ton/ cut

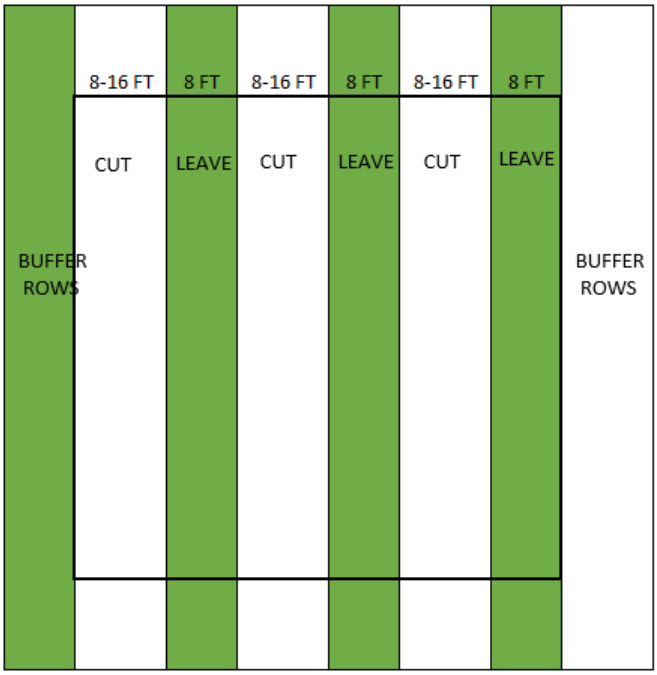


Figure 2—Illustration of plot layout for strip thinning treatment layout. Measurement plots were located in the center plot.

ac on 8 feet strips, 0.013 ton/ cut ac on 12 feet strips, and 0.009 ton/ cut ac on 16 feet strips.

Browsing was very common across the sites. On average, 45 percent of the stems were browsed during the first growing season following treatment. Although browse was widespread, certain species were preferred (Table 1). Sourwood was the most frequently browsed species with 88 percent of stems browsed, although red maple and northern red oak were also browsed more than 70 percent of the time.

Diameter growth for tagged trees in the residual uncut strips ranged from 0.18-0.20 inches for the three strip widths. By contrast, the seedlings on the control plots grew 0.12 inches. However, this difference among treatments was not significant (P=0.11).

DISCUSSION

The coppice yields reported here compare favorably with those reported for commercial plantations (3 to 6 tons/acre/year) in the north central region of the U.S. (Zalesny and others 2011). While bioenergy markets are expected to increase in the future, it is possible that suitable biomass outlets could fail to develop in this region. The advantage of our strip thinning treatments is that they were designed to provide other benefits. As mentioned, young developing stands are infrequent across the landscape (fig. 2). From a wildlife perspective, the browse available in stands past canopy closure is low, and the lack of early succession habitat has been identified for the decline in certain wildlife populations (Litvaitis 2001). By harvesting strips of trees within these younger stands (with closed canopies), the silvicultural objective of reducing stem density can be achieved, while also increasing the availability and persistence of early successional habitat as the cut stems re-sprout. Over time the cut strips will grow and form a closed canopy again. Subsequent harvesting of these stems can perpetuate the cycle again until at some time in the future when the shading from the residual strips prevents regrowth.

In many parts of the central Appalachian region, deer populations can be very high (exceeding 40 deer/ square mile), which is far greater than pre-European settlement estimate of 8-20 deer/sq. mi. Harvesting forest stands in areas with excessive deer pressure often results in regeneration problems. In fact, fencing is a frequent prescription to ensure re-establishment of woody vegetation (Steiner and others 2008). While deer populations at the WVU Research Forest are not considered high, almost one-half of the seedlings were still browsed (table 1), suggesting limited available food sources. Strip thinning existing young stands

Table 1—Occurrence of browsing on coppiced stems

Species	Scientific name	n	% Browsed
sourwood	*Oxydendrum arboreum*	32	88
red maple	*Acer rubrum*	113	75
northern red oak	*Quercus rubra*	108	70
witch hazel	*Hamamelis virginiana*	35	69
spicebush	*Lindera benzoin*	141	67
white oak	*Quercus alba*	9	67
sassafras	*Sassafras albidum*	28	64
devil's walking stick	*Aralia spinosa*	5	60
yellow-poplar	*Liriodendron tulipifera*	50	52
chestnut oak	Quercus montana	12	42
white ash	*Fraxinus americana*	27	37
black birch	*Betula lenta*	52	31
pin cherry	*Prunus pensylvanica*	4	25
black cherry	*Prunus serotina*	282	7
black locust	*Robinia pseudoacacia*	8	0
cucumbertree	*Magnolia acuminata*	1	0

may also serve a secondary benefit in that besides fencing, managers may be able to disperse the deer population across the landscape so to not allow them to overwhelm any one area. Again, an integrated forest management plan that includes strip thinning of young stands (coppice with standards) at appropriate spatial positions and temporal periods to coincide with forest regeneration activities, will improve the overall sustainability of the forested landscape.

The silvicultural treatment of young age classes provides a convenient mechanism to transform the landscape without negatively impacting the future value and sustainability of forest stands. By focusing management efforts earlier in the rotation, future commercial operations can be much more valuable and provide valuable benefits that would otherwise not be available (Siry and others 2004). While the treatments employed in this study did not statistically affect diameter growth of saplings in the residual strips, the first year change in diameter for saplings remaining in the residual strips was more than 1.5 times the average diameter growth on control plots. The limitation to this method of reducing densities in young stands is the inability to alter species composition, as strip thinning does not allow for selection of individual stems.

In order for a system like the one describe here to become practical, managers must have reasonable predictions concerning effects on the residual stand, the amount of biomass/browse being produced, and the expected longevity of the system. The experiment described here was designed to address these questions.

LITERATURE CITED

Bella, I.E.; DeFranceschi, J.P. 1982. Growth of lodgepole pine after mechanical strip thinning in Alberta: 15-year results. *Forestry Chronical.* 58(3): 131-135.

Cain, M.D. 1993. Ten-year results from precommercial strip-thinning: paradigm lost or reinforced? Southern Journal of Applied Forestry. 17(1): 16-21.

Dykstra, D.P. 2010. Extraction and utilization of saltcedar and Russian olive biomass. In: Shafroth, P.B.; Brown, C.A.; Merritt, D.M., eds. Saltcedar and Russian olive control demonstration act science assessment. Scientific Investigations Report 2009-5247. Fort Collins, CO: U.S. Geological Survey: 103-116.

Forest Inventory and Analysis. 2007. http://nrs.fs.fed.us/futures/current_conditions/states/?state=WV. [Date accessed: January 17, 2013].

Greenberg, C.H.; Collins, B.; Thompson III, F.R.; McNab, W.H. 2011. Introduction: What are early successional habitats, why are they important, and how can they be sustained? In: Greenberg, C.; Collins, B.S.; Thompson III, F., eds. Sustaining young forest communities. Managing Forest Ecosystems, Vol. 21. The Netherlands: Springer: 1-10.

Heller, M.C.; Keoleian, G.A.; Volk, T.A. 2003. Life cycle assessment of a willow bioenergy cropping system. Biomass and Bioenergy. 25(2): 147-165.

Litvaitis, J.A. 2001. Importance of early successional habitats to mammals in eastern forests. Wildlife Society Bulletin. 29(2): 466-473.

McCreary, D.D.; Perry, D.A. 1983. Strip thinning and selective thinning in Douglas-fir. Journal of Forestry. 6(1): 375-377.

Miller, G.W.; Stringer, J.W.; Mercker, D.C. 2007. Technical guide to crop tree release in hardwood forests. University of Tennessee Extension Publication PB-1774. Knoxville, TN: University of Tennessee. 24 p.

Pham, C.H. 1985. Thinning of young natural hardwood regeneration in southeastern West Virginia. In: Dawson, J.O.; Majerus, K.A., eds. Proceedings of the 5th central hardwood forest conference. Urbana-Champaign, IL: University of Illinois: 25-34.

Robison, D.J.; Schuler, J.L.; Jervis, L. [and others]. 2004. Individual tree release and enrichment planting in young natural upland hardwoods. In: Conner, K., ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 283-286.

Roise, J.P.; Hannum, L.C.; Catts, G.P. 2009. Machine system for harvesting small diameter woody biomass and reducing hazardous fuels: a developmental report. In: Proceedings of the 2009 ASABE Bioenergy Engineering Conference. Bellevue, WA: 1-15.

Schuler, J.L.; Robison, D.J. 2006. Stand development and growth response of 1 to 6 year old natural upland hardwoods to silvicultural treatments. Forest Ecology and Management. 232 (1/3): 124-134.

Schuler, J.L.; Robison, D.J. 2009. Response of reproduction and residual overstory trees to even-aged regeneration methods in southern hardwoods. In: Columbus, F., ed. Forest regeneration: ecology, management and economics. New York: Nova Publishers: 71-86.

Siry, J; Robison, D.J.; Cubbage, F.W. 2004. Economic returns model for silviultural investments in young hardwood stands. Southern Journal of Applied Forestry. 28(4): 179-184.

Smith, H.C.; Lamson, N.I. 1983. Precommercial crop-tree release increases diameter growth of Appalachian hardwood saplings. Res. Pap. NE-534. Broomall, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 5 p.

Steiner, K.C.; Finley, J.C.; Gould, P.J. [and others]. 2008. Oak regeneration guidelines for the central Appalachians. Northern Journal of Applied Forestry. 25(1): 5-16.

Trimble, G.R. 1973. Response to crop-tree release by 7-yr-old stems of yellow-poplar and black cherry. Res. Pap. NE-253. Broomall, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 10 p.

Zalesny, R.S.; Cunningham, M.W.; Hall, R.B. [and others]. 2011. Woody biomass from short rotation energy crops. In: Zhu, J. [and others], eds. Sustainable production of fuels, chemicals, and fibers from forest biomass. American Chemical Society Symposium Series. Washington, DC: American Chemical Society: 27-63.

THE EFFECTS OF HARVESTING SHORT ROTATION COTTONWOOD WITH TREE SHEARS IN ARKANSAS

Matthew H. Pelkki, Michael Blazier, Jonathan Hartley, Hal Liechty, and Bryce Zimmermann[1]

Abstract—Short-rotation cottonwood plantations were established on a marginal agricultural site in the lower Mississippi Alluvial Valley in southeast Arkansas using two known clones (S7C20 and ST-66) and nursery-run cottonwood stock (MIXED) from the Louisiana Department of Agriculture and Forestry nursery. The cottonwood was grown for five seasons and harvested in the winter of 2013-2014. Harvesting was done by a chainsaw (control) and a mechanical tree shear (treatment). Regeneration in the form of stump sprouts was evaluated after one growing season for survival, total number of sprouts, sprouts taller than 137 cm (4.5 feet), ground line diameter, diameter at breast height (4.5 feet) and total height. Harvesting cottonwoods using tree shears significantly increased mortality, and decreased number of sprouts and diameter and total height of sprouts one year after harvest.

INTRODUCTION

The growing demand for renewable energy is driving the development of dedicated agricultural and forestry systems producing biomass for energy. The Lower Mississippi Alluvial Valley (LMAV) region has great potential for biomass production due to a lengthy growing season, well-developed agricultural industry, and excellent transportation and energy distribution systems (Trip et al. 2009). A biomass production system based on two native species, cottonwood (*Populus deltoides*) and switchgrass (*Panicum virgatum*) was established on marginal soils in the LMAV in 2009 (Pelkki et al. 2009). This particular site was considered marginal due to low productivity and the difficulty in irrigating this site because of its location. Bioenergy systems in the LMAV have the potential to increase farm revenues and by producing biomass for energy on lands that are considered unacceptable for row crop production we can avoid a "food vs. fuels" conflict (Blazier et al. 2014).

The long-term sustainability and economic returns of such agroforestry systems depends on adequate survival and natural regeneration from coppice regenerated cottonwood trees (Liechty et al. 2012). Unless multiple generations of cottonwood biomass can be produced from a single, initial planting, the cost of the establishment and production of cottonwood biomass is too great to justify use in bioenergy production. In this study, first rotation costs of cottonwood establishment and management produced biomass that cost $65 to $140 per oven-dry metric ton in the first rotation. These costs can be reduced to as little as $20-26/ODMT on the stump if four rotations of cottonwood can be harvested from the same stool bed (Liechty et al 2012).

In Europe a great deal of small scale harvesting equipment is commercially available for bioenergy harvesting (Ehlert and Pecenka 2013, Vanbeveren et al. 2015). While these systems can be purchased in the United States, they often cost in excess of $75,000, so we chose to use a harvest system that could be mounted to a tractor of typical size on most farms in the United States.

This research study focuses on the post-harvest survival and growth of three cottonwood clones after harvesting using two different techniques, hydraulic tree sheers or chainsaws to sever the tree from the root system. The objectives of this study are to test for treatment differences in survival, number of sprouts, and height and diameter growth in the initial year after harvest.

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METHODS

In 2009, two known cottonwood clones (S7C20 and ST-66) were planted along with a nursery-run of mixed cottonwood clones (MIXED) from a state-run tree nursery in northwest Louisiana. The trees were planted on the University of Arkansas's Southeast Research and Extension Center Desha County, Arkansas. The site was an abandoned row crop field with Sharkey and Desha clay soils. Pre-planting herbicides burned down all competition, and the site was ripped to a depth of 36 cm prior to planting on a 1.22m x 1.83m spacing, or 4480 trees per hectare. Alternating rows contained ST-66, S7C20, and MIXED trees, and overall plot sizes were 30m by 90m.

During the first rotation, pre-emergent herbicides were applied in the first and second growing season to control competition. At the start of the second growing season, a banded ammonium nitrate application of 168 kg/ha was applied as a fertilizer. During the second growing season, a cottonwood leaf beetle infestation was controlled with an insecticide application.

In 2009, in each of three replicated 30x90m plots, 60 trees of each clone were selected for the study, for a total of 180 trees. These trees were monitored annually for survival and growth in total height (THT), ground-line diameter (GLD) and diameter at breast height (DBH). After five growing seasons, in November/December of 2013, half the trees were selected for harvest by chainsaw felling (by hand) and the other half of the trees were harvested by a hydraulic shear mounted on a trac-loader. This tree shear, a Hydra-Snip manufactured by M&M Engineered Products, LLC in Coffeyville, Kansas, is capable of being mounted on a front-end loader equipped tractor. It is capable or cutting a 12" diameter stem in a single pass. The cost of this shear was under $15,000, making the capital investment for a bioenergy harvest system much lower than other options.

After one growing season, the trees were measured for survival, total number of live sprouts, total number of sprouts with height greater than 137 cm (breast height), total height of tallest sprout, and ground line diameter (GLD) and diameter at breast height (DBH) of the tallest sprout. Two way analysis of variance for main and interaction effects (replication and clone) was completed with SYSTAT 13 software.

RESULTS AND DISCUSSION

Of the 180 trees that were planted in 2009 as part of this study, survival was acceptable, averaging 84 percent after the first year (table 1). From 2010-2013, only another 2 percent of the trees died, leaving 148 of the original 180 trees alive to be harvested in 2013. Prior to harvest, there were no significant differences in trees' diameters by their assigned treatment or clone. However, all trees in replication three were significantly taller than trees in replications one and two. It should be noted that while the study site had very little slope (0-1 percent), it did drain through the third replication's plot.

Stump height was only loosely controlled during the harvest. The chainsaw fellers had instructions to leave a stump with a height of 5-10 cm and the shear operator was instructed to leave as short a stump as possible without cutting below the ground line. Post-harvest analysis showed shear stump heights from 5-15 cm with an average of 10cm, and chainsaw stump heights ranged from 5-10cm with an average of 8 cm. No significant difference was found in stump heights in any treatment or clone.

One year after harvesting, there was a significant harvest effect on mortality (table 2). There was a significant increase in mortality among those trees harvested using tree shears ($p<0.001$). It was also noted that the MIXED trees had a significantly higher survival ($p=0.034$) and that replication three had greater survival ($p<0.001$) than replications 1 and 2. Replication three was the wettest of the three replications, located at the point where the entire field drained into an irrigation ditch.

There was no significant difference ($p=0.096$) in the treatments in the number of large sprouts (THT ≥ 137 cm) among the treatments, nor was there a significant difference ($p=0.697$) in the total number of all sprouts between the two treatments (table 3). Replication 3 did have fewer large sprouts ($p=0.02$) and fewer total sprouts ($p<0.001$) for all genotypes. As the drainage for the study site ended near the plot for replication 3, this site was under water the longest in the spring after harvest, which delayed all cottonwoods from sprouting as early as those in replications 1 and 2.

Shearing had a negative effect on both ground line diameter (GLD) and diameter at breast height (DBH) as shown in table 4. Total height ($p<0.001$) and diameter at breast height ($p=0.001$) were reduced in all clones by harvesting with tree shears. The ST-66 clone also had an interaction effect with shearing ($p=0.024$) on ground line diameter.

Finally, total tree height after one growing season was negatively affected ($p<0.001$) by harvesting the trees using tree shears (table 5). While after five growing seasons the trees in replication 3 had significantly greater total height, this effect was not apparent one growing season after harvest.

CONCLUSIONS

The tree shears used were quite new and the blades very sharp, minimal stump damaged was observed from

Table 1—Survival of three planted cottonwood genotypes in southeast Arkansas over a five-year rotation

Genotype	2009 Planted	End of 2009 Live Trees	End of 2013 Growing Season (pre-harvest) Live Trees (% Survival of trees planted in 2009)
CHAINSAW HARVESTED TREES (CONTROL)			
ST-66	30	22	22 (73%)
S7C20	30	28	27 (93%)
MIXED	30	28	28 (93%)
TREE SHEAR HARVESTED TREES			
ST-66	30	23	23 (77%)
S7C20	30	27	26 (90%)
MIXED	30	23	22 (77%)

Table 2—Survival of three cottonwood genotypes one year after harvesting with chainsaw and tree shear in southeast Arkansas

Treatment: Chainsaw (control)				
	Pre-harvest		Post-harvest	
Clone	Count	% Survival (trees alive in 2013)	Count	% Survival of trees harvested in 2013
ST-66	22	100%	22	100%[a]
S7C20	27	100%	26	96%[a]
MIXED	28	100%	28	100%[a]

Treatment: Tree Shear				
	Pre-harvest		Post-harvest	
Clone	Count	% Survival (trees alive in 2013)	Count	% Survival of trees harvested in 2013
ST-66	23	100%	16	69%[b]
S7C20	26	100%	19	73%[b]
MIXED	22	100%	19	86%[b]

Survival percentages with different superscripts are significantly different at $\alpha = 0.05$

Table 3—Number of large sprouts (THT ≥ 137 cm) per stump and number of total sprouts per stump on cottonwood genotypes one year after harvest in Southeast Arkansas

Large Sprouts (THT ≥ 137 cm)		
	Harvest Method	
Clone	Chainsaw	Tree Shear
ST-66	3.3[a]	2.8[a]
S7C20	5.0[a]	2.3[a]
MIXED	3.4[a]	2.2[a]
All Live Sprouts		
	Harvest Method	
Clone	Chainsaw	Tree Shear
ST-66	7.1[b]	6.1[b]
S7C20	6.4[b]	7.1[b]
MIXED	4.9[b]	6.7[b]

Numbers of sprouts with different superscript are significantly different at α = 0.05.

Table 4—Ground line diameter and diameter at breast height of the tallest sprout one year after harvesting with chainsaw or tree shears for three cottonwood genotypes in southeast Arkansas

Ground Line Diameter (mm)		
Clone	Harvest Method	
	Chainsaw	Tree Shear
ST-66	24.61[a]	17.13[c]
S7C20	24.60[a]	20.90[b]
MIXED	26.29[a]	22.53[b]
Diameter at Breast Height (mm)		
Clone	Harvest Method	
	Chainsaw	Tree Shear
ST-66	10.21[d]	8.84[e]
S7C20	12.17[d]	9.13[e]
MIXED	11.48[d]	9.23[e]

Numbers of sprouts with different superscript are significantly different at α = 0.05.

Table 5—Total height of the tallest sprout one year after harvesting with chainsaw or tree shears for three cottonwood genotypes in southeast Arkansas

Clone	Total Tree Height (cm)	
	Harvest Method	
	Chainsaw	Tree Shear
ST-66	231[a]	202[b]
S7C20	228[a]	201[b]
MIXED	227[a]	203[b]

Tree heights with different superscript are significantly different at α = 0.05.

the shears and by the following growing season, it was impossible to distinguish between chain sawed stumps and sheared stumps. The harvests took place during November and December of 2013, the soil was quite saturated with water though rutting in the site from the trac-loader was minimal. However, soil displacement might have caused some damage to trees harvested in this fashion.

The mortality caused by the tree shears is substantial. After five years in the first rotation, 86 percent of the trees (77 or 90) in the three plots assigned to be chain saw felled were alive. This is a good level of stocking. One year after harvest, 84 percent of the trees (76 of 90) were alive and this is still a good level of stocking for the second rotation. In the three sheared plots, 79 percent of the trees were alive (71 of 90) at the end of the first rotation. One year after harvest with the tree shears, only 60 percent of the stems (54 of 90) were alive. The additional mortality that is attributed to the harvest method would require supplemental planting to maintain full site utilization and this would raise costs of biomass production.

We did not test the impact of harvest timing on cottonwood mortality. Both the chainsaw felling and shearing took place in November and December, and at the time the trees appeared fully dormant. If harvest were to occur in early September at the end of the growing season, the soil on this site would not be as wet, soil disturbance impacts would likely be minimized, and mortality from the equipment might be less.

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LITERATURE CITED

Blazier, M.A.; Liechty, H.O.; Pelkki, M.H. [and others]. [In press]. Cottonwood and switchgrass for bioenergy crops and ecosystem services in the Lower Mississippi Alluvial Valley. In: New crops: bioenergy, biomaterials, and sustainability: proceedings of the 7th national new crops symposium. Washington, DC: 185-199.

Ehlert, D.; Pecenka, R. 2013. Harvesters for short rotation coppice: current status and new solutions. International Journal of Forest Engineering. 24(3):170-182.

Liechty, H.O.; Blazier, M.; Pelkki, M. [and others]. 2012. Potential for using agroforests for bioenergy production in the Lower Mississippi Alluvial Valley. In: Meyer, S.R., ed. IUFRO small-scale forestry conference 2012: science for solutions conference proceedings, 24-27 September 2012. Amherst, Massachusetts: The Center for Research on Sustainable Forests, University of Maine: 88-92.

Pelkki, M; Liechty, H.; Blazier, M. [and others]. 2012. Building a better biomass ecosystem: cottonwood-switchgrass agroforests on marginal land. In: 2012 national conference: science for biomass feedstock production and utilization. New Orleans, LA. 8 p.

Tripp, S.; Powell, S.R.; Nelson, P. 2009. Regional strategy for biobased products in the Mississippi Delta, Executive Summary. Memphis, TN: Battelle Technology Partnership Practice. 23 p.

Vanbeveren, S.P.P., Schweier, J.; Berhongaray, G.; Ceulemans, R. 2015. Operational short rotation woody crop plantations: Manual or mechanised harvesting? Biomass and Bioenergy. 72: 8-18.

Poster Session

WEATHER EFFECTS ON THE SUCCESS OF LONGLEAF PINE CONE CROPS

Daniel J. Leduc, Shi-Jean S. Sung, Dale G. Brockway, and Mary Anne S. Sayer[1]

Abstract—We used National Oceanic and Atmospheric Administration weather data and historical records of cone crops from across the South to relate weather conditions to the yield of cones in 10 longleaf pine (*Pinus palustris* Mill.) stands. Seed development in this species occurs over a three-year time period and weather conditions during any part of this span could have varying effects on the final seed crop. Weather had a significant effect on cone crops, but the relationship across many years was complex and could not be attributed to any small subset of variables.

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.) has long been known to have irregular cone crops, though it is not known what factors control the bounty of the cone crop. There are three critical stages in cone development that take place across three years: reproductive primordia development in July and August of year one, pollination in March of year two, and ovule fertilization in May of year three.

Previous research has attempted to relate weather to cone production of longleaf pine. Shoulders (1967) looked at rainfall effects on flowering while Pederson and others (1999) looked at precipitation and temperature effects on cone production. However, both of these papers looked only at a single location, whereas we have included 10 locations and more years of data.

DATA

In 1958, a spring binocular count of green cones on longleaf pine trees was initiated at the Escambia Experimental Forest in Alabama by the Forest Service, U.S. Department of Agriculture. This one location has now expanded to include 10 locations across the South (table 1; Brockway and Boyer 2014). It is important to

Table 1—Locations in six southern states at which counts of green longleaf pine cones were made and the climate divisions which correspond to them

Cooperator	State and County	State and Climate Division Number
Kisatchie National Forest	Louisiana, Grant	16, 5
Cedar Creek Company	Alabama, Escambia	1, 7
Blackwater River State Forest	Florida, Santa Rosa	8, 1
Eglin Air Force Base	Florida, Okaloosa	8, 1
Apalachicola National Forest	Florida, Leon	8, 1
Jones Ecological Research Center	Georgia, Baker	9, 7
Tall Timbers Research Station	Florida, Leon	8, 1
Fort Benning Military Base	Georgia, Chattahoochee	9, 4
Sandhills State Forest	South Carolina, Chesterfield	38, 4
Bladen Lakes State Forest	North Carolina, Bladen	31, 6

Note: State and climate division numbers are listed by NOAA (2015a).

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note that this paper is based on the count of green cones performed during April and May of the seed year, even though there will be some cone loss during the subsequent summer before the seed is released in the fall. To our knowledge, these are the only longleaf pine cone counts for which a long period of historical data were available.

In addition to this Forest Service data collection, weather data for the same periods were obtained from the National Oceanic and Atmospheric Administration's National Climatic Data Center (NOAA 2014) as averages for each climate division (NOAA 2015b) where cone counts were done (table 1). Variables used included monthly summaries of (1) average air temperature; (2) maximum air temperature; (3) minimum air temperature; (4) Palmer Drought Severity Index (PDSI) (positive is wet, negative is dry); (5) precipitation; (6) heating degree days (the number of degrees that each day's average temperature is <65 °F (NOAA 2015c); higher is colder); and (7) cooling degree days (the number of degrees that each day's average temperature is >65 °F (NOAA 2015c); higher is warmer). The combination of conelet counts and weather data resulted in 390 observations for analysis.

METHODS

Relationships between cone crops and weather were evaluated in two ways: as raw cone counts and as classes of bumper, good, and poor cone crops. The classifications are based on those of Brockway and Boyer (2014), but the number of classes was reduced to highlight differences (table 2).

Table 2—Definitions of longleaf pine cone crop quality used in this paper

Crop Quality	Cones per Tree
Bumper crop	≥100
Fair to Good crop	25 to 99
Poor to Failed crop	<25

There are three time periods thought to be very important in longleaf pine cone development that guided our initial variable selection. Primordia development occurs during July and August two years before the seed crop (year one); pollination occurs in March one year before seed crop (year two); and fertilization occurs in May of the seed year (year three) (Croker 1971, Eggler 1961). To include factors not accounted for by these time periods, we also used monthly weather data values for all other time periods from January of year one through June of year three just after the cone count was assessed. The variables for the pollination and fertilization stages are equivalent to the monthly values, but the variables included to represent the primordia stage were sums (precipitation, cooling degree days, heating degree days) or averages (PDSI, average temperature, maximum temperature, minimum temperature) of the values for the months of July and August. Furthermore, in this dataset, the value for heating degree days in July and August were always zero and could not be used in any analysis. This creates six new variables, but excludes four for the correlation analysis, resulting in a total of 212. For the discriminant analysis, the values for the months of July and August had to be excluded because they were already included in the primordia stage variables, resulting in 200 variables.

We used SAS statistical software to analyze the data. Canonical discriminant analysis was done with the software's PROC CANDISC routine (SAS Institute 2004) using the 200 variables selected and the three cone-crop classes described in table 2. After this initial analysis, PROC STEPDISC (SAS Institute 2004) was used to try to reduce the number of variables included. Significant variables arising from PROC STEPDISC were then analyzed using PROC CANDISC to benefit from the dimension-reducing principal components analysis of this procedure, making it easier to visualize the results (fig. 1).

With PROC CORR (SAS Institute 2004), Pearson correlation coefficients were calculated for all the measured monthly weather variables from January of year one through June of year three, as well as for the six combined primordia variables. The monthly results were then plotted as bar graphs to enable visualization of the ups and downs of correlation through time (fig. 2).

RESULTS

With all 200 variables included for the discriminant analysis, all cone production classes were significantly different ($\alpha = 0.05$) based on Wilk's Lambda. When a stepwise discriminant analysis selected only 36 significant variables, all of the classes were still significantly different; however, this is not obvious from the graph (fig. 1). The variables selected as significant are listed in table 3. Of the significant variables, 17 were related to moisture and 19 were related to temperature. Since there are 58 possible moisture variables and 142 possible temperature variables, it is useful to note that 29 percent of the possible moisture variables and 13 percent of the possible temperature variables entered the model. Of the significant variables, 3 were from conditions in year three, 13 were from year two, and 20 were from year one. As percentages of the possible variables, this is 7 percent from year three, 16 percent from year two, and 26 percent from year one. A linear discriminant function using all of the variables failed to correctly classify the cone production class 8 percent of the time, while a linear discriminant function using the

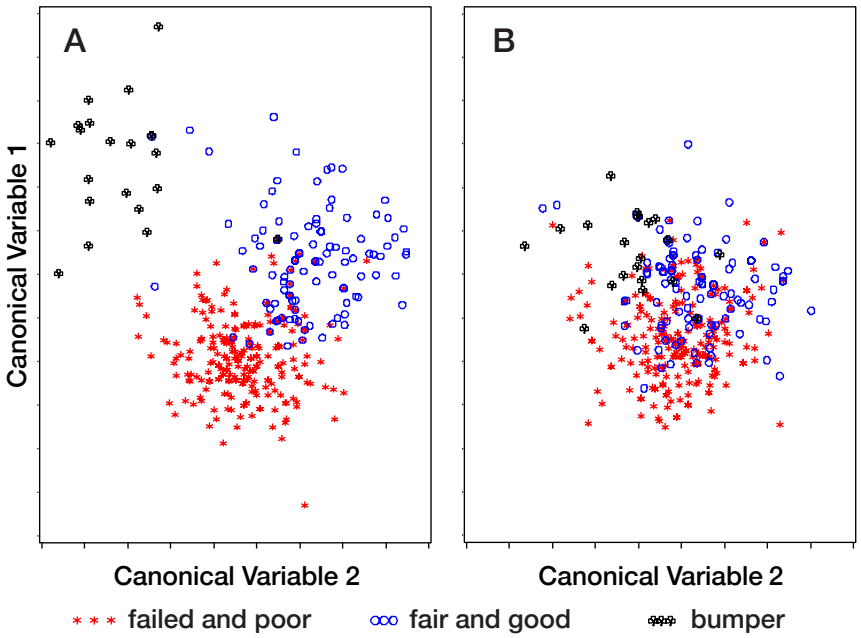


Figure 1—Results of canonical discriminant analysis showing that 200 variables visually delineate cone production classes (A), but that the 36 significant variables alone (B) do not. However, they are both statistically significant for discriminating between all classes. Note that there are no numbers on the axes because the canonical variables are combinations of the real variables, and thus the numbers have no obvious meaning.

36 significant variables failed to correctly classify cone production 24 percent of the time.

In the correlation analysis, there were 57 significant variables ($\alpha = 0.05$). Of these, 32 described conditions in year one, 15 described conditions in year two, and 10 were conditions in year three. As percentages of the possible variables, 37 percent were from year one, 18 percent were from year two, and 24 percent were from year three. Fifteen of the significant variables were moisture related and 42 were temperature related, which is to say that 25 percent of the possible moisture variables and 28 percent of the possible temperature variables were significant.

It is important to know which variables were significantly correlated with cone crop, but it is also important to know if they exert a positive or negative influence on the crop. Table 4 lists the 30 months included in this study along with an indicator of what conditions are best for cone crops based on significant correlations. The results of previous studies by Pederson and others (1999) and Shoulders (1967) are also listed for comparison.

DISCUSSION

Shoulders (1967) found that low rainfall amounts in April, May, June, and July of year one reduced the flowering ability of trees, and high rainfall amounts increased it. We found that only rainfall in June and July of year one was significantly correlated with cone production.

Pederson and others (1999) found that warmer temperatures in May, June, and October of year one were positively correlated to cone yield. In year two, a cool April and warm July and August were positively correlated to cone yield. Our results generally agree, except that we found a warm winter was more important than a warm spring in year one. We also found some additional correlations, such as the positive influence of a cooler July, August, and September and a warmer October, November, and December in year one; a cooler June in year two; and a cooler winter and spring in year three.

Pederson and others (1999) also found that precipitation was positively correlated with cone crops during July of year one and during October and November of year two. This matches our results, except that we also found a positive correlation if conditions were wetter in June, September, and October of year one and wetter throughout the winter and spring of year three (fig. 2A-B). One reason why we may have found additional moisture correlations was that, in addition to rainfall, we also looked at PDSI, which smoothed out the trends of rainfall. While many of our correlations were not significant, the general trend expressed by PDSI is that it is beneficial to cone production for conditions to be dry before primordia formation and wet afterwards.

Figure 2 also shows other correlations between the weather data variables analyzed and cone yield. Average temperature correlations (fig. 2C) show

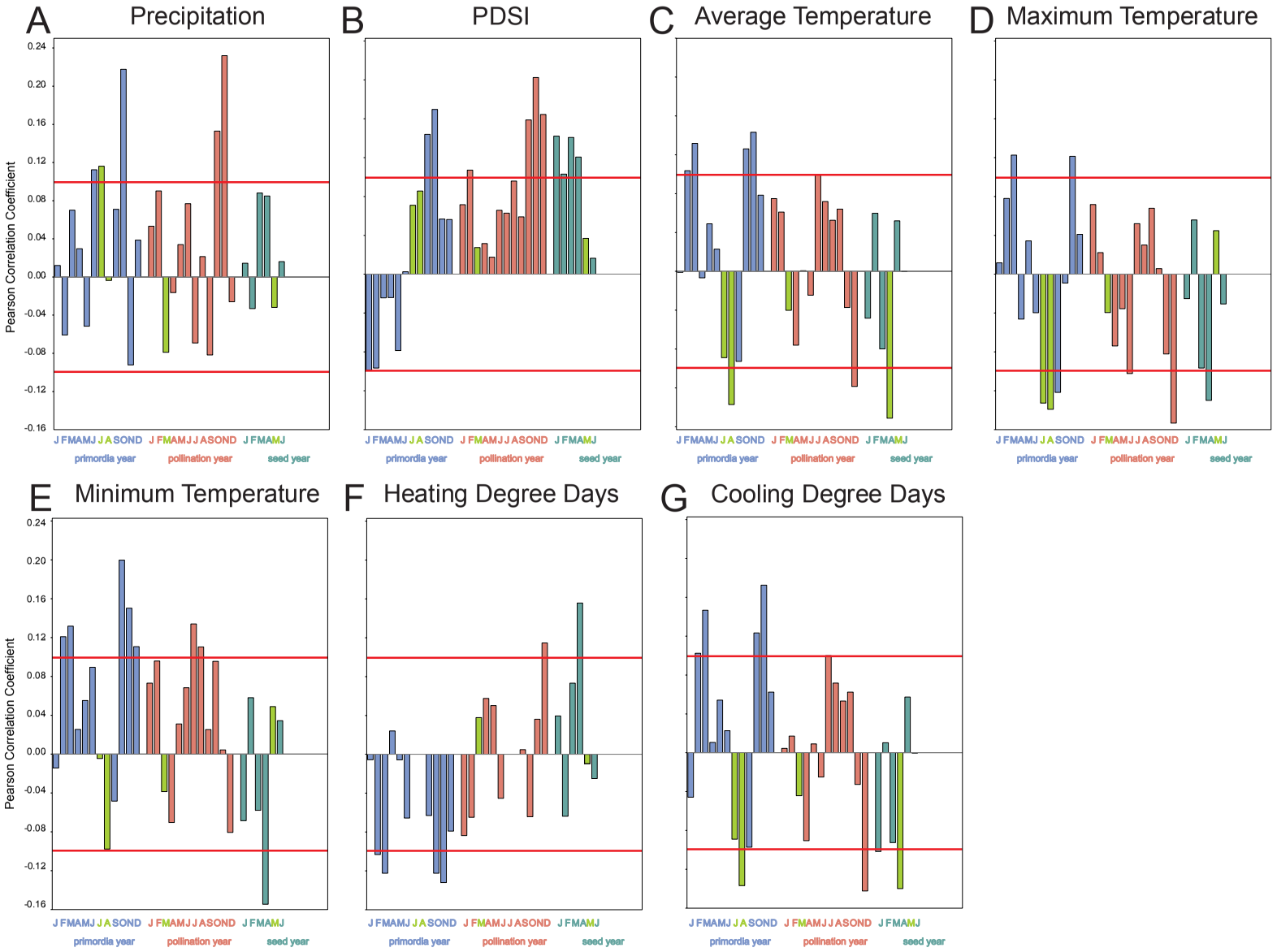


Figure 2—Pearson correlation coefficients are plotted for each of the weather variables analyzed. The red line in each graph marks the approximate limit for significance of the correlations. The bars colored light green indicate the important stages in seed development (primordia development, pollination, and ovule fertilization). This figure includes the following monthly variables: (A) precipitation, (B) Palmer Drought Severity Index (PDSI), (C) average temperature, (D) maximum temperature, (E) minimum temperature, (F) heating degree days, and (G) cooling degree days.

Table 3—Weather data variables included in the significant discriminant function by stepwise discriminant analysis

Variable Description	Correlation
August minimum temperature in year two	positive
July and August temperature in year one	negative
July cooling degree days in year two	positive
June cooling degree days in year three	negative
September precipitation in year two	negative
October precipitation in year one	positive
April heating degree days in year one	positive
May PDSI in year one	negative
November maximum temperature in year one	positive
April minimum temperature in year one	positive
July and August precipitation in year one	positive
April precipitation in year two	negative
April PDSI in year one	negative
March heating degree days in year three	positive
June temperature in year three	negative
August precipitation in year two	positive
October PDSI in year two	positive
April cooling degree days in year two	negative
September heating degree days in year one	negative
June minimum temperature in year two	positive
July precipitation in year two	negative
July maximum temperature in year two	positive
August cooling degree days in year two	positive
May minimum temperature in year two	positive
December PDSI in year one	positive
September PDSI in year one	positive
November heating degree days in year one	negative
October PDSI in year one	positive
October heating degree days in year one	negative
February precipitation in year one	negative
March heating degree days in year one	negative
November PDSI in year one	positive
November precipitation in year one	negative
February precipitation in year two	negative
March PDSI in year one	negative
September maximum temperature in year one	negative

Note: The order of the variables in the table is the order of entry into the model, indicating greater significance to variables at the top. Whether this variable has a positive or negative effect on the cone yield is obtained from the correlation analysis.

PDSI = Palmer Drought Severity Index.

that it is better to be warmer before and after primordia formation, but cooler during the actual event. It is also better to be cooler through the winter and spring of the year after pollination. Maximum temperature correlations with cone yield (fig. 2D) basically mirror those done with average temperature, but they tend to have stronger negative correlations. Minimum temperature correlations (fig. 2E) have stronger positive correlations. These results suggest that temperatures too hot and too cold are negatively associated with cone crop, and that there is likely some moderate temperature that is best. Heating degree days (fig. 2F) is another measure of deviations from an average, and the correlations suggest that it is better to be warm until pollination and mostly cooler afterwards. Correlations between cooling degree days and cone crop (fig. 2G) suggest that it is also good to have a warm summer after pollination; this is not shown as strongly by heating degree days, which go to zero in the summer months.

It should be noted that correlations were tallied by their significance, but that even the highest correlations are still quite low. Exceeding a significance value allows for counting, but it should not be considered definitive. There is always statistical error, and it is expected that some of the tallied significant correlations are not actually significant. The real value of the correlation analysis is in looking at trends and their strengths.

The discriminant analysis primarily showed that weather data alone can be used to predict cone crops. However, it is a complex function requiring many inputs that can be difficult to obtain. Many of the variables that were significant in the discriminant function are also significant in the analysis of correlation coefficients, but several were not. At this time, it is not possible to determine if the inclusion of these variables is the result of a complex interaction or a statistical artifact.

CONCLUSIONS

Cone crops are not simple, cyclical events that are independent of weather. Temperature and moisture during the three years leading up to a cone crop have significant effects on the crop size. One weakness in this study is that it is based on the spring cone count, and events that affect the final seed yield are not all accounted for. However, spring cone counts have been used for years as good estimates of the expected yield of mature cones. The main weakness of using only the spring cone count is that the lead-time for taking advantage of a good or poor seed year is short. Given that the weather two years before a cone crop is often significant, it may be possible to develop a model to predict cone crops two years in advance. This would be a useful tool for forest managers and could be a future research effort.

Table 4—A summary of the results from the correlation analysis of the current study, along with the conclusions of Pederson and others (1999) and Shoulders (1967), showing the weather conditions that were found to be best for cone crops in each study

Year	Month	Current Study	Pederson	Shoulders
bud year	January			
	February	warmer		
	March	warmer		
	April			wetter
	May		warmer	wetter
	June	wetter	warmer	wetter
	July	wetter, cooler	wetter	wetter
	August	cooler		
	September	wetter, cooler		
	October	wetter, warmer	warmer	
	November	warmer		
	December	warmer		
pollination year	January			
	February			
	March			
	April		cooler	
	May			
	June	cooler		
	July	warmer	warmer	
	August	warmer	warmer	
	September			
	October	wetter	wetter	
	November	wetter	wetter	
	December	wetter, cooler		
seed year	January	wetter, cooler		
	February	wetter		
	March	wetter, cooler		
	April	wetter, cooler		
	May			
	June			

Notes: Variables related to moisture are PDSI and precipitation. Variables related to temperature are average monthly temperature, monthly high temperature, monthly low temperature, monthly heating degree days, and monthly cooling degree days. Although these variables express different aspects of moisture and temperature respectively and have different levels of significance, there were no conflicts in the trend expressed.

Missing cells indicate that no significant relationship was found.

LITERATURE CITED

Brockway, D.; Boyer, B. 2014. Longleaf pine cone prospects for 2013 and 2014. http://www.srs.fs.usda.gov/longleaf/2014_Cone_REPORT.pdf . [Date accessed: May 8, 2015].

Croker, T.C. 1971. Female strobilus stages of longleaf pine. Journal of Forestry. 69(2): 98-99.

Eggler, W.A. 1961. Stem elongation and time of cone initiation in southern pines. Forest Science. 7(2): 149-158.

NOAA. 2014. Index of /pub/data/cirs/climdiv. http://www1.ncdc.noaa.gov/pub/data/cirs/climdiv/ [Date accessed: December 5, 2014].

NOAA. 2015a. U.S. climate divisions. http://www.ncdc.noaa.gov/monitoring-references/maps/us-climate-divisions.php. [Date accessed: December 11, 2015].

NOAA. 2015b. An overview of NCDC's new climate division dataset. http://www1.ncdc.noaa.gov/pub/data/cirs/climdiv/nCLIMDIV-briefing-cirs.pdf. [Date accessed: May 29, 2015].

NOAA. 2015c. Questions about heating, cooling degree days. http://www.erh.noaa.gov/cle/climate/info/degreedays.html. [Date accessed: May 29, 2015].

Pederson, N.; Kush, J.S.; Meldahl, R.S. 1999. Correlating climate and longleaf pine cone crops: is there a connection? In: Kush, J.S., comp. Report No. 4. Longleaf pine: a forward look, proceedings of the 2nd Longleaf Alliance conference. Charleston, SC: Longleaf Alliance: 140-143.

SAS Institute Inc. 2004. SAS/STAT 9.1 User's Guide. Cary, NC: SAS Institute Inc. 5121 p.

Shoulders, E. 1967. Fertilizer application, inherent fruitfulness, and rainfall affect flowering of longleaf pine. Forest Science. 13(4): 376-383.

SIMULATION OF DESIGN-UNBIASED POINT-TO-PARTICLE SAMPLING COMPARED TO ALTERNATIVES ON PLANTATION ROWS

Thomas B. Lynch, David Hamlin, and Mark J. Ducey[1]

Total quantities of tree attributes can be estimated in plantations by sampling on plantation rows using several methods. At random sample points on a row, either fixed row lengths or variable row lengths with a fixed number of sample trees can be assessed. Ratio of means or mean of ratios estimators can be developed for the fixed number of trees option but are not design-unbiased. Ducey's estimator samples a fixed number of trees and is design-unbiased, as is sampling fixed row length. Simulations indicted that some of the ratio estimators could have substantial bias on rows containing large gaps, but for other ratio estimators bias was minimal in practical terms. Ducey's method is unbiased but had a root mean square error slightly larger than some of the ratio estimators.

Ducey (2012) suggested a method of selecting a fixed number of sample trees on a line that is design-unbiased. Borders and others (2012) proposed plantation row sampling, in which plantation rows would be sampled by using either a fixed-length plot or a fixed number of sample trees per sample location. Total row lengths in plantation row sampling are determined by remote sensing methods. Ground-based sampling is used to determine attributes such as number of trees and volume per lineal foot of row. Totals are then obtained by multiplication with row length. The fixed number of sample trees method proposed by Borders utilized ratio estimators that are not design–unbiased, though they may have low bias in practice. Ducey's (2012) method of design-unbiased point-to-particle sampling on lines can be combined with Borders' idea of plantation row sampling to obtain design-unbiased estimates with a fixed number of sample trees per sample location on the row.

Here we compared design-unbiased point-to-particle sampling on plantation rows to five alternative methods using computer simulation. Design unbiased point-to-particle sampling applied to plantation rows samples a fixed, even number of trees on a row for each randomly-located point on a plantation row. One simple alternative is sampling the trees on a fixed length of row for each randomly-located point on a row. Other alternatives include methods which measure the length of row occupied by a fixed number of trees for each randomly located point on the row either including or excluding the "sample gap" into which the random sample point falls. We examined four ways to do this: G-MR is a mean of ratios estimator including the sample gap, G-RM is a ratio of means estimator including the sample gap, NG-MR is a mean of ratios estimator that does not include the sample gap, and NG-RM is a ratio of means estimator that does not include the sample gap. We simulated plantation rows for two scenarios – one including substantial gaps similar to those that might arise from thinning and/or mortality, and another that does not include such gaps which might resemble a younger unthinned plantation. Results were obtained for fixed-tree number designs that have an even number of trees from 2 to 12 and for fixed-length plots that have similar expected numbers of trees per plot.

Each of the estimators was tested on two simulated row populations. We included a fixed-length row estimator designed to sample a mean of 2*k* trees. Estimators having numbers of sample trees 2*k* ranging from 2 to 12 were tested. One population was a "Not Gappy" (NG) row with a target length of 100,000 feet and a mean inter-tree distance of 6 feet ranging from 3.6-7 feet, having a total of 16,667 trees. The other population was "Gappy" (G) with a target length of 100,000 feet, mean inter-tree distance of 9 feet and a standard deviation inter-tree distance of 7.4 feet with inter-tree distances ranging from 3.6 feet to 59.9 feet and a total of 11,109 trees on the row population. For each population and each method, bias percent and Root Mean Square Error percent (RMSE%) was simulated for an estimator having n=10 row locations. Exact bias and variances could be calculated for Ducey's method and the Mean of Ratios (MR) methods. For the Ratio of Means estimators (RM) 1 million simulations were performed using an R (R Core Team 2014) simulation program.

The results indicated that the G-MR and the NG-RM had the best Root Mean Square Error percent (RMSE%).On the G row, the G-RM and the NG-MR were biased significantly. The fixed-length plot is design-unbiased and so did not show bias in the simulations. It performed well in RMSE% but not as well as some

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of the other methods. Ducey's method showed slightly higher RMSE% than some of the alternative methods but is design-unbiased and so unbiased for any spatial distribution including the G and NG simulation rows used here.

LITERATURE CITED

Borders, B.E.; Shirver, B.D.; Cass, R. 2012. Row sampling in planted stands. Unpublished report. Presented at the 2012 southern mensurationists meeting, Jacksonville, FL. On file with Dr. Thomas B. Lynch, Dept. NREM, 008C Ag Hall, Oklahoma State University, Stillwater, OK, 74078. 35 p.

Ducey, M. J. 2012. Simple, design-unbiased areal sampling with nearest neighbor methods. Unpublished Report. Presented at the 2012 Northeast Mensurationists Meeting, The Pennsylvania State University, State College, PA. On file with Dr. Mark J. Ducey, Department of Natural Resources and the Environment, University of New Hampshire, 215 James Hall, Durham, NH 03824. 24 p.

R Core Team. 2014. R: A language and environment for statistical computing. R foundation for statistical computing. Vienna, Austria. http://www.R-project.org. [Last accessed: Date Unknown].

DETECTION OF SEVERE STORM SIGNATURES IN LOBLOLLY PINE USING SEVEN-YEAR PERIODIC STANDARDIZED AVERAGES AND STANDARD DEVIATIONS

Stevenson, Douglas; Thomas Hennessey, Thomas Lynch, Giulia Caterina, Rodolfo Mota, Robert Heineman, Randal Holeman, Dennis Wilson and Keith Anderson[1]

A loblolly pine plantation near Eagletown, Oklahoma was used to test standardized tree ring widths in detecting snow and ice storms. Widths of two rings immediately following suspected storms were standardized against widths of seven rings following the storm (Stan1 and Stan2). Values of Stan1 less than -0.900 predict a severe (usually ice) storm when Stan2 is less than -0.290. Values of Stan1 between 0.000 and -0.900 and Stan2 between 0.000 and -0.290 indicate a mild (usually snow) storm.

In 1998 a growth study was established in a loblolly pine stand planted in 1975. Height and diameter data was collected from 18 fifth-acre plots in January 1999 and again in January 2000. The Christmas 2000 ice storm heavily damaged the stand. Increment cores were collected in February 2002 and heights and diameters again measured in January 2003 and March 2004. In February 2005 new height and diameter measurements were made and a second set of increment cores collected. The stand was harvested in 2005.

Total ring width series were made (Stevenson and others 2015) using COFECHA (Holmes 1983) and converted to chronologies using ARSTAN (Cook and Holmes 1986) with a logarithmic decay and straight-line-of-negative-slope as the first and second detrending models respectively, division option. A seven-year running average and standard deviation were calculated for each year and the six years following it from 1985 to 1998.

We created a variable, Stan1, by standardizing the ring width from each growing season using the computed seven-year averages and standard deviations. We then created a second variable , Stan2, by substituting the ring width of the second growing season in place of that from the first season.

Standardization recalculates the mean and standard deviation every year. Stan1 and Stan2 are standardized, resetting them to a new base that reflects the stand's reduced ability to respond to new injuries. By requiring maximum standardized ring width values in both the first and second year following the storm, we separate a severe winter storm from an ordinary winter storm.

Ice storms always produced threshold values of Stan1 less than -0.900 (Stevenson and others 2015). Unfortunately, so did some years that had no ice storms. Likewise, ice storms produced threshold values of Stan2 of less than -0.290, but so did some years that had no ice storms. In the Eagletown data, Stan1 and Stan2 both fell below their respective thresholds in 1985, 1995 and 1996. The Stan1 value for 1985 was -1.181 with a Stan2 value of -0.298. The historical record (NCDC Feb 1987 - Feb 1997) showed severe storms in 1985 and 1995, but none in 1996. The Christmas 2000 ice storm was too recent for Stan1 and Stan2 to be calculated.

Storms in 1987, 1988, 1993, 1995 and 1997 all exceeded at least one of the threshold values. It is unknown whether the storms of 1987 and 1988 were snow or ice storms, but those of 1993, 1995 and 1997 were all snow storms with little ice accumulation. Records from a nearby on-going growth-and-yield study (Saud and others 2015) showed no stem breakage in any of these years, indicating no ice damage, but leaving open the possibility of snow damage. NCDC records for 1993, 1995 and 1997 (NCDC Feb 1987 - Feb 1997) indicate mostly snow with little ice.

Our method predicted two severe storms (1985 and 1995) that were confirmed by weather records and one (1996) that was not. In addition, it detected five moderate storms (1986, 1987, 1988, 1993 and 1997),

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probably snow storms. This is a large number of storms and may be due either to species or age differences. Determining which may allow the effects of tree/stand age to be taken into account when examining tree ring series for climate and weather information.

1. Our method of detecting severe winter storms in tree ring widths works for loblolly pine.
2. The large number of storms indicated in loblolly pine may indicate a greater susceptibility to ice and snow damage among young loblolly stands.

LITERATURE CITED

Cook, E.R.; Holmes, R.L. 1986. Users manual for program ARSTAN. Laboratory for Tree-Ring Research. University of Arizona, Tucson, AZ. [Not paged].

Holmes, R.L. 1983. Program COFECHA user's manual. Laboratory of Tree-Ring Research. University of Arizona, Tucson, AZ: 41-49.

National Climate Data Center (NCDC). 1985-96. Storm data and unusual weather phenomena. Records archived at National Climate Data Center, National Oceanic and Atmospheric Administration, Boulder, CO. Monthly.

Saud, P.; Lynch, T.; Wilson, D. [and others]. 2015. Influence of weather and climate variables on the basal area growth of individual shortleaf pine trees. In: Holley, A., Conner, K.; Haywood, J. eds. Proceedings of the 17th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-203. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 406-408.

Stevenson, D.; Hennessey, T.; Lynch, T. [and others]. 2015. Eagletown loblolly pine chronology – PITA – ITRDB OK039. Boulder, CO: Data archived at World Data Center for Paleoclimatology. [Date accessed: September 2, 2015].

PERFORMANCE OF CONTAINER-GROWN SEEDLINGS OF AMERICAN CHESTNUT BACKCROSS HYBRIDS BC_3F_3 GENERATION IN CENTRAL LOUISIANA

Shi-Jean S. Sung, Stacy L. Clark, Scott Schlarbaum, Daniel Dey and Daniel J. Leduc[1]

Abstract—Seedlings from two families of the BC_3F_3 backcross generation of the American chestnut (*Castanea dentata*) and Chinese chestnut (*C. mollissima*) were cultured in 2013 in Missouri using the Root Production Method®, a container-based system used to avoid disease problems associated with bareroot nursery production. After culling part of the seedlings at initial leaf formation in a greenhouse, both the culled and the remaining "kept" seedlings were transplanted into containers of 1.33, 1.57, and 2.62 L and grown outdoors. Seedlings, remaining in their containers, were brought to central Louisiana in mid-June 2013 and cultured outdoors. Half of the seedlings were harvested in September, and the other half continued to grow until storage in a cold room in mid-November. Of the seedlings harvested in September, family W4938 (W) seedlings had 90 percent more biomass than family D3862 (D) seedlings. Patterns of biomass allocation from each component within a seedling did not differ between families. All but one seedling had short (less than 6 cm) taproots, whereas the lengths of first-order lateral roots and adventitious roots were similar to the depths of the containers used. In family W, seedlings cultured in the 2.62 L containers were larger in size and biomass than those of the smaller volumes. Seedlings were outplanted in central Louisiana in late March, 2014. First-year survival was only 47 and 33 percent for the W and D seedlings, respectively. High summer temperature and local wet planting spots are suspected of causing the high seedling mortality.

INTRODUCTION

The chestnut blight fungus (*Cryphonectria parasitica* Barr) killed almost all mature American chestnut (*Castanea dentata* Borkh) trees throughout the species' native range in the first half of the 20th century. Today, young trees originating from tree stumps or suppressed seedlings established before the arrival of blight typically do not grow much more than 15 m in height before eventually succumbing to the blight (Paillet 2002). The restoration strategy used by the American Chestnut Foundation is a backcross breeding technique designed to produce trees with the American (timber type) genes while maintaining blight resistant genes from the Chinese (*C. mollissima* Blume) (orchard type) (Burnham and others 1986). In theory, the third generation of the third backcross (BC_3F_3) should have the desired growth habit of the American (94 percent American) and the blight resistance of the Chinese (6 percent Chinese) chestnuts (Hebard 2011).

The artificial regeneration of the blight resistant American chestnut hybrids has been the main approach used in recent years to restore this species to its native range. Clark and colleagues reported the nursery cultural protocol, seedling quality, and field performance of backcross chestnut seedlings (Clark and others 2009, 2012a, 2012b). For example, Clark and others (2012a) reported a 3-cm height increase in bareroot stock of chestnut seedlings for every 1-g increase in the mean nut weight. Thus, the overall nursery seedling quality could be improved by culling small nuts. Pinchot and others (2015) used individual nut weight to predict chestnut seedling quality and found a 1-g increase in nut weight resulted in increases of 6 cm in height and 0.5 mm in root collar diameter, and one additional first-order lateral root.

Between 2009 and 2011, Clark and others (2014) outplanted over 4000 1-0 bareroot seedlings of various generations on several National Forests in Virginia, North Carolina, and Tennessee. They found two 2010 plantings and six 2011 plantings were succumbing to a root rot disease (*Phytophthora cinnamomi* Rands). This fungus was present in the bareroot nursery and in some of the planting sites. *Phytophthora* root rot is thought to have contributed to chestnut dieback prior to the arrival

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of chestnut blight (Anagnostakis 2012). Rhoades and others (2003) cautioned that it is crucial to recognize and avoid sites where soil physical factors such as wetness and compactness promote *Phytophthora* root rot. Growing seedlings in containers with sterile media will reduce the risk of planting root rot-infected bareroot seedlings in the field and will hopefully improve the success of restoring American chestnut in the forests. The objective of this study was to examine growth performance, including root system morphology, of the backcross American chestnut seedlings cultured in containers of various dimensions. Because the native range of this species does not include Louisiana, this study was limited to one year after outplanting.

MATERIALS AND METHODS

Seedling Culture

Nuts from two families of the BC_3F_3 generation, D3862 (D) and W4938 (W), came from the open-pollinated B_3F_2 seed orchard of the American Chestnut Foundation in Meadowview, VA (Hebard 2011). Progenies were putative half siblings. Seedlings were grown at the Forrest Keeling Nursery (Elsberry, MO) using a modified version of their Root Production Method® in 2013. This method is designed to produce root systems consisting of many adventitious roots stemming from a shortened taproot. Nuts were sown into 3-cm deep mesh-bottomed trays in a cold room in December of 2012 and then moved to a greenhouse in March 2014 to initiate germination. The taproot was air pruned at around 3 cm, consistent with the depth of the germination tray. The Root Production Method® typically involves culling conducted at the initial leaf formation stage, based on size of shoot, to discard the smallest seedlings. However, we wanted to test the effects of this culling treatment on American chestnut, and seedlings were divided into two groups, "kept" or "culled." After the culling treatments, the germinant seedlings were transplanted into mesh-bottomed, open drainage, square containers of three volumes, namely, small (7.6 cm/22.9 cm/1.32 L, width/depth/volume), medium (10.2 cm/15.2 cm/1.58 L), and large (10.2 cm/25.4 cm/2.64 L). Depth of the small containers is greater than that of the medium containers. Roots were naturally air pruned. The growth media (decayed rice hulls, sand, and pine bark) were amended with time-release fertilizer. Seedlings were cultured outdoors for four weeks after transplanting. Due to low germination rates, only 18 D and 37 W seedlings were transported in containers to the Alexandria Forestry Center in Pineville, LA, in mid-June 2013. Seedlings continued to grow outdoors. In late June, some seedlings began to show signs of heat damage, with brown discoloration and curling of the leaf edge. Two layers of 30 percent shade cloth were applied across the iron frame (3-m tall) over the seedlings to lower the temperature and light intensity.

Seedling Assessment

Growth and photosynthesis—In mid-July, 8 D and 16 W seedlings were randomly selected for photosynthesis measurements with a LiCor® 6400 portable, open-system infrared gas analyzer (LiCor®, Lincoln, NE). Measurements were made between 9 a.m. and 11 a.m. and again between 1 p.m. and 3 p.m. on the same day on the most recently fully expanded leaves. Photosynthetic active radiation was set between 1400 and 1600 $\mu E\ m^{-2}\ s^{-1}$ with a red-blue light source and the CO_2 level for the reference chamber was 400 ppm. The middle section of the attached leaf was enclosed in the measuring chamber (2 x 3 cm). After each photosynthesis measurement, a hole punch was used to obtain a sample of 2.85 cm^2 from the same leaf for chlorophyll analysis. The leaf samples were stored on ice and transported to the laboratory. Chlorophylls a and b were extracted with N,N-dimethylformamide and the absorbance of the extract was read at 664 nm and 647 nm as described in detail by Sung and others (2010). Seedling height and root collar diameter were also measured. Half of the seedlings were harvested in mid-September for biomass and root system morphology. The remaining seedlings were stored in a cold room from December until being outplanted in March 2014.

Root system morphology—Parameters of the root system morphology assessed included the number of first-order lateral roots (FOLR) and adventitious roots (ADV), and lengths and dry weights of taproots, FOLR, and ADV. An FOLR was defined as a lateral root that originated from the taproot, sturdy in structure, and with a diameter, measured 1 cm away from the taproot, of at least 1.5 mm (for D) or 2.0 mm (for W). The ADV initiated near the air-pruned end of the taproot.

Outplanting

The field experiment site is on an open field located on the Palustris Experimental Forest within the Kisatchie National Forest in Rapides Parish of central Louisiana (31°11', -92°41'). The soil is a moderately well-drained, gently sloping Beauregard silt loam (fine silty, siliceous, superactive, thermic, Plinthaquic Paludults). A power auger with a 10-cm bit was used to drill holes large enough to accommodate roots of seedlings cultured in the largest containers. Soil from the holes was used to fill the hole after planting. Seedling mortality was monitored quarterly.

Statistical Analysis

The original study was designed as a 2 by 2 by 3 factorial with 2 nut sizes (large and small) by 2 seedling grades (kept and culled) by 3 container sizes (small, medium, and large) for each of the two backcross families. However, due to low germination rates, only 18 D and 37 W seedlings were used in the analysis.

Seedlings in the D family came from 4 of the original 12 treatments, namely, nut size (large or small) in combination with 2 container sizes (large or medium). All D seedlings were in the kept category. For the W seedlings, each of the 12 treatments had at least 2 seedlings except the small nut, culled, medium-container treatment which had only 1 seedling.

Seedlings from all 12 treatments were combined and comparisons made between the two backcross families for growth, photosynthesis rate, chlorophyll content, and root system morphology. All comparisons between two families were tested for differences at the $\alpha = 0.05$ level using PROC GLM from the SAS Institute (SAS 2004). In order to have a valid statistical test based on the analysis of variance (ANOVA), it is assumed that the data are independent and normally distributed, and that the variance is equal in each group. Because the seedlings were randomly arranged during growth, independence is assumed. All residuals were checked for normality and homogeneity. Often, the assumption of normality was violated, and the assumption of homogeneity was violated occasionally. In cases where either assumption was violated, the data were reanalyzed using PROC NPAR1WAY (SAS 2004) with the Wilcoxon option, but this did not change any results.

Similarly, container size was tested for differences at the $\alpha = 0.05$ level using PROC GLM. Only the W seedlings, which came from containers of all three sizes, were used for the analysis. All residuals were checked for normality and equality. Because the sample sizes were nearly equal, ANOVA is robust to unequal variance. Often, the assumption of normality was violated, but the statistical power lost by this was without effect because the tests that had non-normal residuals either had significant differences or were not close to significance.

RESULTS AND DISCUSSION

Most chestnut seedlings had set a terminal bud and completed their height growth by early August (table 1). This was at least one month sooner in growth termination than the reported height growth termination by bareroot chestnut seedlings grown in Delano, TN (Pinchot and others 2015), probably due to the summer heat in our area. Both W and D family seedlings had at least 40 percent of their leaves suffering from heat damage by late August. Nevertheless, the W seedlings were 72 percent taller than the D seedlings (table 1). The W seedlings harvested in September weighed almost twice as much as the D seedlings (table 1). However, patterns of biomass allocation by each component within a seedling did not differ between the two families.

Mean morning and afternoon photosynthesis rates measured in mid-July were 13.1 and 11.6 µmol m^{-2} s^{-1}, respectively, for the D seedlings and 12.5 and 10.4 µmol m^{-2} s^{-1}, respectively, for the W seedlings. The higher

Table 1—Mean seedling parameters for Root Production Method® container-grown chestnuts from two BC_3F_3 generation families, D3862 and W4938, harvested in September 2013. Height was measured on all seedlings

Parameter[a]	D3862	W4938
Height (cm)		
July 11*	33.0	53.2
July 26*	34.0	57.6
Aug. 8*	34.5	59.4
Aug. 26*	34.2	59.0
Sept. 11*	33.5	58.1
Root collar diameter (mm) Sept. 11	7.8	8.4
Healthy leaves >5 cm long (#) Aug. 26	6.9	11.1
Damaged leaves >5 cm long (#) Aug. 26*	4.9	9.4
Seedling dry weight (g)*	13.0	24.8
Leaf dry weight (g)	4.5 (34.6)[b]	7.7 (31.0)
Stem and branch dry weight (g)	3.9 (30.0)	9.4 (37.9)
Root system dry weight (g)	4.6 (35.4)	7.7 (31.0)

[a]Parameters associated with * have Pr > F values of <0.05.
[b]Numbers in parentheses are percent of biomass allocation within a seedling.

photosynthetic rates measured in this study compared to those of potted American chestnut seedlings grown under full sun (Wang and others 2006) suggested that the two layers of 30 percent shade cloth used in this study did not negatively affect seedling photosynthesis or growth. The chlorophyll a+b content for the D and W seedlings were 60.3 and 55.2 nmol cm^{-2}, respectively. The chlorophyll a-to-b ratio was similar between families (3.70 for D and 3.78 for W). Neither photosynthetic rate nor chlorophyll content of the D seedlings differed from those of the W seedlings. Lower numbers of both healthy and heat-damaged leaves in D than in W may be the main reason for less height growth and biomass accumulation for D seedlings (table 1). Compared to the values of height and root collar diameter reported for bareroot nursery seedlings (Clark and others 2012b, Pinchot and others 2015), container-grown seedlings in this study were much smaller. Family, cultural practices (container versus bareroot), and high temperature and humidity growth conditions in central Louisiana may have contributed to the smaller seedlings in this study.

We used a lower lateral root diameter limit for FOLR in the D seedlings than the W seedlings (1.5 versus 2.0 mm) for three reasons. First, D seedlings were much smaller than W. Second, using a higher diameter limit would render some of the D seedlings without any FOLR, but these seedlings did have some lateral roots with sturdy structure and a diameter ranging between 1.5 and 2.0 mm. Third, the W seedlings' lateral roots with a diameter between 1.5 and 2.0 mm usually were not sturdy in structure. Although not statistically significant, the percentage of biomass allocated to the non-FOLR group was less for the W seedlings, suggesting that only a few potential FOLR, based on the 1.5-mm criterion, were placed in the non-FOLR group with the 2.0-mm criterion. Greater numbers of FOLR have long been associated with larger seedling sizes in pines (*Pinus* spp.), oaks (*Quercus* spp.), and the American chestnut (Kormanik and others 1990, 1998a, 1998b, Pinchot and others 2015). In this study, the greater seedling size of the W seedlings (table 1) cannot be explained by their FOLR number. However, if the 2.0-mm limit had been applied to the D seedlings, their mean FOLR number would have reduced from 4.9 to 2.4, and this difference may have been significant. The W seedlings also had greater ADV diameter than the D seedlings (table 2). The rest of the root system parameters assessed did not differ between families.

Taproots in all but one seedling were shorter than 6 cm (table 2). A W seedling cultured in a large container had a curved taproot of 25 cm and was excluded from the analysis. Otherwise, taproot lengths were similar between the two families. The short taproots and the presence of adventitious roots (table 2, fig. 1) that formed near the end of the taproot met one of the objectives of the modified Root Production Method® used. The taproot was air-pruned in the 3-cm tray prior to culling and transplanting into individual containers. However, 32 percent of the seedlings assessed had taproots longer than 3.5 cm and had more than one curve along their length. It is probable that these taproots extended horizontally in the tray before reaching the tray bottom and eventually being air pruned. It would be interesting to find out if growing chestnut seedlings in deeper germination trays increases the number of FOLR, decreases the number of ADV, or both. The Root Production Method® is in contrast to the general practices of container nurseries. Tree seedlings that have been cultured in containers from seeds without being transplanted in the middle of the culture period generally have taproot lengths consistent with their container depths. For example, Sung and others (2009) reported that 58, 36, and 6 percent of FOLR originated from the top 5 cm, middle 5 cm, and bottom 4.4 cm of taproots, respectively, in container-grown longleaf pine (*P. palustris* Mill). In other words, taproot lengths for most longleaf pine seedlings cultured in 12- to 15-cm deep containers were greater than 10 cm.

Lengths of FOLR and ADV were similar between the two families. Furthermore, lengths for FOLR and ADV were closely associated with container depths. For example, lengths of FOLR from seedlings grown in small (23-cm deep), medium (15-cm deep), and large (25-cm deep) containers were 15.9, 13.1, and 23.0 cm, respectively. Similarly, lengths of ADV were 17.1, 12.0, and 22.4 cm for the seedlings grown in small, medium, and large volume containers, respectively. Some FOLR and ADV were longer than the container depths, as shown by the ranges in table 2, indicating root spiraling inside the containers (fig. 1).

Effects of container size on W seedling size and root system are presented in table 3. The large-container seedlings were larger in all growth parameters than the small-container seedlings except for leaf dry weight and a few root system parameters. The medium-container seedlings had values similar to those of the small-container seedlings for most parameters assessed. Container size did not affect seedling photosynthetic rate (a.m. or p.m.), chlorophyll a+b content, or chlorophyll a-to-b ratio. Taproot length, FOLR diameter, ADV number and diameter, and the FOLR and ADV origination points on the taproot were also unaffected (table 3). As shown previously, lengths of FOLR and ADV were significantly greater for the large-container seedlings than for the rest of the seedlings. Since the small containers are deeper than the medium containers, the small-container seedlings had greater

Table 2—Mean root system morphology of Root Production Method® container-grown chestnut seedlings from BC_3F_3 generation families, D3862 and W4938, harvested in September 2013

Parameters[a]	D3862	W4938
Taproot dry weight (g)	1.3 (28.3%)[b]	1.9 (25.0%)
FOLR[c] dry weight (g)	1.0 (21.7%)	2.3 (29.0%)
Non-FOLR[d] dry weight (g)	0.6 (13.0%)	0.6 (7.9%)
ADV[e] dry weight (g)	1.7 (37.0%)	2.9 (38.2%)
FOLR (#)	4.9 (1-17)	5.9 (1-14)
ADV (#)	3.7 (1-10)	4.7 (2-8)
Taproot length (cm)	2.8 (1.3-4.2)	3.2 (1.3-5.5)[f]
FOLR diameter (mm)*	2.0 (1.5-4.8)[g]	3.4 (2.0-6.7)
ADV diameter (mm)*	3.1 (1.7-6.4)	4.2 (2.0-8.4)
FOLR length (cm)	18.4 (4.1-28.7)	16.7 (1.9-29.3)
ADV length (cm)	17.9 (3.2-26.1)	16.5 (3.9-30.2)
FOLR origination point on taproot (cm)	1.7 (0.6-3.3)	1.6 (0.1-4.2)
ADV origination point on taproot (cm)	2.4 (0.2-3.8)	2.6 (0.3-4.5)

[a] Parameters associated with * have Pr > F values of <0.05.
[b] Numbers in parentheses are percent of biomass allocation within the seedling root system or range for a given parameter.
[c] Lateral roots originated from the taproot and with a diameter at least 1.5 and 2.0 mm for D and W seedlings, respectively.
[d] Lateral roots originated from the taproot but not sturdy or large enough to be counted as FOLR.
[e] Adventitious roots initiated near the taproot end.
[f] One large container-grown seedling with a 25-cm long taproot was excluded.
[g] Range in this section covered all individual roots in each family.



Figure 1—(L) Root system of a small nut, culled, large-container W4938 seedling of the BC_3F_3 backcross generation. It had a 1.3-cm long taproot, 1 first-order lateral root, and 5 adventitious roots with one of them (arrow) spiraling. (R) Root system of a large nut, kept, medium-container W4938 seedling pictured before the bottom root plug media was rinsed off for root assessment. It had a 2.2-cm long taproot, 8 first-order lateral roots, and 6 adventitious roots located near the taproot end.

Table 3—Effects of container size on chestnut seedling parameters of the BC_3F_3 generation family W4938 harvested in September 2013. Height was measured on all seedlings

	Container Volume		
Parameters	Large	Medium	Small
Height (cm)			
July 11	59.7A	52.6B	47.2B
July 26	68.6A	55.1B	48.9B
Aug. 8	73.0A	55.3B	49.3B
Aug. 26	73.1A	55.0B	48.5B
Sept. 11	72.0A	53.7B	48.3B
Root collar diameter (mm) Sept. 11	9.5A	8.1B	7.6B
Healthy leaves >5 cm long (#) Aug. 26	15.3A	9.5AB	7.3B
Damaged leaves >5 cm long (#) Aug. 26	12.3A	6.7AB	9.1B
Seedling dry weight (g)	38.6A	20.4B	21.5B
Leaf dry weight (g)	11.3A	6.1A	7.0A
Stem and branch dry weight (g)	15.8A	7.5B	7.5B
Root system (g)	11.4A	6.8B	6.9B
TAP dry weight (g)	2.8A	2.0AB	1.4B
FOLR dry weight (g)	3.6A	2.3AB	1.8B
Non-FOLR dry weight (g)	0.7A	0.7A	0.4A
ADV dry weight (g)	4.4A	1.7B	3.3AB
FOLR (#)	6.6AB	8.8A	4.1B
ADV (#)	5.8A	4.0A	4.9A
FOLR diameter (mm)	3.8A	3.1A	3.8A
ADV diameter (mm)	4.8A	3.8A	4.4A
TAP length (cm)	3.0A	3.7A	2.5A
FOLR length (cm)	22.8A	13.0B	16.6B
ADV length (cm)	22.8A	12.4C	17.1B
FOLR origination point on TAP (cm)	1.8A	1.6A	1.4A
ADV origination point on TAP (cm)	2.5A	3.0A	2.1A

[a]One large container-grown seedling with a 25-cm long taproot was excluded.

[b]Numbers followed by the same letter are not different significantly at α = 0.05 by Duncan's multiple range test.

lengths in FOLR and ADV compared to the medium-container seedlings.

Nine D (mean height 36 cm) and 15 W (mean height 59 cm) seedlings were stored in a cold room from December 2013 until outplanted to a site at the Palustris Experimental Forest in central Louisiana in late March 2014. Most seedlings had some fully expanded leaves by early May. However, by September, only 47 and 33 percent of the W and D seedlings, respectively, had survived. Although the topographic change was not visually evident, all seedlings planted in lower-lying areas died. Apparently, the hot and humid conditions in central Louisiana were not conducive to the normal growth of the American chestnut hybrid seedlings.

CONCLUSIONS

The idea of planting container-grown, blight-resistant chestnut seedlings to reduce the risk of *Phytophthora* root rot infection and to enhance the artificial regeneration of the American chestnuts in the forests was sound. Although the environmental conditions in central Louisiana were not conducive to the growth and survival of the American chestnuts in the forest, culturing seedlings under shade cloth for the first growing season still provided some vital information regarding the Root Production Method® for American chestnut, which, to our knowledge, had never before been used. The W4938 seedlings were larger than the D3862 seedlings. Seedlings cultured in large containers (2.64 L in volume) had greater values in size and various root system parameters than seedlings from medium (1.58 L) and small (1.32 L) containers. Seedlings produced had short taproots and several adventitious roots, as expected with the Root Production Method®. Furthermore, lengths of the first-order lateral roots and adventitious roots were consistent with the depths of the containers. To validate the benefits of culturing American chestnut seedlings using this method, a greater number of families and more nuts in each family are needed for future testing.

LITERATURE CITED

Anagnostakis, S.L. 2012. Chestnut breeding in the United States for disease and insect resistance. Plant Disease. 96: 1392-1403.

Burnham, C.R.; Rutter, P.A.; French, D.W. 1986. Breeding blight-resistant chestnuts. Plant Breeding Reviews. 4: 347-397.

Clark, S.L.; Schweitzer, C.J.; Schlarbaum, S.E. [and others]. 2009. Nursery quality and first-year response of American chestnut (*Castanea dentata*) seedlings planted in the southeastern United States. Tree Planters' Notes. 53: 13-21.

Clark, S.; McNab, H.; Loftis, D. [and others]. 2012a. American chestnut growth and survival five years after planting in two silvicultural treatments in the southern Appalachians. Forests. 3: 1017-1034.

Clark, S.L.; Schlarbaum, S.E.; Saxton, A.M. [and others]. 2012b. Nursery performance of American and Chinese chestnuts and backcross generations in commercial tree nurseries. Forestry. 85: 589-600.

Clark, S.L.; Schlarbaum, S.E.; Saxton, A.M. [and others]. 2014. The first research plantings of third-generation, third-backcross American chestnut (*Castanea dentata*) in the Southeastern United States. In: Double, M.L.; MacDonald, W.L., eds. Proceedings of the 5th international chestnut symposium, ISHS. Acta Horticulturae. 1019: 39-44.

Hebard, F.V. 2011. The American chestnut foundation breeding program. In: Sniezko R.A.; Yanchuk, A.D.; Kliejunas, J.T. [and others]. tech. coords. Proceedings of the 4th international workshop on the genetics of host-parasite interactions in forestry: disease and insect resistance in forest trees. Gen. Tech. Rep. PSW-GTR-240. Albany, CA: U.S. Department of Agriculture Forest Service, Pacific Southwest Research Station: 221-234.

Kormanik, P.P.; Ruehle, J.L.; Muse, H.D. 1990. Frequency distribution and heritability of first-order lateral roots in loblolly pine seedlings. Forest Science. 36: 802-814.

Kormanik, P.P.; Sung, S.S.; Kormanik, T.L. [and others]. 1998a. Effect of acorn size on development of northern red oak 1–0 seedlings. Canadian Journal of Forest Research. 28: 1805-1813.

Kormanik, P.P.; Sung, S.S.; Zarnoch, S.J. 1998b. Immature loblolly pine growth and biomass accumulation: Correlations with seedlings initial first-order lateral roots. Southern Journal of Applied Forestry. 22: 117-123.

Paillet, F.L. 2002. Chestnut: history and ecology of a transformed species. Journal of Biogeography. 29: 1517-1530.

Pinchot, C.C.; Clark, S.L.; Schlarbaum, S.E. [and others]. 2015. Effects of temporal dynamics, nut weight and nut size on growth of American chestnut, Chinese chestnut and backcross generations in a commercial nursery. Forests. 6: 1537-1556.

Rhoades, C.C.; Brosi, S.L.; Dattilo, A.J. [and others]. 2003. Effect of soil compaction and moisture on incidence of phytophthora root rot on American chestnuts (*Castanea dentata*) seedlings. Forest Ecology and Management. 184: 47-54.

SAS Institute Inc. 2004. SAS/STAT® 9.1 User's Guide. Cary, NC: SAS Institute Inc.

Sung, S.S.; Haywood, J.D.; Zarnoch, S. [and others]. 2009. Long-term container effects on root system architecture of longleaf pine. In: Proceedings national convention. Orlando, FL: Society of American Foresters. 9 p.

Sung, S.S.; Haywood, J.D.; Sword-Sayer, M.A. [and others]. 2010. Effects of container cavity size and copper coating on field performance of container-grown longleaf pine seedlings. In: Stanturf, J.A., ed. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-121. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 241-245.

Wang, G.G.; Bauerle, W.L.; Mudder, B.T. 2006. Effects of light acclimation on the photosynthesis, growth, and biomass allocation in American chestnut (*Castanea dentata*) seedlings. Forest Ecology and Management. 226: 173-180.

SHORTLEAF PINE *(PINUS ECHINATA* MILL.) AND HARDWOOD REGENERATION AFTER THINNING NATURAL SHORTLEAF PINE FORESTS IN SOUTHERN UNITED STATES

Anup KC, Thomas B. Lynch, and James M. Guldin[1]

Abstract—Understory pine and hardwood regeneration in the Ozark and Ouachita National Forests were measured in 1995 for the first time following thinning and hardwood control at plot establishment 1985-87. Red maple (*Acer rubrum*), shortleaf pine and flowering dogwood (*Cornus florida*) were the most frequently recorded species. Understory shortleaf pine stems have declined consistently since 1995. The binomial logistic regression model showed that site index and overstory shortleaf pine basal area were negatively related to shortleaf pine regeneration ($P<0.05$). Overstory shortleaf pine thinning may initiate shortleaf pine regeneration at early stage; however, newly regenerated hardwood species will dominate the entire regeneration process long-term, if hardwood control treatments are not applied.

INTRODUCTION

Mixed shortleaf pine hardwood forests are common in southern United States (Kabrick and others 2011, Moser and others 2006). Previous studies have suggested that the overstory shortleaf pine stand area is declining (Moser and others 2006; Lilly and others 2012). At the same time, little is known about the long term regeneration pattern of the understory vegetation in naturally occurring shortleaf pine forests; especially after thinning the overstory shortleaf pine. The purpose of the present study was to evaluate the temporal pattern of regeneration of understory pine and hardwoods after thinning the overstory shortleaf pine. We also predicted the probability of achieving adequate shortleaf pine regeneration after thinning using overstory and understory characteristics.

MATERIALS AND METHODS

Study plot locations were ranged from the Ozark and Ouachita National Forests. There were total of 182 permanent circular study plots 0.2 acres in area. Data were collected approximately every five years. Four circular subplots 0.005 acre area were created inside the 0.2 acre plots, and only two of these were measured in 1996. Understory hardwood trees greater than one inch in diameter at breast height were removed while establishing the plots in 1985-1986. Woody plants greater than breast height were measured beginning in 1995. Species richness, percent density, mean stem density, and relative dominance were calculated. The binomial logistic regression model was used to predict the probability of shortleaf pine regeneration approximately 30 years after thinning.

RESULTS AND DISCUSSION

A total of 68 woody-vegetation species were recorded throughout the measurement periods. The lowest and the highest number of species were recorded in 1996 and 2006 respectively. Red maple was the major species (12.42±3.79 percent in 1996, 15.46±2.93 percent in 2001, 12.47±2.48 percent in 2006 and 22.68±3.79 percent in 2013) in terms of relative density in Ozark National Forest. Flowering dogwood (11.75±1.70 percent in 1996), shortleaf pine (8.56±1.57 percent in 2001) and blackjack oak (9.32±1.63 percent in 2006 and 10.72±1.60 percent in 2013) were the major species in Ouachita National Forest.

Binomial logistic regression model showed probability of getting shortleaf pine stems greater than 500 per acre decreased by 6 percent with each unit increase in site index. Similarly, chances of getting shortleaf pine stems greater than 500 stems per acre decreased by 2 percent with each unit increase in overstory basal area. Results from the present study were consistent with previous findings (Lynch and others 2002).

Results showed red maple and several oak species exceeded shortleaf pine regeneration in recent years. Therefore, control of hardwoods is required to achieve adequate shortleaf pine regeneration. Our results suggested one time thinning of shortleaf pine may not

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be sufficient to obtain the desired amount of shortleaf pine regeneration. Other forest management practices such as controlled burning may be required to achieve the desired shortleaf pine regeneration for long term management.

LITERATURE CITED

Kabrick, J.M.; Dey, D.C.; Shifley, S.R.; Villwock, J.L. 2011. Early survival and growth of planted shortleaf pine seedlings as a function of initial size and overstory stocking. In: Fei, S.; Lhotka, J.M.; Stringer, J.W. [and others]. eds. Proceedings of the 17th central hardwood forest conference. Gen. Tech. Rep. NRS-P-78. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 277-286.

Lilly, C.J., Will, R.E., Tauer, C.G., [and others]. 2012. Factors affecting the sprouting of shortleaf pine rootstock following prescribed fire. Forest Ecology and Management. 265: 13-19.

Lynch, T.B.; Nkouka, J.; Huebschmann, M.M.; Guldin, J.M. 2002. Estimating the probability of achieving shortleaf pine regeneration at variable specified levels. In: Outcalt, K.W., ed. Proceedings of the 11th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-48. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 473-474.

Moser, W.K.; Hansen, M.; McWilliams, W.H.; Sheffield, R.M., 2006. Shortleaf pine composition and structure in the United States. In: Kabrick, J.M., Dey, D.C., Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 19-27.

TWENTY FIVE YEARS LONG SURVIVAL ANALYSIS OF AN INDIVIDUAL SHORTLEAF PINE TREES

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A semi parametric cox proportion hazard model is preferred when censored data and survival time information is available (Kleinbaum and Klein 1996; Alison 2010). Censored data are observations that have incomplete information related to survival time or event time of interest. In repeated forest measurements, usually observations are either right censored or interval censored. Interval censoring occurs if the exact year of tree death is unknown, since measurement periods are typically longer than one year. Right censoring occurs if tree has not died at the end of the study.

Site index is generally assumed to be static over time while variables such as diameter, basal area, and crown ratio change over time are termed time varying covariates in the Cox model. The extended Cox model does not requires the assumption of proportional hazard and is appropriate to model time dependent nature of mortality with censoring of observations (Fisher and Linn 1999; Hosmer and others 2008). This model tests covariates for their significance with regard to individual tree survival. Thus we used the Cox extended model to identify effects of prognostic and protective variables on the survival of shortleaf pine for two data sets; 1) No ice damage: plots excluded having an ice damage event, and 2) all plots after an ice damage event (ice damage occurred during the fourth measurement).

A growth study of natural stands of shortleaf pine (*Pinus Echinata* Mill.) that was measured 6 times from 1985 to 2014 provided an opportunity to investigate the influence of time dependent covariates on the survival of individual trees. Over 200 permanent plots located in naturally occurring shortleaf pine forests on the Ozark and Ouachita National Forests were measured every 4 to 7 years. For details see Lynch and others (1999). The first measurement was conducted in 1985-1987, and the last (sixth) measurement occurred in between 2012-2014. The total sample included 208 plots that were 0.08 hectare (1/5th of an acre) in size. An ice storm in 2000 (during the 4th measurement) caused considerable damage on 111 of these plots.

Cox model is widely used (Fisher and Linn 1999; Alison 2010) in the biomedical field with an assumption: a) a study starts at t_0 and individual clinic visits occur at intervals where covariates (e.g. blood pressure, body weight) are recorded at each visit, b) an event occurs (e.g. glaucoma) between the visits, and exact time is unknown (interval censoring), and c) subjects are right censored and/or interval censored. Similar, assumption were made for tree mortality data while modelling Cox time dependent model. The standard Cox model (Eq. 1) assumes that hazard ratio (Hzr) is proportional between groups, while extended Cox model (time dependent model) (Eq. 2) assumes that Hzr is not proportional and depends upon time (Fisher and Linn 1999; Hosmer and others. 2008). The Hzr is the ratio of the hazard rate in the one group versus another group. The hazard rate is the probability that the event of interest will occur in the next time interval, divided by the length of that interval.

$$h(t) = h_0(t)\exp\left(\sum_{i=1}^{K}\beta_i x_i\right) \quad (1)$$

$$h(t,x) = h_0(t)\exp\left(\sum_{i=1}^{K}\beta_i x_i + \sum_{i=1}^{k}\gamma_i x_i g_i(t)\right) \quad (2)$$

where *h(t)* is hazard function at time *t*, $h_0(t)$ is base line hazard function, *h(t, x)* is hazard function for *x* covariate that depends on time *t*, β_i is the parameter to be estimated associated with variables, x_i is number of variables, γ_i is a parameter associated the variable x_i of the time dependent function ($g_i(t)$); t is the upper time interval in month for the period. We tested to see if = 0; if = 0, then the model was proportional hazard model, otherwise it was a time dependent model.

The mortality rate for measurement periods was an average of 4 percent with no ice storm, and 4.2 percent after ice damage. The Schoenfeld residual test showed that the parameter estimates of a few of the univariate models violated the assumption of proportionality of hazards and therefore were time dependent models

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(table 1). The parameter estimates of the multivariate time dependent model for both no ice damage and after ice damage indicated violation of an assumption of proportionally of hazard while the fixed covariate *SIND* did not (table 1). A semi parametric 'Cox proportional extended model' including time dependent covariates, showed an agreement for hazard ratio assignment of 84 percent for with no ice damage, and 90 percent after an ice damage event (fig. 1).

No Ice Damage: The univariate model with variable *RAQD* and *PLTHT* appeared highly significant as time dependent model (table 1). The univariate model with *RAQD* performed as prognostic model while model with *PLTHT* performed as protective model (table 1). The multivariate model with the variables *RAQD, CRT, SIND* and interaction of *DBH* and log of *TIME2* (upper bound of time interval) was found to be the best time dependent model (table 1). The multivariate model had hazard ratio assignment of 84 percent for plots and time periods with no ice damage.

After Ice Damage: The univariate model with *RAQD*, *BAHA*, *DBH*, and *PCTL* was highly significant ($p < 0.0001$) as time dependent model (table 1). Univariate models using *RAQD*, *BAHA*, and *PCTL* performed as prognostic (associated with increased mortality) models while a model with *DBH* performed as protective model (associated with reduced mortality) (table 1). The multivariate model included *RAQD*, *PLTHT*, *SIND*, *PCTL* and interaction *PLTHT* with *TIME2* was found to be best time dependent model (table 1). The mortality risk of an individual tree belonging to *PCTL1* was 10 times higher than a tree belonging to *PCTL0* (table 1). The multivariate model had hazard ratio assignment of 90 percent for plots and time periods after ice damage.

The position of a tree (*RAQD*), and site productivity (*SIND*) were important prognostic variables in determining survival of an individual shortleaf pine. The variable crown ratio (*CRT)* behaved as significant protective variable, and had marginally increasing influence in survival probability. The influence of *RAQD*

Table 1—Prognostic (*Hzr* >1) and protective (*Hzr* <1) behavior of variables in univariate, and multivariate time dependent models for no ice damage, and after ice damage

Data sets	Univariate models			Multivariate model		
	Variables	Hazard ratio	Probability	Variables	Hazard ratio	Probability
No Ice	RAQD⁺	4.2146	80.82	RAQD⁺	2.1029	67.77
	PLTHT⁺	0.9036	47.47	CRT⁺	0.1204	10.75
	BAHA	1.0578	51.40	SIND	1.1718	53.96
	QDHA	0.9289	48.16	DBH(log(Time2))⁺	0.9773	49.43
	PAG	0.9708	49.26			
	DBH	0.8765	46.71			
	HT⁺	0.8388	45.62			
	CRL	0.5801	36.71			
After Ice	RAQD⁺	3.923	79.29	RAQD⁺	5.1359	83.70
	CRT⁺	0.0003	0.03	PLTHT⁺	1.5754	61.17
	PLTHT⁺	1.0228#	50.56	PLTHT(TIME2)⁺	0.9932	49.83
	BASQM⁺	0.0044	0.44	SIND	1.1145	52.71
	BAHA⁺	1.031	50.76	PCTL1⁺	9.5058	90.48
	DBH⁺	0.9609	49.0			
	DAG	0.0091	0.90			
	PCTL⁺	13.256	92.99			

Note: Univariate models are without fixed covariate '*SIND*', and multivariate models were with fixed covariate '*SIND*'. *No ice* = Data set with all plots (observations) that had no ice damage; *After Ice* = Data set with all plots (observations) after ice damage event. #Coefficient estimate was significant with $p < 0.05$, while other estimates were significant with $p < 0.0001$.

⁺ Variable appeared significant to violate an assumption of proportionality of hazard. Hazard ratio are based on exp (coefficient). *RAQD* = Ratio of quadratic mean diameter to diameter at breast height (dbh); *PLTHT* = Average dominant and co-dominant height (m); *CRL* = Crown length (m); *CRT* = Crown Ratio; *SIND* = Site Index(m); *BAHA* = Stand basal area per hectare (m^2ha^{-1}); *QDHA* = Quadratic mean diameter (cm); *PAG* = Plot age (years); *DBH* = Diameter at breast height (cm); *DAG* = Ratio of dbh to plot age ($cm\ yr^{-1}$); *HT*= Individual tree height (m); *TIME2* = Time at upper interval (months); *PCTL* = Percent of crown loss due to ice damage; *PCTL0* = Crown loss 0-50%; *PCTL1* = Crown loss >50%.

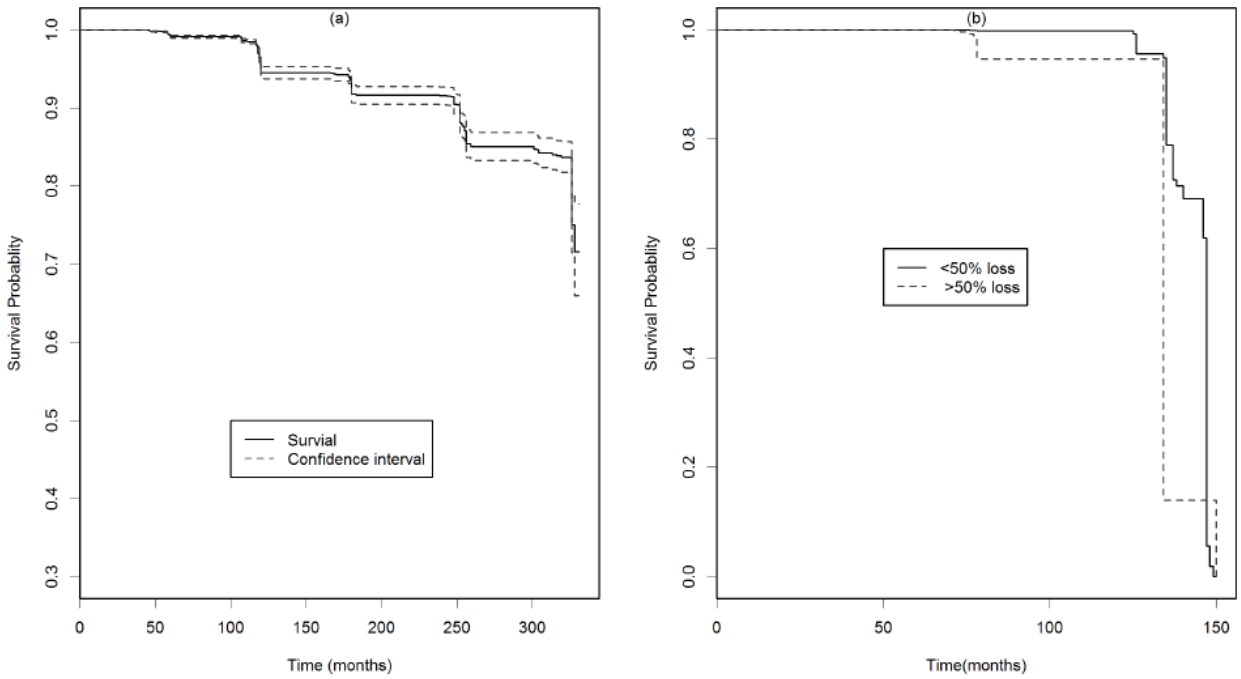


Figure 1—Survival probability by potential by multivariate model with fixed covariate for no ice damage (a) and after ice damage (b). In figure (b) an estimated median survival time (S ($t_{0.5}$) = 0.5) for trees that died during the study was 146 months for *PCTL0*, and was 134 months for *PCTL1*.

in mortality increases over the time. Though average dominant plot height (*PLTHT)* was a prognostic variable, its influence decreases over time. An individual tree with greater than 50 percent crown loss has higher mortality risk than a tree with less than or equal to 50 percent crown loss.

LITERATURE CITED

Allison, P.D. 2010. Survival analysis using SAS: a practical guide. SAS Institute. 336 p.

Fisher, L.D.; Lin, D.Y. 1999. Time-dependent covariates in the Cox proportional-hazards regression model. Annual Review of Public Health. 20(1): 145-157.

Hosmer, D.W.; May, S.; Lemeshow, S. 2008. Applied survival analysis. New York: John Wiley and Sons, Inc. 416 p.

Lynch, T.B.; Hitch, K.L.; Huebschmann, M.M.; Murphy, P.A. 1999. An individual-tree growth and yield prediction system for even-aged natural shortleaf pine forests. Southern Journal of Applied Forestry. 23(4): 203-211.

Kleinbaum, D.G.; Klein, M. 1996. Survival analysis. New York: Springer-Verlag. 700 p.

FUEL AND LITTER CHARACTERISTICS IN FIRE-EXCLUDED AND RESTORED NORTHERN MISSISSIPPI OAK-HICKORY WOODLANDS

Darcy H. Hammond and J. Morgan Varner[1]

Abstract—Oak-hickory communities are a widespread component of the landscape in the southeastern United States, often providing critical habitat and containing high plant species richness. With changing land use and fire exclusion, however, species composition has shifted in many areas to off-site species that are more fire-sensitive. These fire-sensitive species often create a closed-canopy structure (termed "mesophication") and composition that changes fire dynamics and complicates restoration efforts. One important, but understudied, aspect of mesophication is the potential change in fuelbed characteristics that affect fire behavior and future regeneration environments. This study combines fire exclusion controls with canopy thinning/prescribed burning treatments at two oak-hickory woodland sites in northwest Mississippi in order to examine how fuels change with mesophication and restoration efforts. We sampled downed woody fuels and duff depths using planar intercepts and also collected tree litter in destructively sampled quadrats. Most fuels characteristics (1-, 10-, 1000-hour woody fuel loading and fuelbed depth) did not differ between controls and treatments, however at one site the treatment had significantly lower duff depth and higher 100-hour fuel loading. In both restoration treatments, litter composition shifted towards a greater composition of flammable red oak species. Overall, restoration treatments at these sites are resulting in the functional restoration of flammable oak-hickory woodlands.

INTRODUCTION

Fire-maintained woodlands were an important and extensive component of the landscape throughout the eastern U.S. prior to European settlement (Stambaugh and others 2015). The vast majority of these ecosystems have experienced significant declines and degradation across most of their extent. In pyrogenic oak-hickory (*Quercus-Carya*) dominated woodlands in the southeastern U.S. a major source of degradation is fire exclusion, which allows the invasion and establishment of fire-sensitive species (Nowacki and Abrams 2008). These fire-sensitive species can in turn reduce the flammability of the community through a positive feedback process termed "mesophication" due to the lower flammability litter, faster decay rates of litter and woody debris, and alteration of microclimates to promote moister forest floor conditions (Nowacki and Abrams 2008). In addition, Kreye and others (2013) found that fire-sensitive species have litter that absorbs more moisture and dries at a significantly slower rate than fire-tolerant species, both of which would result in dampened ignition and fire spread rates.

Restoration treatments in the region generally target the removal of selected fire-sensitive species, however the effectiveness of these treatments on fuels and flammability have not been well-studied. As efforts to restore fire-maintained woodlands proceed it is critical to develop an understanding of how restoration changes community flammability (Stambaugh and others 2015). Our objectives were to examine the effects of restoration (selective cutting and prescribed fire) on site flammability by examining surface fuels and litter species composition in a southeastern oak-hickory woodland.

METHODS

The study sites are located in mature (containing dominant trees older than 100 years) upland oak-hickory woodlands at the Strawberry Plains Audubon Center near Holly Springs, Mississippi. Dominant tree species include southern red oak (*Quercus falcata*), scarlet oak (*Q. coccinea*), black oak (*Q. velutina*), blackjack oak (*Q. marilandica*), white oak (*Q. alba*), post oak (*Q. stellata*), mockernut hickory (*Carya tomentosa*), sand hickory (*C. pallida*), winged elm (*Ulmus alata*), sweetgum (*Liquidambar styraciflua*), and American sycamore (*Platanus occidentalis*). Other species were present on the sites but they did not contribute significantly to the litter composition.

Paired, adjacent control and treatment plots (each plot ~ 2.5 acres) were established at two sites approximately 2 miles apart (Wildflower and Sharecropper [fig. 1]).

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Figure 1—Study site photos of two north Mississippi oak-hickory woodland sites, Wildflower (left) and Sharecropper (right), during 2014 litter collection. Photo credit: Rochelle Bailey.

Treatment plots were selectively thinned to remove targeted fire-sensitive tree species and to reduce remaining dominant species canopy cover by ~10 percent to approximate pre-settlement conditions (Brewer 2001). Thinning was mostly accomplished by girdling or herbicide application, but some trees were also cut and felled. The Wildflower treatment plot has had five prescribed fires since 2005 and the Sharecropper treatment has had three prescribed fires since 2008; at both sites the burns generally occurred in spring/summer. The Wildflower site has experienced little anthropogenic disturbance, although it has experienced fire exclusion since ~ early 1900's. Areas around the Sharecropper site were farmed historically and subsequently grazed by cattle until ~ early 2000's (Personal communication. Chad Pope. 2014. Staff Ecologist, Strawberry Plains Audubon Center).

Litter was collected in ten 3.3×3.3 foot frames randomly distributed throughout each plot. Four samples per plot were then randomly selected in the lab and sorted to separate out non-leaf material and all leaves that were identifiable to the species level. Due to the similarity of their leaves, decaying scarlet and black oak could not be accurately separated and were therefore lumped into a single category called scarlet oak, which appeared to be the majority of leaves. Leaves that were too damaged to confidently identify were also included in an "unidentifiable" category to ensure the full litterbed was represented. Sorted leaves were then oven-dried at 140° F until no further weight loss was observed.

Downed woody fuels were measured in seven planar intercepts (extending in random directions from systematically distributed start points) per site. Planar intercepts followed the methods in Brown (1974), with 1-hr (diameter <0.25 inches) and 10-hr (0.25 to 1 inch) woody fuels sampled along the first 6 feet, 100-hr (1 to 3 inches) fuels sampled along the first 12 feet , and 1,000-hr fuels (> 3 inches) sampled along the entire 50 foot transect. Fuels within the 1,000-hr category were divided into sound and rotten categories according to Brown (1974). Fuelbed depth was measured three times at the highest point per one foot section along the first three feet of the transect. Duff depth was measured at two points one foot and three feet from the start of the transect.

Percent similarity between plot litter species composition was measured using the Proportional Similarity Index (calculated by summing the lowest percentage for each species in the two plots being compared) on plot averages of dry litter weight percent per species. Site differences in fuels (1-hr, 10-hr, 100-hr, 1,000-hr, and duff) were analyzed using ANOVA or the non-parametric Kruskal-Wallis method when assumptions of normality and equal variance were not met. When differences were detected, post-hoc Tukey-Kramer or Kruskal-Wallis z-tests (non-parametric) were used to isolate pair-wise differences. Analyses were run in NCSS or R statistical software and evaluated using $\alpha = 0.05$.

RESULTS

There were clear changes in litter species composition after the restoration treatments (fig. 2). The Wildflower control (fig. 2A) and treatment (fig. 2C) were 39 percent similar in the relative contribution of different species to leaf litter. The Sharecropper control (fig. 2B) and treatment (fig. 2D) were slightly more similar (45 percent similarity). The two controls were 58 percent similar, whereas the two restoration treatments were 33 percent similar. At both sites there was an increase in the contribution of red oaks (mainly southern red oak, scarlet oak, and blackjack oak) to the leaf litter of the restoration treatment sites and a corresponding decrease in the contribution of white oaks (post oak and white oak). At Sharecropper (fig. 2, B and D) the contribution of hickory remained similar, at 6.9 and 7.6 percent for the control and treatment respectively.

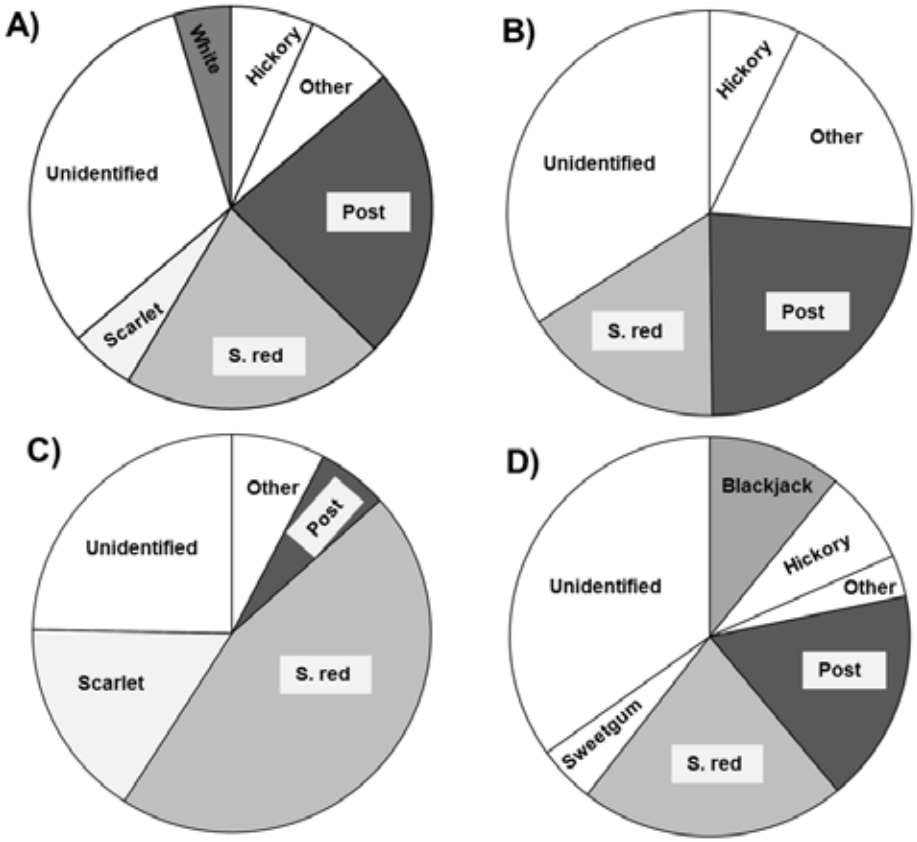


Figure 2—Average percent composition of leaf litter at two north Mississippi oak-hickory woodland sites, Wildflower (left, A and C) and Sharecropper (right, B and D). Control plots are shown in the top row, with corresponding treatment plots on the bottom row. Species that individually contributed less than 5% by weight to litter have been grouped and labeled as "Other" for clarity.

At Wildflower, however, the contribution of hickory decreased from 6.7 to 2.2 percent in the control and treatment.

The richness of identified species contributing to the litter decreased in the treatments at both sites. At Wildflower, the control had identifiable leaves of 14 different species, while the restoration treatment had only 8 identifiable species. The Sharecropper site control had 16 species contributing to the litter composition, while the restoration treatment had only 12 species.

Woody fuels did not generally differ between sites (table 1), however there were significant differences detected for duff depth ($p=0.014$) and 100-hr fuels ($p<0.0001$). Wildflower, but not Sharecropper, had significantly lower duff depth (fig. 3A) in the treatment (0.19 inches) than in the control (1.05 inches). Wildflower, but not Sharecropper, also had lower 100-hr fuel loading (fig. 3B) in the control (0.00 tons/acre) than in the treatment (4.84 tons/acre).

DISCUSSION

In general, the restoration treatments on these sites resulted in greater dominance of the leaf litter bed by flammable red oak species (fig. 2). Southern red oak, blackjack oak, and scarlet oak are known to be highly flammable (Varner and others 2015), with characteristically rapid drying (Kreye and others 2013) followed by high intensity flaming combustion of their curled litter. These oaks also accrue protective bark at young ages, enabling survival in frequent surface fires (Jackson and others 1999).

Classical mesophytic species such as red maple did not comprise a substantial portion of the litter bed even in control plots, generally contributing less than 10 percent of the litter bed combined. Many of these species are also absent from the litter of the treatment plots, although this does not necessarily indicate that those species have been extirpated from the plot. These species have likely only been reduced to a low enough density that they were not detectable in the leaf litter of the treatment plots. Aside from red maple, species in the "other" category included American beech, winged elm, American sycamore, and sweetgum. These species

Table 1—Fuels averages from planar intercepts at two north Mississippi oak-hickory woodland sites, Wildflower and Sharecropper. Means are given with standard deviation, with p-values from Kruskal-Wallis ANOVA (tests significant at an α=0.05 level indicated by an asterisk)

	Wildflower control	Wildflower treatment	Sharecropper control	Sharecropper treatment	P-value
Duff depth (in)	1.05 (0.63)	0.19 (0.14)	0.66 (0.31)	0.59 (0.45)	0.014 *
Fuelbed depth	2.60 (0.56)	2.80 (1.27)	2.60 (0.43)	3.48 (2.93)	0.050
1-hr (tons/ac)	0.20 (0.08)	0.16 (0.05)	0.12 (0.03)	0.16 (0.11)	0.310
10-hr	0.74 (0.55)	0.96 (0.67)	0.39 (0.49)	1.57 (1.60)	0.199
100-hr	0.00 (0.00)	4.84 (2.42)	0.69 (1.18)	3.82 (3.11)	<0.001 *
1000-hr sound	0.60 (1.58)	1.60 (1.40)	0.69 (1.30)	0.45 (0.89)	0.133
1000-hr rotten	0.18 (0.48)	5.11 (6.54)	0.25 (0.66)	2.11 (4.28)	0.205

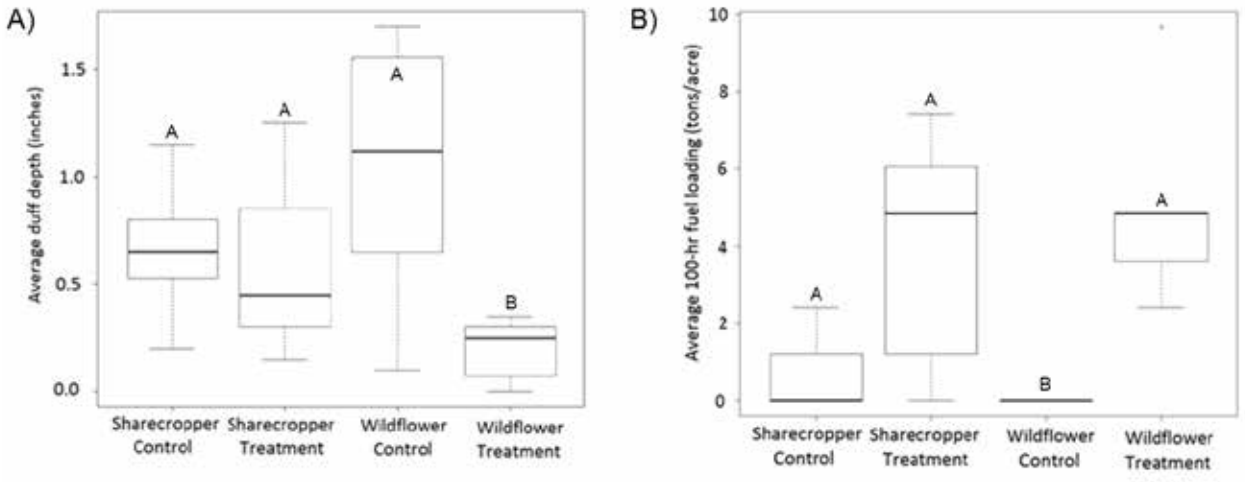


Figure 3—Boxplots highlighting the significant differences (see Table 1) in duff depth (A) and 100-hr fuel loading (B) at two north Mississippi oak-hickory woodland sties. Significant differences between plots (Kruskal-Wallis z-test) are indicated by different letters.

are generally characterized by slow drying rates and typically poor combustion (Kreye and others 2013, Mola and others 2014). These species also accrue protective bark slowly (Jackson and others 1999), preventing their survival and persistence where fires are frequent, though adults of these species typically have bark thick enough to survive prescribed fires.

While it might be expected that the restoration treatments would have lower woody fuels loading due to consumption in prescribed fires, the restoration methods likely resulted in an ongoing influx of woody material due to girdling. This slow attrition likely resulted in the significant increase in 100-hr fuels in the treatment plot at Wildflower (fig. 3B). Sharecropper appeared to have a similar pattern; however the variation is too high for the differences to be considered significant. As killed trees fall and recruit to the surface, likely increases in fire severity will occur. Regardless, the current trend appears to be that girdling increases the amount of 100-hr fuels on the ground and thus would increase community flammability.

The protracted fire-free period prior to restoration resulted in accumulation of forest floor litter and duff. Wildflower has a significantly thinner duff in the restoration treatment while no differences were detected at Sharecropper (fig. 3A). These differences in duff depth could be a result of site history and location. The control at Wildflower has likely experienced a longer period of disturbance exclusion than Sharecropper and thus might be expected to have developed a thicker and more continuous duff layer, which is compounded by the fact that Sharecropper is a generally steeper site.

The components examined in this study are only a small part of the broader picture of community flammability. Fuel moisture changes caused by canopy opening,

changes in herbaceous understory biomass and cover, and potential effects of varying leaf litter mixtures are other key factors that need to be examined in order to get a better picture of changing fire potential with restoration of this woodland. As restoration treatments continue, it is important to consider and evaluate whether these treatments are actually restoring functional site flammability to desired levels. The increases in dominance of litter by high flammability red oaks and increases in 100hr fuels restoration treatments found at these sites are likely increasing the flammability of these oak-hickory woodlands as a direct result of the restoration treatments implemented. As litter contributions shift over time, these species may facilitate surface fires at frequencies previously recorded across the region and maintain species dependent on open woodlands (Stambaugh and others 2015).

ACKNOWLEDGMENTS

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LITERATURE CITED

Brewer, J.S. 2001. Current and presettlement tree species composition of some upland forests in northern Mississippi. Journal of the Torrey Botanical Society. 128: 332-349.

Brown, J.K. 1974. Handbook for inventorying downed woody material. Gen. Tech. Rep. INT-16. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 32 p.

Jackson, J.F.; Adams, D.C.; Jackson, U.B. 1999. Allometry of constitutive defense: A model and a comparative test with tree bark and fire regime. The American Naturalist. 153: 614-632.

Kreye, J.K.; Varner, J.M.; Hiers, J.K.; Mola, J. 2013. Toward a mechanism for eastern North American forest mesophication: differential litter drying across 17 species. Ecological Applications. 23: 1976-1986.

Mola, J.M.; Varner, J.M.; Jules, E.S.; Spector, T. 2014. Altered community flammability in Florida's Apalachicola ravines and implications for the persistence of the endangered conifer *Torreya taxifolia.* PLoS ONE. 9: e103933.

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the eastern United States. BioScience. 58: 123-138.

Stambaugh, M.C.; Varner, J.M.; Noss, R.F. [and others]. 2015. Clarifying the role of fire in the deciduous forests of eastern North America: reply to Matlack. Conservation Biology. 29: 942-946.

Varner, J.M.; Kane, J.K.; Banwell, E.M.; Kreye, J.K. 2015. Flammability of litter from southeastern trees: A preliminary assessment. In: Holley, A.G.; Connor, K.F.; Haywood, J.D., eds. Proceedings of the 17th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-203. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 183-187.

CHANGES IN SOIL CHEMISTRY SIX MONTHS AFTER PRESCRIBED FIRE IN A LONGLEAF PINE PLANTATION IN MISSISSIPPI

John R. Butnor, Kurt H. Johnsen, and C. Dana Nelson[1]

Prescribed fire is used to reduce hardwood competition, enhance herbaceous biodiversity, and improve forage quality in longleaf pine stands. These are primarily low intensity, dormant season burns, during which a portion of the biomass in shrub, herb, and the forest floor layers are combusted. Burning releases elemental nutrients bound in biomass, and there are several potential short-term outcomes: 1) volatilization, 2) surface deposition, 3) uptake by autotrophs, 4) stabilization in soil, and 5) leaching. Several studies have examined long-term effects of repeated burn cycles in southern pine stands [e.g., Binkley and others (1992)] or periods of 1 to 3 years post-burn [e.g., Lavoie and others (2010)], though no significant changes in mineral soil C or N have been reported after one year. There are no detailed reports of intra-annual effects of prescribed fire on soil chemistry in longleaf pine stands. Dormant season burns are followed by leaf-out and the growing season, when uptake of newly released nutrients would be likely to occur.

We sought to quantify changes in forest floor mass and mineral soil nutrients (0–5 and 5–10 cm depths) during the first 6 months after fire in a 53 year-old longleaf pine study located on the Harrison Experimental Forest near Saucier, MS (Schmidtling 1973). After an absence of fire of more than seven years, the plantation was burned in 2010 and thinned in 2011. Sixteen plots were sampled pre-burn (January 2014) and repeated 1, 3, and 6 months post-burn. Forest floor and soil samples (0–5 cm, 5–10 cm depths) were collected at 4 cardinal points 1 m from plot center and composited for nutrient analysis. Bulk density was measured before and after the experiment by determining the dry mass of a known volume of soil. Prescribed fire occurred on February 28, 2014. Forest floor mass loss, soil bulk density, and soil C contents (dry combustion) were analyzed at Forestry Sciences Laboratory in Research Triangle Park, NC. Micronutrients (P, K, Ca, Cu, Fe, Na, Mg, Mn) were analyzed at Spectrum Analytic, Washington Court House, OH. Repeated Measures Analysis (RMA) was used to evaluate change over time with SAS® 9.4 software (SAS Institute Inc. 2013).

After the fire, the residual forest floor mass had enriched C (46.3 percent) and N (0.66 percent) concentrations compared to initial concentrations of forest floor C (38.5 percent) and N (0.54 percent). The fire reduced forest floor mass by >70 percent compared to the mean pre-burn condition (16,413 kg biomass ha^{-1}), releasing 4,122 kg C ha^{-1}. There were no changes in mineral soil C concentration at either depth 1, 3, or 6 months after burning, indicating a loss of forest floor C to combustion and subsequent leaching\surface flow of residual char and soot with precipitation. Removal of the forest floor may also stimulate oxidation of labile soil C as forest floor temperatures increase, further reducing C contents. It is interesting that soil C was unaffected by burning as pyrogenic transformation of organic matter to biochemically stable black carbon has garnered much interest for C sequestration. Current thinking is that unless the residual material is mixed or incorporated into soil, it is rapidly transported off site. Soil pH increased significantly after burning at both depths: 4.37 to 4.55 at 0–5 cm and 4.58 to 4.78 at 5–10 cm. This was anticipated due to the liming effect of calcium carbonate released from the forest floor. Burning of organic matter mineralizes a portion of nutrients making them readily available. Analysis of soil micronutrients at 0–5 cm shows several elements (P, K, Mg, Mn) spike immediately after the fire, peaking at months 1 or 3, and decline by month 6, while increases in Ca persist in the upper 5 cm of soil (fig. 1). The site is P deficient making the spike in available soil P well timed at the beginning of the growing season. Changes in soil micronutrients were less pronounced at depths of 5–10 cm. Soil N concentrations were lower than our analyzer was able to measure (< 0.1 percent) and will be examined once the analyzer has been reconfigured for low-range analysis (>0.01 percent) . Although there is transfer of nutrients from forest floor to soil associated with burning, a portion is volatilized based on temperature-specific nutrient sensitivities (DeBano 1991). Nitrogen is very sensitive to volatilization at temperatures as low as 200 to 375°C (DeBano 1991), P and K are moderately sensitive (e.g., 774°C) (Raison and others 1985), while Ca, Mg, and Mn are insensitive

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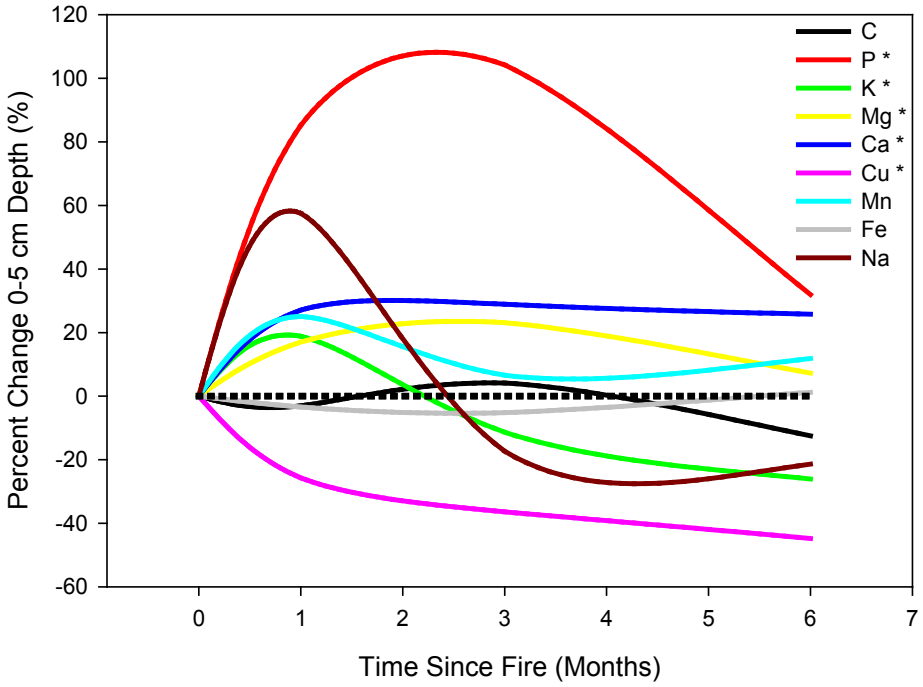


Figure 1—Percent change in soil C and soil micronutrients at 1, 3, and 6 months post-fire in the upper 5 cm of soil collected from a 53 year-old longleaf pine stand (n=16). Element concentration over time (4 data points) was fitted to a spline curve for ease of viewing. No change from initial condition is represented by the dotted black line. Element concentrations that have significantly changed in the months following burning are indicated by an asterisk.

with volatilization temperatures higher than observed for flaming woody biomass >1100°C (DeBano 1991).

LITERATURE CITED

Binkley, D.; Richter, D.; David, M.B.; Caldwell, B. 1992. Soil chemistry in a loblolly longleaf pine forest with interval burning. Ecological Applications. 2(2): 157-164.

DeBano, L.F. 1991.The effect of fire on soil properties. In: Harvey, A.E.; Neuenschwander, L.F., comps. Proceedings-management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station: 151-156.

Lavoie, M.; Starr, G.; Mack, M.C. [and others]. 2010. Effects of a prescribed fire on understory vegetation, carbon pools, and soil nutrients in a longleaf pine-slash pine forest in Florida. Natural Areas Journal. 30(1): 82-94.

Raison, R.J.; Khanna, P.K.; Woods P.V. 1985. Mechanisms of element transfer to the atmosphere during vegetation fires. Canadian Journal of Forest Research. 15: 132-140.

SAS Institute Inc. 2013. SAS® 9.4 software. Cary, NC: SAS Institute Inc.

Schmidtling, R.C. 1973. Intensive culture increases growth without affecting wood quality of young southern pines. Canadian Journal of Forest Research. 3: 565-573.

TEMPORAL AND SPATIAL PATTERNS OF SOIL CO_2 EFFLUX, SOIL CARBON, AND ROOT BIOMASS ASSOCIATED WITH BEDDING IN YOUNG LOBLOLLY PINE PLANTATIONS

Chris A. Maier, Pete Anderson, John R. Butnor, Phillip M. Dougherty, Kurt Johnsen, and Daniel McInnis[1]

Abstract—We measured soil CO_2 efflux (Fs) in four loblolly pine plantations in the coastal plain of South Carolina in an effort to understand how site preparation, drainage, and microclimate affect Fs, root biomass, and soil carbon pools during early stand development. All plantations were site prepared: sheared, raked, and bedded. Soil CO_2 efflux, temperature (Ts), moisture (θ), root biomass (Rb), coarse (COF) and fine (FOF) organic fragments, and mineral soil carbon (Cs) were measured quarterly during the first two years of stand growth. Mean daily Fs were similar between sites and ranged from 0.5 to 12 mmol m^{-2} s^{-1} during the winter and summer, respectively. Soil CO_2 efflux, COF, FOF, and Cs were significantly greater in the beds than inter-rows on wet sites, but not on the dry site. Soil temperature accounted for 26-55 percent of the variation in Fs across all sites. Soil θ and Cs explained a significant, but small amount (6-22 percent) of variance in Fs. Annual soil carbon efflux ranged from 12 to19 Mg C ha^{-1} yr^{-1}. We conclude that bedding during site preparation can have significant effects on the spatial variation in Fs and associated drivers, with some site-specific caveats.

INTRODUCTION

Managed pine plantations in the Southeastern United States play a prominent role in the regional and global carbon cycle (Turner and others 1995). Net ecosystem productivity (NEP), a measure of carbon sequestration, reflects the change in carbon stored in vegetation and soil and is the small difference between carbon uptake in photosynthesis and loss through autotrophic and heterotrophic respiration (Chapin and others 2002). Intensively managed pine plantations have the potential to increase NEP by increasing net primary productivity (NPP; Maier and others 2004). However, factors regulating the soil carbon cycle, i.e. soil carbon inputs, transformations, and decomposition may be more important for determining NEP (Janssens and others 2001, Valentini and others 2000). Regenerating pine plantations are a net carbon source (-NEP) immediately after harvest because heterotrophic respiration exceeds NPP. The recovery time for a new plantation to become a net carbon sink (+NEP) will differ with site and depends on the degree of soil disturbance during site preparation (e.g. burning, disking, and bedding), site fertility, and NPP of the regenerating stand (Sampson and others 2008).

Soil CO_2 efflux (Fs) is comprised of autotrophic (root and associated fungi) and heterotrophic (microbial decomposition of soil organic matter) respiration (Hanson and others 2000). Soil temperature and moisture greatly influence the component processes of Fs (see Hanson and others 2000 and references therein); however, following a disturbance such as harvesting and site preparation, soil organic matter and nitrogen content are important factors regulating F_s (Rustad and others 2000). Soils in the Coastal Plain of the Southeastern United States are some of the most productive sites for intensive pine management (Allen and Campbell 1988). These soils are often carbon and nutrient rich and have the potential for significant releases of carbon when disturbed. A common management practice is whole-tree harvesting followed by intensive site preparation that includes stump shearing, raking, and bedding. Bedding mixes surface organic layers into the mineral soil, enriches soil carbon and nutrients, increases soil aeration, and improves drainage (Haines and others 1975, McLaughlin and others 2000, Trettin and others 1996). This heavy soil disturbance can increase decomposition and Fs (Ewel and others 1987a, Mallik and Hu 1997) potentially leading to a net loss in soil carbon (Henderson 1995).

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Quantifying the effects of site preparation on the carbon dynamics during early stand growth is fundamental to understanding both the carbon cycle and the role of intensive management in sequestering carbon.

We examine the spatial and temporal variation in Fs in loblolly pine (*Pinus taeda* L.) plantations during the first two years of stand growth. Ancillary measurements of soil temperature, moisture, organic matter, and root biomass were made to determine their importance in explaining variability in Fs. Plantations were growing on four soil types that differed in drainage class. All sites received standard site preparation protocols that included bedding. Our objectives were to 1) measure the temporal and spatial variability in Fs, 2) develop empirical models of Fs based on site-specific factors, and 3) estimate annual soil carbon efflux.

MATERIAL AND METHODS

Study Sites

The study took place on commercial forestlands in the upper Coastal Plain of South Carolina, USA. Measurements were made on four sites: Andrews (A), Camphall (C), Oswald (O), and Watson Hill (W). The sites differed in soil type, drainage class, and soil physical characteristics (table 1). Sites were clear-cut harvested to remove the 20–25 year-old loblolly pine and then site-prepared (sheared, raked, and bedded). Bedding created three distinct microsites (bed, inter-row, and trough) with distinct soil temperature, moisture, organic matter, and physical characteristics. Beds were 1.83 m wide and were 25-30 cm and 30-40 cm higher than adjacent inter-rows (0.92 m wide) and troughs (0.45 m wide), respectively.

Seedlings were planted on beds at a 1.8 m spacing (1290 trees ha^{-1}) in the winter of 1999. On each site, three 50x50 m plots were selected for measurement. Each plot contained 12 rows with 27 seedlings per row (324 seedlings). Measurement plots were confined to the inner eight rows.

Measurement of Soil CO_2 Efflux (Fs)

Soil CO_2 efflux (Fs) was measured with the multi-chambered Automated Carbon Efflux System (ACES) developed at the USDA Forest Service, Southern Research Station Laboratory in Research Triangle Park, NC (Butnor and others 2003). Soil chambers were 25 cm in diameter (491 cm^2) and were equipped with air and soil (5 cm, Ts) thermocouples. The system has been shown to give consistent responses regardless of differences in soil and litter properties and has been calibrated to provide true efflux rates (Butnor and others 2005). Fifteen soil chambers were placed in groups of three in a diagonal transect across the plot. Transects were randomly located each measurement period. Within a group, the first chamber was placed on the bed 25 cm from a seedling (tree), the second on the bed equal distance between trees (between-tree), and the third on the adjacent inter-row. Occasionally, troughs were measured. Chambers were measured sequentially, six to nine times, over a 24 hour period and then averaged to compute daily average Fs. Volumetric soil moisture (θ) to a 30 cm depth was measured at each chamber location with time domain reflectometry (CS615; Campbell Scientific, Ogden, Utah).

Organic Matter, Carbon, Nitrogen and Root Measurements

Following Fs measurements, a 10 cm diameter by 20 cm depth soil core was taken at each chamber. The soil was sieved through a 6.4 mm mesh screen to remove large live roots and coarse organic fragments (COF). A 500 g subsample of sieved soil was washed with a hydropneumatic elutriator (Gillison's Variety Fabrication, Inc., Benzonia, MI) to separate fine organic fragments (FOF) and small live roots. Root biomass (Rb, large and small), COF, and FOF were dried at 65°C and weighed and expressed per unit surface area (kg m^{-2}). Soil carbon (Cs) and nitrogen (Ns) concentration (mg g^{-1}) were determined on a 20 g sample of oven-dried soil with a Carlo Erba NA 1500 Series II C/N/S Analyzer (Fison Instruments, Danvers, MA).

Table 1—Study site soil characteristics and site index of the previous stand

Site	Soil Series	Soil type	Description	Drainage	SI25[1]
Andrews	Bladen	thermic Typic Albaquults	Fine sandy loam (<35 cm), clay (>35cm)	Very poorly	25
Camphall	Rains	thermic Typic Paleaquults	Deep sandy loam	Poorly	24
Oswald	Ocilla/ Yemassee	thermic Aquic Arenic Paleudults	Deep loamy sand	Somewhat Poorly	22
Watson Hill	Alpin	thermic, coated Aquic Quartzipsamments	Fine sand	Well	22

[1]SI – site index at 25 years (meters)

Statistical Analysis

Soil CO_2 efflux, Ts, **θ**, COF, FOF, Cs, Ns, and Rb were measured quarterly over two years beginning in July 1999. Plot averages, the average of 4-5 measurements per plot, served as the experimental unit. Site and location within site (bed or inter-row) effects were tested by using a randomized complete block analysis of variance with repeated measures (PROC MIXED, SAS Institute Inc. 1987). Main or interactive effects were tested at α=0.05. Tukey's adjustment was used for pair-wise comparison of site and site x location means.

Correlation analysis (PROC CORR) and linear regression (PROC REG) were used to assess the spatial and temporal variation in measured parameters and to quantify the response of Fs to environmental and site variables. Individual chamber measurements were used for these analyses. Equation 1 was used to describe the relationship between Fs and temperature:

$$\ln(Fs) = a + b*\ln Ts \quad (1)$$

where ln Fs and ln Ts are log transformed Fs and Ts. Analysis of covariance was used to test for site and site x location effects on the regression parameters. Regression lines were first analyzed by testing the entire line (i.e. intercepts and slopes simultaneously) with full and reduced models (Zarnoch 2009). If significant site or site x location effects were detected, separate analyses were performed for slope or intercepts effects. Linear contrasts were used to test for differences between regressions and for making pairwise comparisons. When making multiple comparisons, Type I experimentwise error was minimized by using the Bonfferoni correction to derive the appropriate significance level (α). For example, with a full model comparing four regression lines, there would be six contrasts and the appropriate significance level would be: α=(0.05/6)=0.008 (Zarnoch 2009).

RESULTS

Temporal and Spatial Variation in Fs and Site Characteristics

Average daily Ts varied seasonally from around 5 °C in the winter to greater than 30 °C in the summer (fig. 1). Sites were measured on different days within each period, and day to day variation in temperature resulted in a significant site and site x period interaction (table 2). Within a site, Ts was similar among chamber locations on beds and inter-rows (site x location, p=0.78). Averaged θ over the study was 47.5, 30.1, 23.4, and 12.9 $m^3\ m^{-3}$ for sites A, C, O, and W, respectively. There was a significant site x location x period interaction; where θ was significantly higher in the inter-rows than beds for at least part of the year on all of the sites (fig.1).

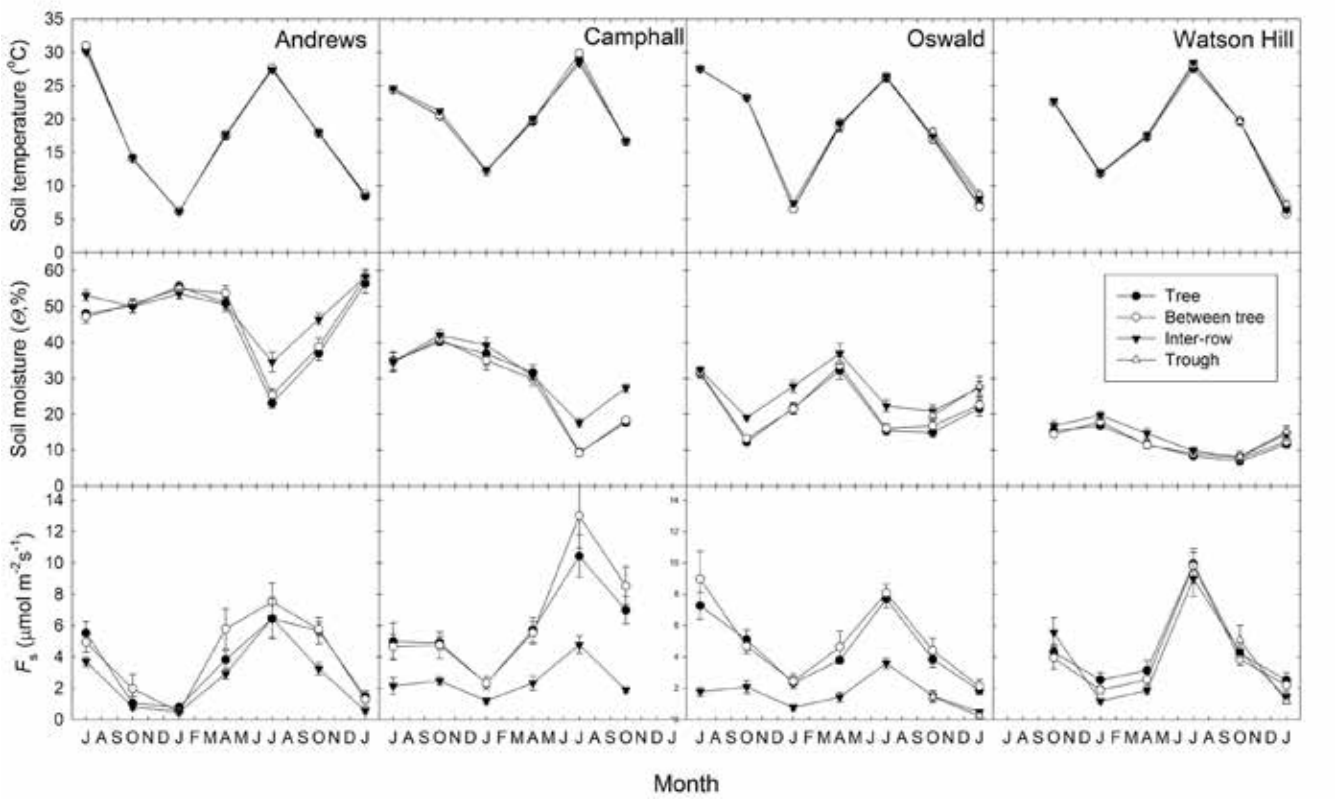


Figure 1—Temporal and spatial patterns of daily average soil temperature (Ts) measured at 10 cm, volumetric soil moisture (θ), and soil CO_2 efflux (Fs) measured on the beds adjacent to and half-way between planted seedlings and between the beds in the inter-row, and trough. Data are least square means (LSMEAN) and standard error.

Table 2—Probability values for effects of site (S), sampling location (L, bed or inter-row), and sampling period (P) on coarse organic (COF, kg m^{-2}) and fine (FOF, kg m^{-2}) organic fragments, mineral soil carbon (Cs, mg g^{-1}) and nitrogen (Ns, mg g^{-2}), total root biomass (Rb, kg m^{-2}), soil CO_2 efflux (Fs, µmol m^{-2} s^{-1}), soil temperature (Ts, °C), and volumetric soil moisture (θ) (n = number of observations)

Effect	COM	FOM	Cs	Ns	Rb	Fs[1]	Ts	θ
S	0.0139	0.0053	0.2436	0.0007	0.1195	0.0447	0.0032	<0.0001
L	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	0.1425	<0.0001
S x L	<0.0001	<0.0001	<0.0001	<0.0069	0.3834	<0.0001	0.7890	0.0861
P	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
S x P	0.0992	<0.0001	0.0122	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
L x P	0.2245	0.6225	0.1102	0.1993	<0.0001	0.0082	0.5723	0.0015
S x L x P	0.1515	0.4950	0.7932	0.9159	0.1098	0.1465	0.9233	0.0072
n	1209	1209	1209	1209	1209	853	853	853

[1] Analysis is for sampling periods when all sites were measured.

Soil CO_2 efflux averaged across all measurement periods, was 3.74, 5.41, 3.76, and 4.57 µmol m^{-2} s^{-1} (se=0.33) for sites A, C, O, and W, respectively, and site C was significantly greater than sites A (p=0.034) and O (p=0.034) (table 2). Fs varied from <0.5 mmol m^{-2} s^{-1} in January to >12 mmol m^{-2} s^{-1} in July and closely followed the seasonal trend in Ts (fig. 1). Within a site, there was no significant difference in Fs measured on beds next to a tree or between trees (figs.1 and 2); however, Fs was significantly higher on beds than inter-rows at sites C and O (p<0.0001) and marginally greater at site A (p=0.1067). Measurement location had no effect on Fs at site W (p=0.998). Fs measured in troughs were similar to inter-rows (sites O and W).

There were significant period and site x period interactions for COF, FOF, Cs, and Ns (table 2); however, there were no discernible trends over time for any of the parameters (data not shown). Site A had greater COF than site W (p=0.036), while site C had greater FOF than site A (p=0.049) and O (p=0.0007) (fig.2). There was a significant site x location interaction where beds had greater COF at sites A and O and greater FOF at sites A, C, and O than inter-rows. In contrast, there were no location effects on COF or FOF at site W. There was no site effect on Cs (table 2); however, there was a significant site x location effect where Cs was greater in the beds than inter-row (p<0.05) for sites A, C, and O, but not W (p=0.99). Site A had significantly greater Ns than the other sites (table 2, fig.2) and Ns was greater in beds than in the inter-row at A, C, and O, but not at W. There were no site differences in Rb (table 2) and there was no site x location interaction; although beds tended to have more Rb than the inter-rows (fig.2).

Fs Response to Site and Environmental Variables

Soil CO_2 efflux was best correlated with Ts at all the sites (table 3). There was a significant (p<0.001) site* Ts effect (i.e. slope) on the relationship between Fs and Ts (equation 1) (data not shown), indicating the need for site specific regressions. Within site, Ts explained 26 to 55 percent of the variance in Fs and inter-row locations had better fits with Ts than beds (fig. 3). The temperature sensitivity (slope) of Fs was similar for bed and inter-row locations at sites A (p=0.457) and C (p=0.377); however, beds had a greater intercept indicating that at a given temperature; beds had higher Fs than inter-rows. In contrast, Fs was more sensitive to Ts in the inter-rows than beds at sites O (p=0.011) and W (p=0.019).

Fs was negatively correlated with θ at sites A, C, and W, but not site O, while Fs was positively correlated with Rb at sites C, O, and W, but not site A (table 3). There was a positive correlation between Fs and FOF at all sites and positive correlation with COF, Cs, and Ns at some sites, but not others. There was also significant covariation among environmental and site variables. For example, θ was higher during the winter when Ts was low (fig.1) resulting in a negative correlation between θ and Ts at sites A, C, and W. Rb was positively correlated with Ts, but negatively correlated with θ at sites C, O, and W, and FOF was positively correlated with Ts at all of the sites.

The covariation between Ts and other variables made it difficult to discern independent relationships with Fs. To minimize the confounding temperature effect, Fs was normalized to 20 °C (Fs(20)) by using equation 1 and the parameter estimates in figure 3. Relationships between Fs(20) and other variables were examined with stepwise regression. Soil moisture, Cs, COF, FOF, and Rb together accounted for 6 to 32 percent of the variance in Fs(20); however, the effect of any individual parameter was relatively weak and the relative importance differed with site (table 4). Soil moisture and Cs were significant

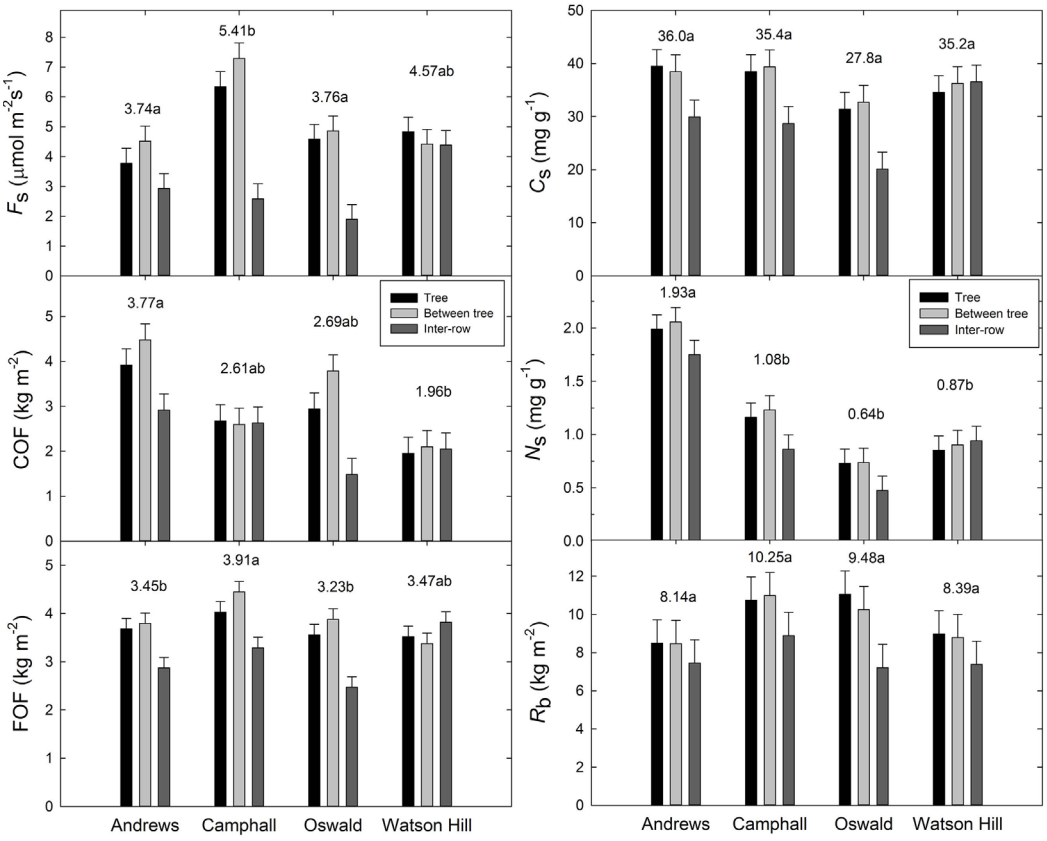


Figure 2—Means and standard errors for soil CO_2 efflux (Fs), coarse (COF) and fine (FOF) organic fragments, mineral soil carbon (Cs) and nitrogen (Ns) and live root biomass (Rb) measured on the beds adjacent to and half-way between planted seedlings and between the beds in the inter-row, and trough. Each bar is the average and standard error across all measurement periods (n=6 or 7). Values above bars are the site average across measurement locations and the letter denotes a significant difference between sites at p=0.05. Data are least square means (LSMEAN) and standard error.

at all of the sites, while COF, FOF, and Rb were important variables at site O. Root biomass was the single most important variable at the dry site (W). When all site were considered together, Cs, θ, Rb, and COF explained 18 percent of the variance in Fs(20).

Annual Estimates

Annual soil carbon efflux was computed by using the site-specific temperature equations (fig. 3) and average daily Ts measured at on-site weather stations. Carbon efflux from the beds was more than twice that in inter-rows at sites C and O (fig. 4), and was 42 and 6 percent greater at sites A and W, respectively. Accounting for the spatial coverage of beds, troughs, and inter-rows and assuming troughs had similar Ts and Fs to inter-rows, the annual carbon efflux was 12.7, 18.7, 11.7, and 15.9 Mg C ha^{-1} at sites A, C, O, and W, respectively.

DISCUSSION

The range of average daily Fs (0.5 – 12 µmol m^{-2} s^{-1}) were similar to those observed in nearby one to three year-old plantations that received the same site preparation (Gough and others 2005, Tyree and others 2014). These values are much higher than that measured in 1 to 2 year-old stands located in the Virginia piedmont (< 2 µmol m^{-2} s^{-1}) (Pangle and Seiler 2002, Wiseman and Seiler 2004). These large regional differences are likely due to increased heterotrophic respiration caused by bedding on the South Carolina sites (Gough and others 2005). However, the effect of bedding on Fs is site specific. On the wetter sites (A, C, and O), Fs was 42 to193 percent greater on beds than inter-rows, while bedding had no effect on Fs on the well-drained site (site W). Annual soil C loss ranged from 11.7 to 18.7 Mg C ha^{-1}, greater than measured in other loblolly pine stands (Butnor and others 2003, Gough and others 2005, Maier and Kress 2000, Palmroth and others 2005), but lower than that reported for a recently clearcut *Pinus eliottii* plantation (22.7 Mg C ha^{-1}, Ewel and others 1987a). The high rates of annual carbon flux are likely to persist. Gough and others (2005) measured Fs over a loblolly pine chronosequence (0 to 22 years) on sites close to our study sites and found that Fs rates were stable over time decreasing only slightly with stand age. They attributed this response to offsetting effects of root and heterotrophic respiration as stands age. These stands may take 5-8 years to become annual net carbon sinks (+NEP) depending on soil type, severity

Table 3—Correlation coefficients for soil CO_2 efflux (Fs), soil temperature (Ts), volumetric soil moisture (Θ), live root biomass (Rb), coarse (COM) and fine (FOM) organic fragments, and mineral soil carbon (Cs) and nitrogen (Ns)

	Fs	Ts	Θ	Rb	COM	FOM	Cs
				Andrews			
Ts	0.55						
Θ	-0.50	-0.52					
Rb	0.05	0.05	-0.26				
COM	0.07	-0.01	0.02	-0.05			
FOM	0.15	0.21	-0.09	-0.00	0.44		
Cs	0.11	-0.18	0.05	-0.09	0.28	0.33	
Ns	-0.05	-0.28	0.17	0.00	0.12	0.16	0.71
				Camphall			
Ts	0.51						
Θ	-0.51	-0.40					
Rb	0.47	0.43	-0.55				
COM	0.23	0.13	0.09	-0.01			
FOM	0.30	0.49	-0.23	0.35	0.22		
Cs	0.24	0.02	-0.01	-0.05	0.14	0.37	
Ns	0.10	-0.03	0.20	-0.02	0.03	0.15	0.55
				Oswald			
Ts	0.44						
Θ	-0.19	-0.07					
Rb	0.39	0.24	-0.29				
COM	0.42	0.10	-0.00	-0.11			
FOM	0.38	0.19	-0.42	0.48	0.08		
Cs	0.32	0.03	-0.03	0.03	0.30	0.29	
Ns	0.12	0.05	-0.10	-0.05	0.12	0.05	0.56
				Watson Hill			
Ts	0.69						
Θ	-0.32	-0.30					
Rb	0.68	0.46	-0.25				
COM	-0.08	-0.12	0.25	-0.09			
FOM	0.50	0.45	-0.06	0.59	0.28		
Cs	0.13	-0.02	0.25	0.12	0.34	0.31	
Ns	-0.00	-0.04	0.14	-0.01	0.14	0.05	0.58

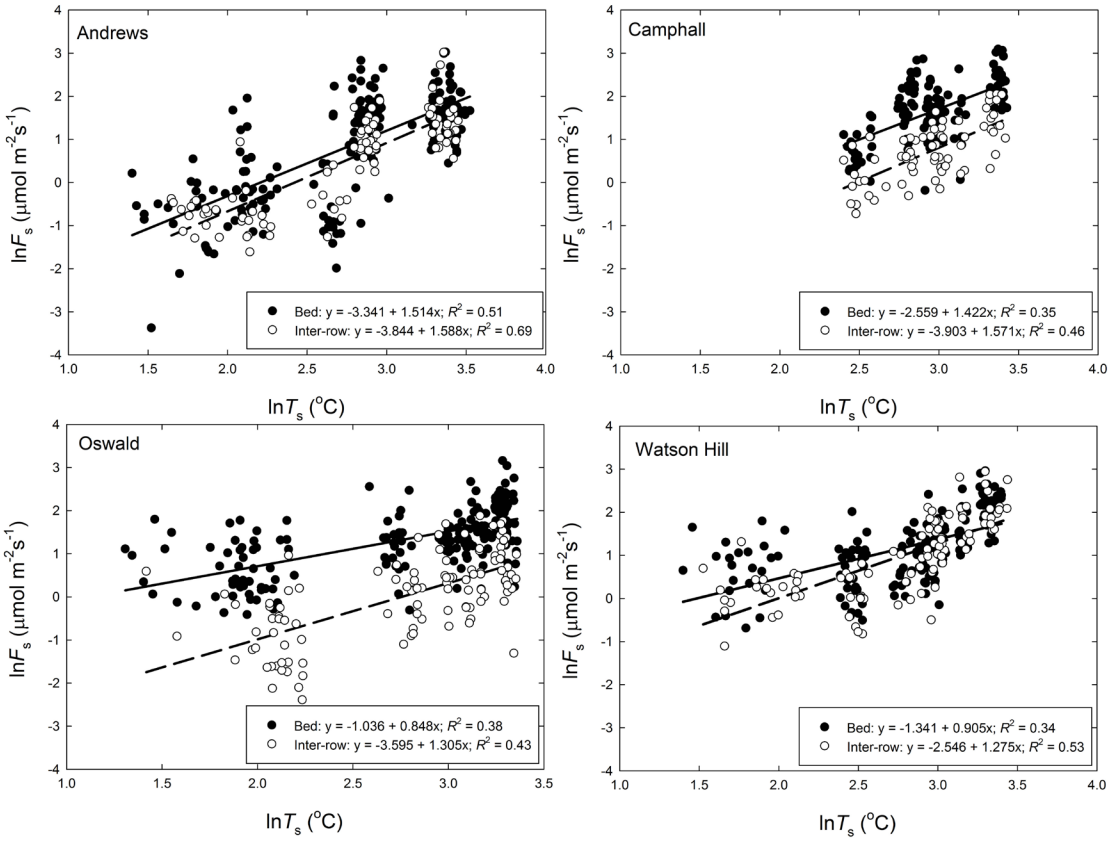


Figure 3—The relationship between log transformed soil CO_2 efflux (Fs) and soil temperature (Ts). Data are individual chamber measurements across all sampling periods. The line is the least square fit for beds (solid) and inter-row (dashed) locations (equation 1).

of soil disturbance, and management (e.g. fertilization, weed control) (Sampson and others 2008). Fertilization can reduce the time for stands to gain positive NEP. For example, four years of fertilization of an infertile sandy site shifted NEP of a 12 year-old loblolly pine stands from carbon neutral (0.28 Mg C $ha^{-1}yr^{-1}$, non-fertilized) to strong carbon sinks (6.4 Mg C $ha^{-1}yr^{-1}$, fertilized) (Maier et al 2004). The increase in NEP was primarily a function of NPP. However, fertilization may also increase NEP by decreasing Fs (Butnor and others 2003, Haynes and Gower 1995, Samuelson and others 2004) primarily through decreased soil organic matter decomposition (Janssens and others 2010).

Soil temperature was the primary driver of Fs explaining 26-55 percent of the variation, comparable to that reported in other studies for young loblolly pine (Gough and others 2005, Maier and Kress 2000, Pangle and Seiler 2002, Samuelson and others 2009). There were significant site differences in the temperature response indicating that site-specific equation were needed to model Fs. Furthermore, on the drier sites (O and W), the sensitivity of Fs to Ts was greater in the inter-rows than beds. The cause of this site specific effect is unknown, but may be due to the dissimilar sensitivities of root and heterotrophic respiration to changes temperature, moisture, and substrate supply (Boone and others 1998, Davidson and others 2006, Johnsen and others 2007).

Spatial variation in Fs was correlated to differences in θ, COF, FOF, Cs, Ns, and Rb associated with bedding (fig. 2, table 3); however, individual relationships with Fs were generally weak. After accounting for temperature effects, Cs explained a significant but small amount of variation (3to13 percent) in Fs at all of the sites, while COF and FOF explained an additional 10 percent of the variation at site O. Root biomass had little influence on Fs except at the dry site (site W) where it was the single most important variable explaining 17 percent of the variance in Fs(20). It makes sense that due to low root biomass, heterotrophic processes are the dominant source of Fs in young developing stands (Ewel and others 1987b). Several studies have found a correlation between Fs and Cs, and coarse organic debris or fragments (Gough and Seiler 2004, Hanson and others 1993, Mallik and Hu 1997, Pangle and Seiler 2002,); however, relationships are generally weak. Tyree and others (2014) found that soils augmented with logging residues increased heterotrophic respiration, but had no effect on Fs. These studies indicate that static estimates of carbon pool size in Cs, COF, FOF, or Rb are not particularly useful for predicting instantaneous

Table 4—Summary of stepwise multiple regression of factors that influence soil CO_2 efflux normalized to 20°C (Fs(20)). Factors are volumetric soil moisture (θ), coarse organic fragments (COF), fine organic fragments (FOF), live root biomass (Rb), and mineral soil carbon (Cs). Only parameters significant at p=0.05 were included in the analysis (n=number of observations)

Variable	Partial R^2	Model R^2	C(p)	F-value	P>F	n
			Andrews			
θ	0.0310	0.0310	15.64	9.10	0.0028	286
Cs	0.0297	0.0607	8.52	8.94	0.0030	
			Camphall			
Cs	0.1028	0.1028	29.66	22.23	<0.0001	196
θ	0.1186	0.2214	2.37	29.39	<0.0001	
			Oswald			
Cs	0.1329	0.1329	82.30	46.13	<0.0001	303
FOF	0.0683	0.2012	54.26	25.66	<0.0001	
θ	0.0508	0.2520	33.93	20.29	<0.0001	
COF	0.0437	0.2957	16.70	18.51	<0.0001	
Rb	0.0289	0.3246	6.00	12.7	0.0004	
			Watson Hill			
Rb	0.1674	0.1674	27.12	53.48	<0.0001	268
Cs	0.0460	0.2134	13.04	15.49	<0.0001	
θ	0.0295	0.2429	4.71	10.3	0.0015	
			All Sites			
Cs	0.0809	0.0809	123.27	92.46	<0.0001	1053
θ	0.0557	0.1366	54.22	67.74	<0.0001	
Rb	0.0301	0.1667	17.79	37.93	<0.0001	
COF	0.0124	0.1791	4.03	15.77	<0.0001	

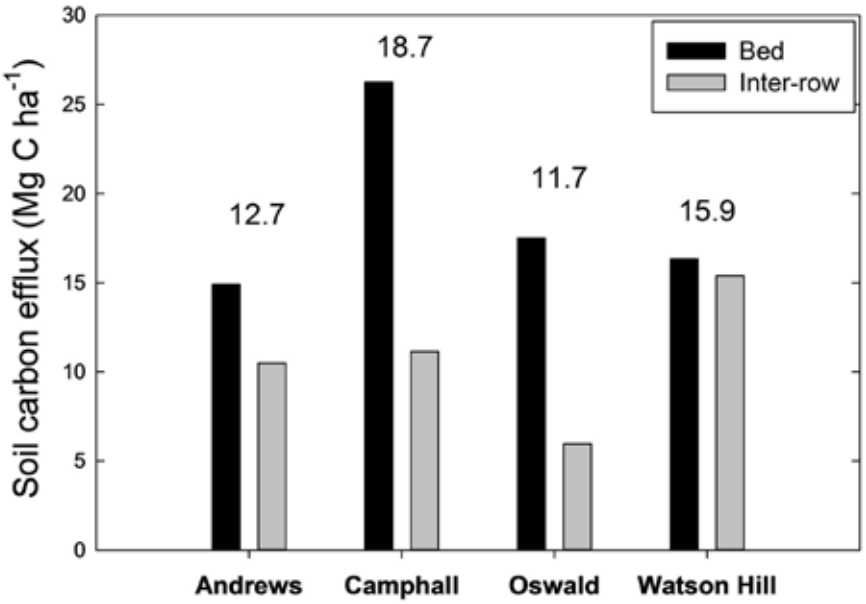


Figure 4—Annual soil carbon efflux from the bed and inter-row locations for each site. The value above the bar is the annual carbon efflux for the site accounting for spatial coverage of the bed, inter-row, and trough.

measures of Fs (Gough and Seiler 2004, Reichstein and others 2003).

Soil CO_2 efflux was weakly (<12 percent) negatively related to θ. A strong soil moisture effect on Fs has not been observed in loblolly pine plantations even when measured across a wide range of θ (Gough and Seiler 2004, Pangle and Seiler 2002, Selig and others 2008, Samuelson and others 2009). The shape of the response of Fs to θ is variable, and depends on soil physical characteristics, organic matter content, and to the differential effects of moisture on root and heterotrophic respiration (Hanson and others 2000). Extreme wet or dry soil can inhibit Fs and between the extremes, θ may have no obvious effect (Fang and Moncrieff 2001, Lavigne and others 2004). For example, on well drained sandy soils, irrigation treatments increased Fs in loblolly pine but only when the soil was dry (Maier and Kress 2000, Samuelson and others 2009). In mixed pine stands, Fs was positively correlated to θ on sandy soils but not on fine textured clays (Dilustro and others 2005) and on a clay piedmont soil, Fs increased with θ, but only when $\theta < 0.2$ $m^3 m^{-3}$ (Palmroth and others 2005).

These studies suggest that loblolly pine plantations may rarely experience critical levels of θ that inhibit Fs. However, infrequent (e.g. weekly or monthly) measurements of Fs may miss or may not be able to discern subtle moisture effects on Fs such as short-term changes that occurs after rainfall. For example, Ford and others (2012) estimated annual soil carbon efflux in irrigated and non-irrigated longleaf pine stands using continuous (i.e. hourly) and biweekly measurements of Fs. Irrigated stands had 37 percent greater carbon flux than non-irrigated stands when estimated using continuous measurements of Fs; however, there was no significant irrigation effect on carbon efflux when estimated from biweekly measurements. They concluded that biweekly measurements missed short-term increases of Fs that occurred after irrigation treatments. Clinton and others (2011) made hourly measurements of Fs in a mid-rotation longleaf pine stand, found that Fs increased sharply following a 13 mm rainfall event, and then steadily fell over the next two weeks as θ declined. Others have also reported short-term increases in Fs after rainfall events (Jarvis and others 2007, Xu and Qi 2001). These short-term pulses in Fs can account for 5 to 37 percent of annual carbon flux (Daly and others 2008, Lee and others 2002, Lee and others 2004, Palmroth and others 2005). Short-term changes in θ following rainfall can also increase the temperature sensitivity of Fs (Palmroth and others 2005). These studies indicate that infrequent chamber-based measurements of Fs probably do not have the resolution for capturing soil moisture effects on Fs and potential interactions between soil moisture and temperature.

CONCLUSION

Annual soil carbon efflux rates ranged between 11 and 18 Mg C ha^{-1} yr^{-1} and are some of the highest reported for young loblolly pine. Bedding had site specific effects on the spatial variation in Fs, θ, and soil carbon pools. Bedding increased Fs, soil carbon stocks, and decreased θ on sites with poor to moderately poor drainage, but not on a well-drained site. Wet sites will likely experience accelerated carbon loss because of bedding. Conversely, bedding should increase site carbon uptake through increased tree survival and growth and may quickly offset high Fs. The site-specific spatial variation in Fs and associated drivers should be considered when modeling Fs in young pine stands.

As expected, Ts explained the largest amount of variation in Fs. Soil moisture, soil carbon, and root biomass explained only a small amount of spatial variation in Fs. These variables are not likely to be useful for predicting instantaneous measures of Fs. Furthermore, infrequent chamber-based measurements of Fs probably do not have the resolution for capturing transient θ effects on Fs.

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LITERATURE CITED

Allen, H.; Campbell, R.; Hook, D. 1988. Wet site pine management in the southeastern United States. In: Hook, D.D.; McKee, W.H.; Smith, H.K. [and others], eds. The ecology and management of wetlands. Vol. 2: Management, use and value of wetlands. Kent, UK: Croom Helm Ltd.: 173-184.

Boone, R.D.; Nadelhoffer, K.J.; Canary, J.D.; Kaye, J.P. 1998. Roots exert a strong influence on the temperature sensitivity of soil respiration. Nature. 396: 570-572.

Butnor, J.R.; Johnsen, K.H.; Maier, C.A. 2005. Soil properties differently influence estimates of soil CO2 efflux from three chamber-based measurement systems. Biogeochemistry. 73: 283-301.

Butnor, J.R.; Johnsen, K.H.; Oren, R.; Katul, G.G. 2003. Reduction of forest floor respiration by fertilization on both carbon dioxide-enriched and reference 17 year-old loblolly pine stands. Global Change Biology. 9: 849-861.

Chapin, F.S., III; Matson, P.A.; Mooney, H.A., eds. 2002. Principles of Terrestrial Ecosystem Ecology. New York: Springer-Verlag. 436 pp.

Clinton, B.D.; Maier, C.A.; Ford, C.R.; Mitchell, R.J. 2011. Transient changes in transpiration, and stem and soil CO2 efflux in longleaf pine (*Pinus palustris* Mill.) following fire-induced leaf area reduction. Trees. 25: 997-1007.

Daly, E.; Oishi, A.C.; Porporato, A.; Katul, G.G. 2008. A stochastic model for daily subsurface CO_2 concentration and related soil respiration. Advances in Water Resources. 31: 987-994.

Davidson, E.A.; Janssens, I.A. 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature. 440: 165-173.

Dilustro, J.J.; Collins, B.; Duncan, L.; Crawford, C. 2005. Moisture and soil texture effects on soil CO_2 efflux components in southeastern mixed pine forests. Forest Ecology and Management. 204: 87-97.

Ewel, K.; W.P. Cropper, W.P.; Gholz, H.L. 1987a. Soil CO_2 evolution in Florida slash pine plantations II. Importance of root respiration. Canadian Journal of Forest Research. 17: 330-333.

Ewel, K.; W.P. Cropper, W.P.; Gholz, H.L. 1987b. Soil CO_2 evolution in Florida slash pine plantations. I. Changes through time. Canadian Journal of Forest Research. 17: 325-329.

Fang, C.; Moncrieff, J.B. 2001. The dependence of soil CO_2 efflux on temperature. Soil Biology and Biochemistry. 33: 155-165.

Ford, C.R.; McGee, J.; Scandellari, F. [and others]. 2012. Long- and short-term precipitation effects on soil CO_2 efflux and total belowground carbon allocation. Agricultural and Forest Meteorology. 156: 54-64.

Gough, C.M.; Seiler, J.R. 2004. The influence of environmental, soil carbon, root, and stand characteristics on soil CO_2 efflux in loblolly pine (*Pinus taeda* L.) plantations located in the South Carolina coastal plain. Forest Ecology and Management. 191: 353-363.

Gough, C.M.; Seiler, J.R.; Wiseman, P.E.; Maier, C.A. 2005. Soil CO_2 efflux in loblolly pine (*Pinus taeda* L.) plantations on the Virginia Piedmont and South Carolina Coastal Plain over a rotation-length chronosequence. Biogeochemistry. 73: 127-147.

Haines, L.; Maki, T.; Sanderford, S. 1975. The effect of mechanical site preparation treatments on soil productivity and tree (*Pinus taeda* L. and *P. elliottii* Engelm. var. *elliottii*) growth. Forest soils and forest land management. Quebec, Canada: University of Laval Press: 379-395.

Hanson, P.J.; Edwards, N.T.; Garten, C.T.; Andrews, J.A. 2000. Separating root and soil microbial contributions to soil respiration: A review of methods and observations. Biogeochemistry. 48: 115-146.

Hanson, P.J.; Wullschleger, S.D.; Bohlman, S.A.; Todd, D.E. 1993. Seasonal and topographic patterns of forest floor CO_2 efflux from an upland oak forest. Tree Physiology. 13: 1-15.

Haynes, B.E.; Gower, S.T. 1995. Belowground carbon allocation in unfertilized and fertilized red pine plantations in northern Wisconsin. Tree Physiology. 15: 317-325.

Henderson, G.S.; McFee, W.; Kelly, J. 1995. Soil organic matter: a link between forest management and productivity. In: Carbon forms and functions in forest soils. Soil Science Society of America Inc.: 419-435.

Janssens, I.A.; Dieleman, W.; Luyssaert, S. [and others]. 2010. Reduction of forest soil respiration in response to nitrogen deposition. Nature Geoscience. 3: 315-322.

Janssens, I.A.; Lankreijer, H.; Mattaucci, G. [and others]. 2001. Productivity overshadows temperature in determining soil and ecosystem respiration across European forests. Global Change Biology. 7: 269-278.

Jarvis, P.; Rey, A.; Petsikos, C. [and others]. 2007. Drying and wetting of Mediterranean soils stimulates decomposition and carbon dioxide emission: the “Birch effect”. Tree Physiology. 27: 929-940.

Johnsen, K.; Maier, C. ;Sanchez, F. [and others]. 2007. Physiological girdling of pine trees via phloem chilling: proof of concept. Plant, Cell and Environment. 30: 128-134.

Lavigne, M.B., Foster, R.J., Goodine, G. 2004. Seasonal and annual changes in soil respiration in relation to soil temperature, water potential and trenching. Tree Physiology. 24: 415-424.

Lee, M.S.; Nakane, K.; Nakatsubo, T. [and others]. 2002. Effects of rainfall events on soil CO_2 flux in a cool temperate deciduous broad-leaved forest. Ecological Research. 17: 401-409.

Lee, X.; Wu, H.-J.; Sigler, J. [and others] 2004. Rapid and transient response of soil respiration to rain. Global Change Biology. 10: 1017-1026.

Maier, C.A.; Albaugh, T.J.; Allen, H.L.; Dougherty, P.M. 2004. Respiratory carbon use and carbon storage in mid-rotation loblolly pine (*Pinus taeda* L.) plantations: the effect of site resources on the stand carbon balance. Global Change Biology. 10: 1335-1350.

Maier, C.A.; Kress, L.W. 2000. Soil CO_2 evolution and root respiration in 11 year-old loblolly pine (*Pinus taeda*) plantations as affected by moisture and nutrient availability. Canadian Journal of Forest Research. 30: 347-359.

Mallik, A.U.; Hu, D. 1997. Soil respiration following site preparation treatments in boreal mixedwood forest. Forest Ecology and Management. 97: 265-275.

McLaughlin, J.W.; Gale, M.R.; Jurgensen, M.F.; Trettin, C.C. 2000. Soil organic matter and nitrogen cycling in response to harvesting, mechanical site preparation, and fertilization in a wetland with a mineral substrate. Forest Ecology and Management. 129: 7-23.

Palmroth, S.; Maier, C.A.; McCarthy, H.R. [and others]. 2005. Contrasting responses to drought of forest floor CO_2 efflux in a loblolly pine plantation and a nearby Oak-Hickory forest. Global Change Biology. 11: 421-434.

Pangle, R.E.; Seiler, J. 2002. Influence of seedling roots, environmental factors and soil characteristics on soil CO_2 efflux rates in a 2-year-old loblolly pine (*Pinus taeda* L.) plantation in the Virginia Piedmont. Environmental Pollution. 116: S85-S96.

Reichstein, M.; Rey, A.; Freibauer, A. [and others]. 2003. Modeling temporal and large-scale spatial variability of soil respiration from soil water availability, temperature and vegetation productivity indices. Global Biogeochemical Cycles. 17: 15-1 – 15-15.

Rustad, L.E.; Huntington, T.G.; Boone, R.D. 2000. Controls on soil respiration: Implications for climate change. Biogeochemistry. 48: 1-6.

Sampson, D.A.; Wynne, R.H.; Seiler, J.R. 2008. Edaphic and climatic effects on forest stand development, net primary production, and net ecosystem productivity simulated for Coastal Plain loblolly pine in Virginia. Journal of Geophysical Research: Biogeosciences. 113: G01003.

Samuelson, L.; Mathew, R.; Stokes, T. [and others]. 2009. Soil and microbial respiration in a loblolly pine plantation in response to seven years of irrigation and fertilization. Forest Ecology and Management. 258: 2431-2438.

Samuelson, L.J.; Johnsen, K.; Stokes, T.; Lu, W. 2004. Intensive management modifies soil CO_2 efflux in 6 year-old Pinus taeda L. stands. Forest Ecology and Management. 200: 335-345.

Selig, M.F.; Seiler, J.R.; Tyree, M.C. 2008. Soil Carbon and CO_2 Efflux as Influenced by the Thinning of Loblolly Pine Plantations on the Piedmont of Virginia. Forest Science. 54: 58-66.

Trettin, C.C.; Davidian; M.; Jurgensen, M.F.; Lea, R. 1996. Organic Matter Decomposition following Harvesting and Site Preparation of a Forested Wetland. Soil Science Society of American Journal. 60: 1994-2003.

Turner, D.P.; Koerper, G.J.; Harmon, M.E.; Lee, J.J. 1995. A carbon budget for forests of the conterminous United States. Ecological Applications. 5: 421-436.

Tyree, M.C.; Seiler, J.R.; Maier, C.A. 2014. Contrasting genotypes, soil amendments, and their interactive effects on short-term total soil CO_2 efflux in a 3-year-old *Pinus taeda* L. plantation. Soil Biology and Biochemistry. 69: 93-100.

Valentini, R.; Matteucci, G.; Dolman, A.J. [and others]. 2000. Respiration as the main determinant of carbon balance in European forests. Nature. 404: 861-865.

Wiseman, P.E.; Seiler, J.R. 2004. Soil CO_2 efflux across four age classes of plantation loblolly pine (Pinus taeda L.) on the Virginia Piedmont. Forest Ecology and Management. 192: 297-311.

Xu, M.; DeBiase, T.A.; Qi, Y. [and others]. 2001. Ecosystem respiration in a young ponderosa pine plantation in the Sierra Nevada Mountains, California. Tree Physiology. 21: 309-318.

Zarnoch, Stanley J. 2009. Testing hypotheses for differences between linear regression lines. e-Res. Note SRS-17. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 20 p.

IS NATURAL DEFENSE CAPACITY CORRELATED WITH ALLOCATION OF DRY MASS TO THE STEM IN LOBLOLLY PINE?

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Abstract—In addition to selecting loblolly pine (*Pinus taeda* L.) genotypes for superior growth, the concept of customized genetic selection may apply where tree vigor is threatened by insects and disease. A study conducted with seedlings from 15 loblolly pine genotypes found significant correlation between phenolic production and foliage mass when dry mass allocation to the stem was relatively low. Validation of this relationship in trees was attempted among four genetic sources of juvenile loblolly pine. Tree biomass allocation information was collected from six individuals per genotype at age 5 years in 2009. Five years later in January 2015, foliage and branch tip tissues were assessed for total phenolic concentrations. While tree biomass allocation differed significantly, total phenolic concentrations were similar among the four loblolly pine genotypes. Observations during this study suggest that our effort to validate this relationship in trees should be repeated during the growing season when carbon demands are at their highest and in stand conditions that exclude the possibility of light limitations to carbon fixation.

INTRODUCTION

Phenolic compounds are an important source of both naturally occurring and inducible plant defenses (Lattanzio and others 2006). Past research has reported that among many plant species, both forms of phenolic defense are heritable (Agrawal 1999, Datta and Lal 2012, Witzell and Martín 2008). For example, Danielsson and others (2011) compared levels of naturally occurring and induced terpene and phenol compounds in response to inoculation with *Heterobasidion annosum s.l.* among Norway spruce (*Picea abies* L.) genotypes that exhibited high or low susceptibility when challenged by this pathogen. Constitutive phenolic profiles, but not terpene profiles, differed between the two levels of *Heterobasidion* susceptibility, with a negative relationship between astringin compound content and susceptibility. Between 5 and 15 days after inoculation with *Heterobasidion,* there was an increase in catechin concentration in the less susceptible Norway spruce genotypes. Gene expression analyses indicated that genes involved in the phenylpropanoid, epicatechin, and catechin pathways were activated by inoculation with *H. annosum*.

Carbon allocation among plant components has been characterized by a hierarchy that meets the carbon demand of foliage and stem before the root system and secondary metabolites (Dickson 1991). More recently in a compilation of data collected from 63 forested ecosystems, Litton and others (2007) revised this model. They suggested that carbon allocation to foliage and maintenance respiration is relatively constant regardless of stand environment, but that the fraction of fixed carbon allocated to the stem, roots, and other entities varies by stand factors. Furthermore, with the carbon needs of foliage and respiration met, a larger fraction of fixed carbon is allocated to wood production in the stem and branches when soil resources are plentiful. Alternatively, poor soil fertility and repeated water deficit shift carbon allocation away from wood production toward root system growth. Therefore, in most forest settings, after the carbon support of foliage and respiration are met, the fate of fixed carbon varies by an array of factors that influence the availability of essential resources.

We propose that within the context of the carbon allocation model outlined by Litton and others (2007), genotype exerts some control on the carbon allocation pattern of loblolly pine (*Pinus taeda* L.) (Bongarten and Teskey 1987, Li and others 1991, Stovall and others 2013), and selection of loblolly pine genotypes with a high capacity to produce constitutive or induced defenses would benefit commercial forestry in the South on sites where insect attack or disease are anticipated. Greenhouse experimental results recently

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showed that stem total phenolic concentration and dry mass allocation patterns differed significantly among 15 families of commercially deployed loblolly pine seedlings (Singh 2012). Furthermore, stem total phenolic concentration was correlated with both stem dry mass fraction and total foliage dry mass, so that as dry mass allocation to the stem decreased, the relationship between stem total phenolic concentration and foliage mass became more robust. A large foliage dry mass could benefit phenolic defense capacity by producing ample photosynthate for phenolic biosynthesis (Aspinwall and others 2011). Our objective was to determine if this relationship between total phenolic concentration, dry mass allocation to the stem, and foliage dry mass in seedlings would also be present in trees. Using existing allometric data from four genotypes of loblolly pine at age 5 years and total phenolic concentrations of plant tissues collected at age 10 years in January 2015, we hypothesized that the genotypes characterized by significantly lower stem dry mass allocation would have significantly higher total phenolic concentrations and that they would be positively correlated with foliage dry mass.

MATERIALS AND METHODS

Four rapidly growing sources of loblolly pine were planted at a 1.8 m x 4.9 m spacing as container-grown seedlings at the Louisiana State University AgCenter Hill Farm Research Station near Homer, LA, in January 2005. Each source was replicated 12 times in 0.06-ha plots with four plots per source assigned to one of three blocks by soil textural attributes contributing to drainage. Two sources were open-pollinated, half-sibling families originating in the eastern United States (7-56 and 8-103). The other two sources were clones propagated by rooted cuttings, also originating in the eastern United States (CL9 and CL93). Two of the sources were characterized by a compact crown shape (7-56 and CL93), and two of the sources (8-103 and CL9) were characterized by a broad crown shape.

In August 2009, one tree in one randomly chosen plot per source in each block was destructively harvested so that the dry mass of all foliage age classes, the stem, and the branches could be determined after total tree height and diameter at breast height (dbh) measurements. From these data, total aboveground dry mass was calculated and allometric equations were developed to predict aboveground dry mass and its distribution for all trees per plot in 2009. Allometric equations and growth measurements of all trees per plot were applied and plot means were calculated to predict mean tree aboveground total and component dry masses and the percentage of dry mass allocated to foliage, stem, and branches by tree in 2009.

In late February to early March 2010, five fascicles from the first flush of 2009 were sampled from five trees in all experimental plots and pooled by plot. Foliage samples were oven dried to equilibrium at 70°C, ground to pass a 0.85 mm^2 screen, and evaluated for macronutrient and micronutrient concentrations (A&L Laboratories, Memphis, TN).

In January 2015, two sunlit shoots from two randomly selected trees per plot used for the development of allometric equations were sampled. Shoots were taken from the mid-crown by telescoping pruning shears. First-flush foliage grown in 2014 and the terminal 10 cm (excluding buds) of branches were excised from the shoots, pooled by tissue and plot, freeze dried, and ground to pass through a 0.50 mm^2 screen. Processed tissues were analyzed for total phenolic concentration by a modification of the Folin-Ciocalteu method originally described by Singleton and Rossi (1965) and adapted by Booker and Maier (2001) for loblolly pine. Total phenolic concentration was expressed as micrograms of catechin equivalents per milligram dry weight.

Mean dry mass allocation to foliage, stem, and branches at age 5 years; mean percentage of dry mass allocated to foliage, stem, and branches at age 5 years; and mean foliage and branch tip total phenolic concentrations at age 10 years were evaluated by analysis of variance using a randomized complete block design with three blocks. Means by source were evaluated by the Tukey pair-wise comparison test. Relationships between tree dbh and total phenolic concentrations were evaluated by Pearson correlation coefficients. The *F*-statistics and mean differences were considered significant at an α level of 0.05.

RESULTS AND DISCUSSION

At age 5 years, the fraction of dry mass allocated to the stem was significantly greater for 8-103 (65 percent) and CL9 (62 percent) compared to 7-56 (50 percent) and CL93 (45 percent). At the same time, the fraction of dry mass allocated to foliage was significantly lower for 8-103 (18 percent) and CL9 (21 percent) compared to 7-56 (33 percent) and CL93 (43 percent). A similar pattern was observed for foliage dry mass, with that of 7-56 (2.7 kg) and CL93 (3.1 kg) being significantly greater than that of 8-103 (0.9 kg) and CL9 (1.2 kg). These foliage dry mass observations follow a similar trend in measurements of tree leaf area conducted at ages 4 and 5 years by Osbon and others (2012). The aboveground total dry weight of 7-56 (8.8 kg) was significantly greater than that of 8-103 (5.2 kg) and CL9 (5.4 kg) but not CL93 (7.2 kg). One reason for the large aboveground dry mass of 7-56 was its significantly larger branch size compared to the other three loblolly pine sources. As anticipated, the strong positive relationship between leaf area and tree growth (Vose and Allen 1988) was beneficial to stemwood and branch production in 7-56 and stemwood production in CL93.

Mean total phenolic concentrations of foliage and branch tissues were 70.2 ± 2.2 (standard error) and 56.1 ± 0.7 µg catechin mg^{-1}, respectively. These values are within the range of total phenolic concentrations reported by other investigators for loblolly pine seedlings and saplings. For example, seedlings grown under ambient light in a greenhouse between May and October had 39.2 and 22.3 µg catechin mg^{-1} in the foliage and stem, respectively (Gebauer and others 1998). Booker and Maier (2001) assessed the total phenolic concentration of a single cohort of loblolly pine seedling foliage. In September of the year of fascicle initiation, total phenolic concentration averaged 35 µg catechin mg^{-1}. In February, July, and September of the next year, total phenolic concentrations averaged 65, 88, and 100 µg catechin mg^{-1}. Aspinwall and others (2011) found that in October, the foliar total phenolic concentration of nine sources of genetically improved 2-year-old loblolly pine ranged between 104 and 130 µg catechin mg^{-1}. Results of these studies indicate that the phenolic concentration of foliage is greater than that of woody tissue, and the total phenolic concentration of foliage can be expected to increase over time.

We hypothesized that phenolic defense production would be hindered under conditions of high dry mass allocation to the stem and limited foliage dry mass accumulation. The 8-103 and CL9 sources of loblolly pine produced a relatively large fraction of stem dry mass and a relatively small fraction of foliage dry mass. We expected, therefore, to see lower total phenolic concentrations for 8-103 and CL9 compared to 7-56 and CL93. Total phenolic concentration among the four sources of loblolly pine did not differ significantly in foliage ($Pr > F = 0.4571$) or branch tips ($Pr > F = 0.8129$) (fig. 1). Clearly, a new effort to validate the relationship that we observed among total phenolic concentration, fraction of dry mass allocated to the stem, and foliage dry mass in seedlings will be required. The present effort provides insight for this second validation effort.

For the foliage tissue, but not the branch tip tissue of CL9, we found a significant negative relationship ($r^2 = 0.9700$, $Pr > F = 0.0033$) between total phenolic concentration and tree size represented by dbh (figs. 2a and b). In contrast, correlations between total phenolic concentration and tree growth were not observed for 7-56, CL93, or 8-103. Aspinwall and others (2011) also found departure in this relationship between loblolly pine clones grown in two locations with similar site indices but different stocking levels. They suggested that where stocking was high, resource constraints caused by intraspecific competition reduced carbon fixation such that it met the carbon demands of growth but not phenolic biosynthesis. Where stocking was less competitive, ample resources benefited both tree growth and phenolic production.

The accessibility of fixed carbon is driven by light availability in the crown. Trees in our study will reach canopy closure within the next 3 years, and thus intraspecific competition for light and other essential resources is currently high (Personal communication. Michael Blazier. 2015. Associate Professor, LSU AgCenter, Hill Farm Research Station, 11959 Highway 9, Homer, LA. 71040). Application of Aspinwall and others' (2011) theory to the relationship between total phenolic concentration and tree growth in our study indicates that at the time of tissue sampling for total phenolics, CL9 may have been experiencing one or more resource deficits that limited carbon allocation to phenolic biosynthesis. A similar problem was not apparent for 7-56, CL93, and 8-103.

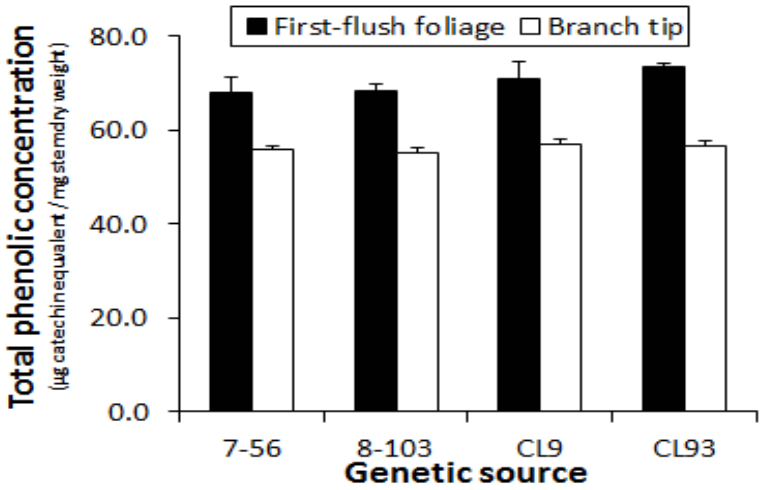


Figure 1—Total phenolic concentration means of foliage from the first flush and branch tips of four genetic sources of 10-year-old loblolly pine in January 2015. Bars represent one standard error of the mean.

In the dormant season of 2009, foliar nutrition was sufficient for all sources of loblolly pine (table 1). Also, while annual precipitation was 12 percent less than normal in 2014, this year was not characterized by drought (NDMC 2015). Thus, it is unlikely that nutrition or water limited the synthesis of phenolic compounds in the 2014 first-flush foliage or branch tips that were sampled in January 2015. Because the trees in our study were approaching canopy closure, it is possible that light availability, and thus whole-crown carbon fixation, was negatively affected by interaction between crown architecture and tree size.

The wide crown shape combined with the lower leaf area of CL9 compared to 7-56 and CL93 may have caused CL9 to reach a plateau of whole-crown carbon fixation with the approach of canopy closure. In contrast, it is possible that the compact crown shapes of 7-56 and CL93 aided light interception by the 2014 first-flush foliage. As a result, high leaf area and the way that leaf area was displayed in 2014 could have prolonged the period when fixed carbon satisfied the both tree growth and phenolic synthesis in 7-56 and CL93.

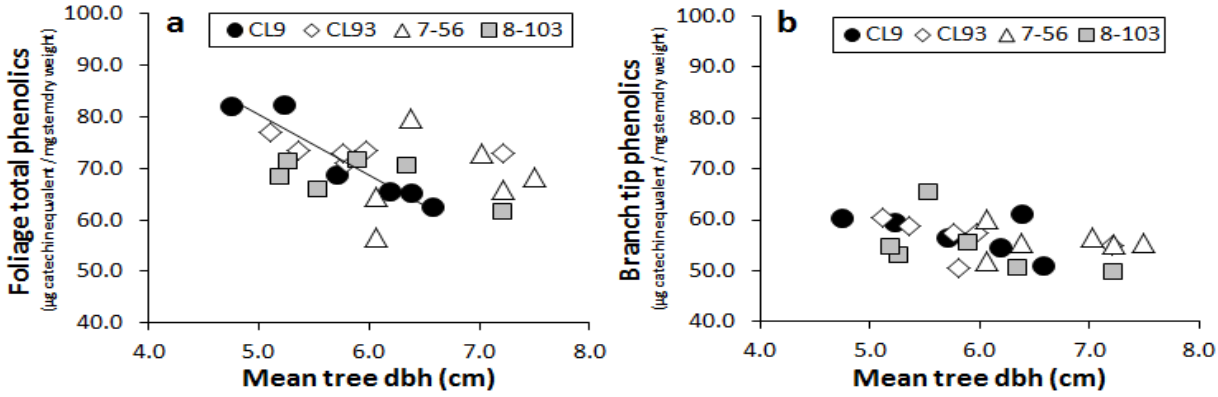


Figure 2— Relationship between mean tree diameter at breast height (dbh) at age 5 years and total phenolic concentration of (a) first-flush foliage and (b) branch tips at age 10 years for four genetic sources of loblolly pine. For the first-flush foliage of CL9, this relationship was statistically significant (r^2 = 0.9700, $Pr > F$ = 0.0033).

Table 1—Foliar concentrations of plant-essential mineral nutrients in four genetic sources of 5-year-old loblolly pine located at the Louisiana State University AgCenter Hill Farm Research Station near Homer, LA

Mineral nutrient[1]	Genetic source			
	CL9	CL93	7-56	8-103
Nitrogen (%)[2]	1.3 (0.1)	1.4 (0.1)	1.3 (0.1)	1.3 (0.1)
Phosphorus (%)	0.10 (0.03)	0.11 (0.01)	0.10 (0.02)	0.10 (0.01)
Potassium (%)	0.58 (0.08)	0.55 (0.06)	0.58 (0.07)	0.55 (0.08)
Calcium (%)	0.19 (0.03)	0.22 (0.07)	0.21 (0.04)	0.20 (0.03)
Magnesium (%)	0.09 (0.01)	0.10 (0.02)	0.08 (0.01)	0.09 (0.02)
Boron (ppm)	30 (9)	31 (10)	32 (8)	33 (6)
Zinc (ppm)	40 (5)	37 (6)	39 (5)	39 (5)
Manganese (ppm)	462 (163)	591 (120)	491 (172)	455 (77)
Iron (ppm)	66 (21)	74 (22)	73 (39)	67 (20)
Copper (ppm)	4 (1)	15 (42)	4 (1)	4 (2)
Aluminum (ppm)[3]	339 (59)	340 (42)	397 (82)	385 (67)

[1] Foliage is from the first flush of 2009 and was collected at age 5 years in late February to early March 2010. Values represent the mean and standard deviation of foliage sampled from the upper crown of five trees per plot.

[2] %: percentage; ppm: parts per million.

[3] While aluminum is not plant essential, it is included for informational purposes.

The dissimilar relationship between foliar phenolic concentration and dbh of CL9 and 8-103 cannot be explained by observations made in this study. Knowledge of root system architecture in addition to that of the crown, however, might reveal additional factors responsible for the potential carbon limitation to phenolic biosynthesis as CL9 tree size increased. For example, root system excavations at the time of aboveground biomass determinations showed that CL9 averaged fewer first-order lateral roots in a 0.5 m^3 block of soil around the stem that was 0.5 m deep compared to 8-103, 7-56, and CL93 (n=6; CL9: 8; 8-103: 17; 7-56: 20; CL93: 12). If the number of first-order lateral roots was correlated with root system expansion, then soil resource acquisition may have been lower for CL9 than 8-103. A subsequent negative effect on photosynthesis could have led to reduced carbon allocation to phenolic synthesis in CL9 compared to 8-103.

Foliage and branch tips were sampled for phenolic analyses outside the normal growing season, when the metabolic activity of woody tissues is low (Blanche and others 1992). During the dormant season, foliage physiology is closely tied to air temperature, with the potential for photosynthesis rates to be nearly as high as those during the growing season (Tang and others 2003). The difference in dormant-season metabolic rates between branch tip and foliage tissues could account for the observation of a significant negative relationship between total phenolic concentration and dbh in foliage, but not branch tips, of CL9.

CONCLUSION

Our present goal was to determine if a relationship between total phenolic concentration and dry mass allocation pattern among 15 genetic sources of loblolly pine seedlings would be observed among a different set of 4 genetic sources of loblolly pine trees. Validation of this relationship in trees would justify development of an index of dry mass allocation pattern that could be used to screen genetic sources of loblolly pine for natural phenolic defense capacity. Ultimately, this index would help land managers choose the loblolly pine genotype best suited for planting where the risks of insect attack and disease are high. Aboveground dry mass allocation pattern differed among our four genetic sources of loblolly pine planted in early 2005. Foliage and branch tip total phenolic concentrations, however, did not differ significantly among genotypes. Regression between total phenolic concentration and mean tree dbh indicated that the impact of crown shape on light interception, and therefore carbon fixation, might have been greater for one genotype than the other genotypes.

This effort also provides insight about how a second validation effort should be conducted. First, genotypes should be chosen that have genetically controlled dry mass allocation differences. Because aboveground dry mass allocation pattern and total phenolic production are under some degree of genetic control, a lag in time between measurements of these attributes is acceptable. It is critical, however, that the tested genotypes are in forest stand environments free of soil resource and light limitations. The resolution of differences in total phenolic concentration among genetic sources will likely be increased by sampling in the active growing season rather than the dormant season. Also, because its total phenolic concentration increases over time and is relatively high, mature foliage rather than developing foliage seems to be the best tissue for total phenolic surveys.

LITERATURE CITED

Agrawal, A. 1999. Induced plant defense: Evolution of induction and adaptive phenotypic plasticity. In: Agrawal, A.A.; Tuzun, S.; Bent, E., eds. Inducible plant defense against pathogens and herbivores: biochemistry, ecology, and agriculture. St. Paul, MN: American Phytopathological Society Press: 251-266.

Aspinwall, M.J.; King, J.S.; Booker, F.L.; McKeand, S.E. 2011. Genetic effects on total phenolics, condensed tannins and non-structural carbohydrates in loblolly pine (Pinus taeda L.) needles. Tree Physiology. 31: 831-842.

Blanche, C.A.; Lorio, P.L. Jr.; Sommers, R.A.; [and others]. 1992. Seasonal cambial growth and development of loblolly pine: xylem formation, inner bark chemistry, resin ducts, and resin flow. Forest Ecology and Management. 49: 151-165.

Bongarten, B.C.; Teskey, R.O. 1987. Dry weight partitioning and its relationship to productivity in loblolly pine seedlings from seven sources. Forest Science. 33: 255-267.

Booker, F.L.; Maier, C.A. 2001. Atmospheric carbon dioxide, irrigation, and fertilization effects on phenolic and nitrogen concentrations in loblolly pine (Pinus taeda) needles. Tree Physiology. 21: 609-616.

Danielsson, M.; Lundén, K.; Elfstradn, M. [and others]. J. 2011. Chemical and transcriptional responses of Norway spruce genotypes with different susceptibility to Heterobasidion spp. infection. BMC Plant Biology. 11: 154-169.

Datta, J.; Lal, N. 2012. Temporal and spatial changes in phenolic compounds in response to Fusarium wilt in chickpea and pigeonpea. Cellular and Molecular Biology. 58: 96-102.

Dickson, R.E. 1991. Assimilate distribution and storage. In: Raghavendra, A.S. ed. Physiology of trees. New York: John Wiley & Sons: 51-85.

Gebauer, R.L.E.; Strain, B.R.; Reynolds, J.F. 1998. The effect of elevated CO_2 and N availability on tissue concentrations and whole plant pools of carbon-based secondary compounds in loblolly pine (Pinus taeda). Oecologia. 113: 29-36.

Lattanzio,V.; Lattanzio, V.M.T.; Cardinali, A. 2006. Role of phenolics in the resistance mechanisms of plants against fungal pathogens and insects. In: Phytochemistry: Advances in research. Kerala, India: Research Signpost: 23-67.

Li, B.; Allen, H.L.; McKeand, S.E. 1991. Nitrogen and family effects on biomass allocation of loblolly pine seedlings. Forest Science. 37: 271-283.

Litton, C.M.; Raech, J.W.; Ryan, M.G. 2007. Carbon allocation in forest ecosystems. Global Change Biology. 13: 2089-2109.

NDMC. 2015. National Oceanic and Atmospheric Agency National Drought Mitigation Center. http://droughtmonitor.unl.edu/MapsAndData/ChangeMaps.aspx. [Date accessed: May 8, 2015].

Osbon, B.S.; Blazier, M.A.; Tyree, M.C.; Sword-Sayer, M.A. 2012. Whole-canopy gas exchange among four elite loblolly pine seed sources planted in the Western Gulf Region. In: Butnor, J.R., ed. Proceedings of the 16th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 233-234.

Singh, A. 2012. Variation in resistance of loblolly pine (Pinus taeda L.) and slash pine (P. elliottii Englem.) families against Leptographium and Grosmannia root fungi. Auburn, AL: Auburn University. 141 p. M.S. thesis.

Singleton, V.L.; Rossi, J.A., Jr. 1965. Colorimetry of total phenolics with phosphomolybdic-phosphotungstic acid reagents. American Journal of Enology and Viticulture. 16: 144-158.

Stovall, J.P.; Fox, T.R.; Seiler, J.R. 2013. Allometry varies among 6-year-old Pinus taeda (L.) clones in the Virginia Piedmont. Forest Science. 59: 50-62.

Tang, Z.; Chambers, J.L.; Sword, M.A.; Barnett, J.P. 2003. Seasonal photosynthesis and water relations of juvenile pine relative to stand density and canopy position. Trees. 17: 424-430.

Vose, J.M.; Allen, H.L. 1988. Leaf area, stemwood, and nutrition relationships in loblolly pine. Forest Science. 34: 547-563.

Witzell, J.; Martín, J.A. 2008. Phenolic metabolites in the resistance of northern forest trees to pathogens-- past experiences and future prospects. Canadian Journal of Forest Research. 38: 2711-2727.

A PRELIMINARY MODEL OF YELLOW-POPLAR SEEDLING ESTABLISHMENT TWO YEARS AFTER A GROWING SEASON PRESCRIBED FIRE IN SOUTHERN APPALACHIAN OAK STANDS

W. Henry McNab[1]

Abstract—Factors affecting the density and distribution of yellow-poplar regeneration after a single growing season prescribed fire were studied in mature upland oak stands in the southern Appalachian Mountains. In burned and unburned stands, density of one and two year old yellow-poplar seedlings was inventoried within 50 m from isolated yellow-poplar canopy seed trees in response to distance from seed source, litter layer present and competition by other species. Yellow-poplar regeneration was absent on the forest floor in unburned stands, but averaged over 97 thousand seedlings/ha around seed trees in burned stands. Correlation analysis indicated yellow-poplar seedling density decreased with increasing leaf litter on the forest floor, increasing distance from the source seed tree and increasing competition by seedlings of other species. A parsimonious Poisson regression model using Akaike Information Criterion for variable selection included (in order of relative importance) percent forest floor covered by leaf litter and distance from the seed source. Results of this study suggest that prescribed fire may promote establishment of yellow-poplar regeneration where seed trees are a component of upland oak stands.

INTRODUCTION

Yellow-poplar (*Liriodendron tulipifera*) is a wind-disseminated, shade intolerant, highly productive pioneer species of mesic sites that typically dominates lower slopes and coves in the Southern Appalachian Piedmont and Mountains (Olsen 1969). Seed production by this species is abundant, dependable annually and accumulates in the forest floor for up to seven years (Carvell and Korstian 1955). Disturbance of the canopy and forest floor from silvicultural activities, such as harvesting, typically results in establishment of dense yellow-poplar regeneration that grows rapidly in height and typically excludes other, slower growing, desirable species (Beck and Hooper 1986).

Although yellow-poplar is typically classified as a mesophytic species (Olsen 1969) seedlings can become established following disturbance on dry sites with adequate soil moisture, such as during years of above average summer precipitation, and particularly on sites where soil temperature is high, up to 35°C. (Stephens 1965). In contrast, Whipple (1968) reported the combination of drought and high soil temperature (>38°C) on disked and bulldozed sites in north Alabama resulted in high mortality of new yellow-poplar seedlings. On some dry sites, yellow-poplar can become an unintended competitor with other desirable regeneration following harvests of mixed oak (*Quercus spp*) and hickory (*Carya spp*) stands (McGee 1975). Prescribed fire is an effective method for controlling competition from yellow-poplar saplings when regenerating oaks on Piedmont sites (Brose and others 1999). Other studies, however, suggest burning may promote yellow-poplar regeneration from stored seeds in the forest floor, particularly on moist sites (Shearin and others 1972, McNab and others 2013) in the southern Appalachians. With increasing use of prescribed burning for restoration of fire tolerant vegetative communities in the southern Appalachians (Brose and others 2001), information is lacking on its effects on establishment of yellow-poplar seedlings, particularly on dry sites dominated by oaks where codominant seed trees may be present as a result of past disturbances.

I utilized the design and several treatments of a large study on the effects of season of prescribed burning on vegetation in upland oak stands. My objectives were to determine if the density and distribution of yellow-poplar seedlings near seed trees were affected by (1) prescribed burning, (2) distance from a seed source, (3) amount of leaf litter on the forest floor and (4) competition from regeneration of other species. Although my study utilizes replicated treatment units, I characterize it as preliminary because my research questions were to determine if yellow-poplar regeneration differs significantly between unburned and burned upland oak stands, and to evaluate some of the important factors that should be examined in subsequent investigations of broader scope.

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METHODS

Study Area

The study is in the Bent Creek Experimental Forest, located in the Pisgah Ranger District of the Pisgah National Forest, in the Southern Appalachian Mountains of western North Carolina (35.5°N, 82.6°W). The study area occupies 47.5 ha on the southerly exposure of a low, east-west trending ridge with elevations ranging from 700 - 730 m. Soils are primarily Ultisols that formed from gneisses and schists in the Late Proterozoic Ashe Metamorphic Suite. Soils in the study area consist predominantly of two map units: a complex of Evard-Cowee series (Typic Hapludults) on ridges and middle slopes and Tate series (Typic Hapludults) on lower slopes and in coves. Soils in both map units are deep (>80 cm), acidic (pH<5.5) and relatively infertile. Annual precipitation averages 125 cm and is uniformly distributed, 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 (*Quercus spp.*) and hickories (*Carya spp.*). Oaks include scarlet (*Q. coccinea*), chestnut (*Q. prinus*), southern red (*Q. falcata*), white (*Q. alba*) and black (*Q. velutina*); hickories are mockernut (C. tomentosa) and pignut (C. glabra). A shade tolerant midstory consists of red maple (*Acer rubrum*), black gum (*Nyssa sylvatica*), sassafras (*Sassafras albidum*) and sourwood (*Oxydendron arboreum*). The shrub layer consists of tall (2-4 m) evergreen mountain laurel (*Kalmia latifolia*) and short (<0.5 m) deciduous hillside blueberry (*Vaccinium vacilians*). Yellow-poplar is a minor and variable canopy component throughout the stand, occurring with an estimated mean stem density of about 1 ha^{-1} with greater frequency on lower slopes and in drainages. The presence of scattered, shade-intolerant, shortleaf pines (*Pinus echinata*) in the canopy suggests the area was used as a woodlot and pasture for livestock when farming ceased and old-field succession began (Nesbitt 1941). There is no record or evidence of fire in the study area. Arborescent vegetation on the study area likely became established around 1900, when the land was acquired by the Biltmore Estate following approximately 50 years of subsistence farming.

Study Design and Sampling

The 47.5 ha study area was subdivided into nine treatment units of approximately uniform size, each of which extended from ridge top to lower slope position. The core prescribed burning study was a completely randomized design consisting of three replications of three treatments: (1) no burning (control), (2) growing season burn and (3) dormant season burn; the latter treatment was not utilized in this investigation (fig. 1). The growing season burn treatment was applied to the three replication units sequentially on the same day in late April 2013, using a strip head fire ignition method beginning on the ridge and extending down the slope at about 10 m intervals along the contour. The fire consumed the entire upper layer of the 1 to 2 year old, generally non-decomposed leaf litter (the O_1 - horizon) and varying amounts of the compact, decomposed duff (the O_2 horizon above the mineral soil) depending on local variation of fuel loading, fuel moisture content, wind velocity and convergence of flame zones between two fire types consisting of heading and backing flame zones. Fire intensity ranged from 19 to 80 kcal/sec/m as estimated from crown scorch of eastern white pine (*Pinus strobus*) foliage, which averaged about 3 m and ranged from 2 m to 7 m (VanWagner 1973).

In October 2014, two growing seasons after the burn treatment in April 2013, I located all codominant yellow-poplar trees in the six unburned and burned treatment

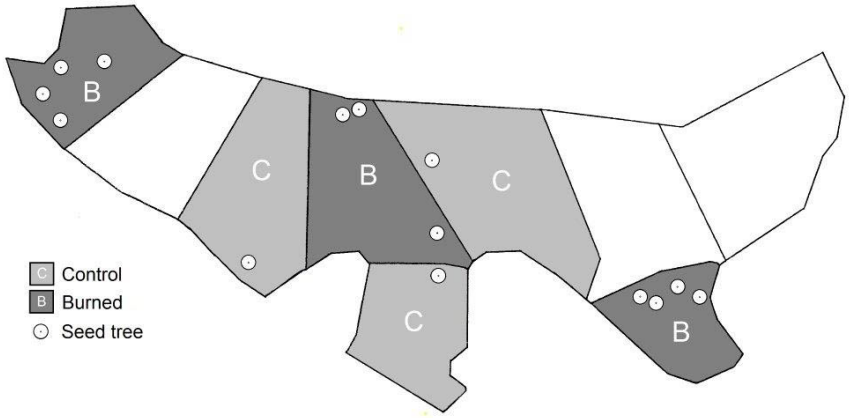


Figure 1—Study area in Bent Creek Experimental Forest showing unburned control and prescribed burned treatment units and approximate locations of yellow-poplar seed trees mentioned in the text.

units and evaluated them as candidate sample sites for my study. Ideally, seed trees would have been widely separated (>200 m) but the scarcity of suitable trees in some units resulted in some being separated by only 100 m. Although the seed shadow of yellow-poplar trees can extend 100 m or more (Engle 1960), 90% of seed fall occurs within 50 m of the seed tree (Carvell and Korstian 1955). Whipple (1968) found that yellow-poplar seed dispersal was strongly affected by wind direction and most seeds fell within 2 to 5 times the height of seed trees. I measured diameter breast height (dbh), total height, and basal area of the surrounding stand for each suitable seed tree in the six treatment units.

I counted live yellow-poplar seedlings at 30 randomly located sample plots (0.44 m^2) within 50 m radius of each seed tree. Sample plots were excluded that were located outside of the treatment unit (if the tree was near a unit boundary) or in disturbed areas such as a fire line or a trail. The number of sample plots was reduced around several seed trees because of their proximity to other seed trees or location near the edge of a treatment. Data were collected for the following variables at each sample plot: (1) density (n/ha) of yellow-poplar seedlings (designated hereafter as DEN), (2) distance (m) from the seed tree (DIS), (3) density (n/ha) of competition (COM) consisting of the pooled number of seedlings and sprouts of all species excluding yellow-poplar and (4) estimated percent of the pre-burn (O_1 and O_2) litter layer (LIT) present on the forest floor.

Data Analysis

I used two-tailed t-tests with assumed unequal variances to compare size and competition (i.e. surrounding stand basal area) characteristics of the yellow-poplar seed trees in the two treatments. For testing treatment effects on the four response variables (e.g. DEN, LIT, DIS and COM) I pooled sample plot data by unit and seed tree and used the nonparametric Kolmogorov-Smirnov test to compare means. Observations during field data collection suggested non-normal distribution of regeneration DEN for sample plots in the unburned control treatment (i.e. seedling counts on nearly all plots were zero). Levenes test indicated inequality of sample population variances for DEN between the control and burned treatments. Spearman rank correlation was used to determine the relationship of yellow-poplar seedling density with each of the three independent variables of DIS, LIT and COM. I used multiple Poisson regression to model the individual and collective relationships of the count of yellow-poplar seedlings on each sample plot in response to variation of LIT, DIS and COM on burned treatment units. For the Poisson analysis of DEN as a function of LIT, DIS, and COM, I considered the random sample sites around the seed trees as independent observations that responded to variation of fuel loading, fire intensity and other factors, and therefore did not pool my data. Akaike's Information Criterion (AIC) was used as a guide for ranking model formulations consisting of each variable and combinations of variables. Because AIC is not a measure of significant improvement of one Poisson model compared to another, the reduction of deviance between two competing models was evaluated for significance using the chi-square test statistic. Finally, a measure of goodness of fit of various model formulations was calculated using deviance of regression (i.e. $R^2 = 1 - (SS\ model / SS\ total)$). R statistical software was used for analysis (R Core Team 2014). Statistically significant differences were tested at the alpha = 0.05 level.

RESULTS AND DISCUSSION

Yellow-poplar Seed Trees

Fourteen yellow-poplar seed trees were suitable for data collection; 3 in control treatments and 11 in the prescribed fire treatments (fig. 1). Seed trees were significantly smaller in dbh in the control stand compared to the burned stand (37.5 cm vs 52.0 cm), however there was no difference in total height (31.2 m vs 30.8 m) (table 1). Mean basal area of the predominantly oak stand around the yellow-poplar seed trees was significantly higher in the control treatment (26.8 m^2/ha) compared to the burn treatment (19.1 m^2/ha). None of the seed trees in the burned units showed evidence of heat damage to the crown or bole.

Table 1—Mean (SD) of yellow-poplar seed tree dbh, height and surrounding stand basal area by treatment

Parameter	Treatment Control (n=3)	Burn (n=11)
Dbh (cm)	37.5 (4.5)	52.0 (17.1)[a]
Height (m)	31.2 (2.3)	30.8 (6.4)
Basal area (m^2/ha)	26.8 (1.6)	19.1 (3.4)[a]

[a]Significantly different from control at $p < 0.05$ level by two-tailed t-test with unequal variances.

Table 2—Mean (SD) density of yellow-poplar regeneration, leaf litter cover on forest floor, distance sampled from seed tree and competition from other species by prescribed burn treatment

Variable	Treatment Control (n=82)	Burn (n=179)
Yellow-poplar density (n/ ha)	244.7 (423.8)	97,181.8 (56,447.2)[a]
Litter cover (%)	99.9 (0.2)	54.2 (13.8)[a]
Distance (m)	22.3 (4.0)	21.8 (3.2)
Competition (n/ ha)	59,266.7 (11,249.1)	39,118.2 (12,235.9)

[a]Significant treatment difference at $p < 0.05$ by Kolmogorov-Smirnov test.

Seed production, as a prerequisite for seedling establishment, by the sample trees was not quantified, but was estimated from a relationship with dbh developed by Carvell and Korstian (1955) in the Piedmont of North Carolina. Estimated mean annual production of sound yellow-poplar seeds was 43,500 /ha and 75,600 /ha in the control and burned treatments respectively. Because yellow-poplar seeds can remain viable up to 7 years in leaf litter on the forest floor (Carvell and Korstian 1955), the potential total supply of sound seeds available for regeneration was approximately 283,000 seeds/ha in the control treatment and 492,000 seeds /ha in the burned treatment, assuming a conservative viability period of 5 years. Although potential seed production was less in the control units than in the burned units that fact alone does not appear to be a plausible explanation for the nearly complete lack of yellow-poplar regeneration.

Control vs Burn Treatments Effects

Histograms of density distributions of the two populations of yellow-poplar seedlings suggested different variances for the control vs burn treatments. Levene's test indicated significant heteroscedasticity of population variances for yellow-poplar DEN, between the two treatments. The non-parametric Kolmogorov-Smirnov test indicated a significant difference of mean yellow-poplar DEN on the control treatment (245 seedlings/ha) compared to the burn (97,182 seedlings/ha) (table 2). The leaf litter on the forest floor was greater for the control treatment (99.9 percent) than for the burn treatment (54.2 percent). I found no difference between the control and burn treatments for DIS (22.3 m vs 21.8 m) and COM (59,267 seedlings/ha vs 39,118 seedlings/ha).

The almost complete lack of yellow-poplar DEN of any size in the control treatment was unexpected, but not surprising considering silvical characteristics of the species. Olsen (1969) reported that light and adequate moisture enhance seed germination and development of seedlings. On the 82 sample plots examined in the three units of the control treatment, only one seedling was found; it was on bare soil of an upturned root system of a wind thrown tree. Apparently the combination of thick, undisturbed leaf litter in combination with reduced light on the forest floor was sufficient to retard seed germination. Observations elsewhere in the control treatments outside of sample plots were similar, except that seedlings were occasionally observed on exposed soil in the pit of recently wind thrown trees. Clark and Boyce (1964) reported that yellow-poplar seeds can germinate on an undisturbed forest floor but seldom survive beyond one year.

Yellow-poplar Regeneration

Yellow-poplar DEN in the burn treatment was strongly correlated with LIT ($r = -0.45$, $p < 0.01$, $n = 179$) and DIS ($r = -0.26$, $p < 0.01$, $n = 179$), and weakly correlated with COM ($r = -0.15$, $p < 0.05$, $n = 179$) (table 3). The negative relationship indicated seedling numbers decreased with increasing LIT, DIS, and COM. There were no significant correlations among the three independent variables in the burn treatment. In the control treatment the almost perfect and highly significant correlation of DEN with LIT ($r = -0.99$, $p < 0.01$, $n = 82$) is an artifact of the field data, where a single seedling was recorded on a microsite with reduced leaf litter. Because of the spurious relationship of yellow-poplar regeneration with the control treatment, I omitted it from further investigation in this study.

I used Poisson multiple regression to assess interrelationships of yellow-poplar DEN in the burn treatment with the three significant independent variables (table 4). I began model development for prediction of DEN with trial formulations using each of the three factors. Yellow-poplar DEN was significantly affected by each factor, ranging from the weakest relationship for COM, which explained about 4 percent of variation, to LIT, which accounted for 24 percent. Including either COM or DIS with LIT resulted in significantly better prediction models. The decrease of AIC for the three variable model suggested it was an appropriate parsimonious formulation. However, the chi-square test of deviance reduction indicated the three

Table 3—Spearman rank correlation coefficients among study variables measured on sample plots within 50 m of yellow-poplar seed trees on control treatment (upper diagonal, n = 82) and the burned treatment (lower diagonal, n = 179)

Variable	Seedling density	Litter cover	Distance	Competition
	n/ha	%	m	n/ha
Seedling density (n/ha)	–	-0.99[a]	-0.02	0.17
Litter cover (%)	-0.45[a]	–	0.02	-0.17
Distance (m)	-0.26[a]	-0.06	–	-0.14
Competition (n/ha)	-0.15[a]	0.10	0.11	–

[a]$p < 0.05$ (two-tailed)

Table 4—Comparison of sequential Poisson models based on Akaike's Information Criterion (AIC), deviance explained and pseudo R^2 for yellow-poplar seedling density two-years after a growing season prescribed burn in upland oak forests of the southern Appalachian Mountains (n=179)

Model formulation	df	AIC[a]	Deviance	R^{2b}
Null	178	—[c]	11903.5	—
COM[d]	177	12353.7	11484.1[e]	0.04
DIS	177	11738.6	10869.0[e]	0.09
LIT	177	9937.4	9067.7[e]	0.24
LIT+COM	176	9731.2	8859.6[e]	0.26
LIT+DIS	176	8624.6	7753.0[e]	0.35
LIT+DIS+COM	175	8517.9	7644.3	0.36

[a]Akaike Information Criterion; smaller values indicate a preferred model with fewest variables that provides an adequate fit to the data; models are ordered by decreasing AIC.

[b]Pseudo R^2 calculated as 1-(SS model/SS total).

[c]Not applicable.

[d]COM, competition from regeneration of species other than yellow-poplar; DIS, distance from the seed tree to a maximum of 50 m; LIT, leaf litter remaining on the forest floor post burn.

[e]Deviance explained by this model formulation differs significantly from the model above; chi-square $p < 0.05$.

variable model was not significantly better ($p = 0.0502$) than the best two-variable model, which included only LIT and DIS, and accounted for approximately 35 percent of the variation of DEN.

The relationship of yellow-poplar seedling density with DIS and LIT is shown in Figure 2. The model trend lines suggest that yellow-poplar seedlings could be established at distances farther than 50 m from seed trees, which agrees with my field observations. The best model also supports my field observations that competition from seedlings and sprouts of other species was a variable minimal influence of DEN on some sample sites, but was not a consistent significant source of variation at the end of the second growing season. However, because of the likely rapid growth rate of competition from sprouts, particularly red maple and oaks, coupled with canopy shade on the forest floor and intolerant yellow-poplar seedlings, COM will probably be a factor of subsequent greater importance.

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

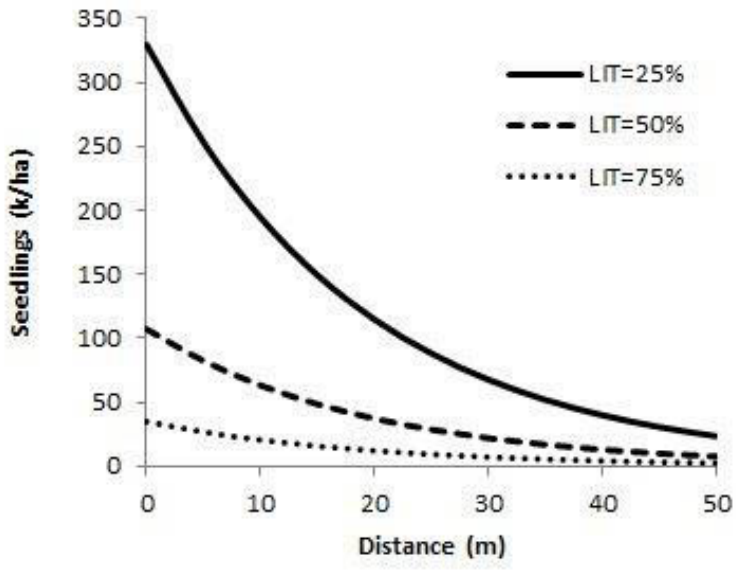


Figure 2—Estimated density of yellow-poplar regeneration following a single growing season prescribed burn in relation to distance from seed trees and three levels of litter cover (LIT25%, LIT50%, LIT75%) on the forest floor.

stored seeds in the forest floor. Although much research has been directed toward description of yellow-poplar seed shadows and quantification of seed production, this study is among the first to use a designed study with replicated treatments to identify factors affecting establishment of regeneration, particularly in relation to prescribed burning. Of the three variables investigated in this study affecting the density of one and two year old yellow-poplar regeneration after a single prescribed fire, the amount of leaf litter on the forest floor was found to be the most important explanatory factor.

It is doubtful that yellow-poplar regeneration will be competitive on this dry site, particularly because shade from the over story and sprout competition will likely retard height growth of this intolerant species. This study does demonstrate the potential for establishment of yellow-poplar regeneration from scattered seed trees in a predominantly oak stand, particularly if silvicultural activities were planned following burning, such as thinning or harvest. If yellow-poplar regeneration is not desirable on dry sites such as this, then a series of prescribed burns will be needed to control the currently established seedlings, exhaust remaining seeds that might be stored in litter on the forest floor and prevent accumulation of seeds in the forest floor resulting from annual production.

Results from my preliminary study leaves many questions unanswered. For example, will the one and two year old yellow-poplar seedlings have high mortality rate during beneath the closed oak canopy in following years? Is the observed regeneration of yellow-poplar seeds typical or the result of a fortuitous combination of above average seed production followed by above average precipitation, which allowed unusually high seedling germination? Was the buried seed bank exhausted by a single burn, or is a series of closely spaced repeat burns needed to accomplish that? What is the rate of seedling establishment on burned seedbeds from annual yellow-poplar seed fall? Also, because so little of the variation of yellow-poplar regeneration was explained by the variables examined in this study (about 35 percent), future study should include other variables potentially influencing regeneration of yellow-poplar. Follow up study is particularly desirable at other locations distant from Bent Creek Experimental Forest, with different soils and weather conditions during the growing season.

ACKNOWLEDGMENTS

I am grateful for a suggestion by former project leader, C.H. Greenberg, to visit a remote part of a treatment unit used in this study to view site conditions in a large canopy gap, which resulted from tree mortality caused by high fire intensity. Information presented in this paper resulted from a serendipitous observation of yellow-poplar seedlings while going to the remote location.

LITERATURE CITED

Beck, D.E.; Hooper, R.M. 1986. Development of a Southern Appalachian hardwood stand after clearcutting. Southern Journal of Applied Forestry. 10: 168-172.

Brose, P.H.; Van Lear, D.H.; Keyser, P.D. 1999. A shelterwood-burn technique for regenerating productive upland oak sites in the Piedmont region. Southern Journal of Applied Forestry. 16: 158-163.

Brose, P; Schuler, T.; VanLear, D.; Berst, J. 2001. Bringing fire back: the changing regimes of the Appalachian mixed-oak forests. Journal of Forestry. 99: 30-35.

Carvell, K.L.; Korstian, C.F. 1955. Production and dissemination of yellow-poplar seed. Journal of Forestry. 53: 169-170.

Clark, F.B.; Boyce, S.G. 1964. Yellow-poplar seed remains viable in the forest litter. Journal of Forestry. 62: 564-567.

Engle, L.G. 1960. Yellow-poplar seedfall pattern. Station Note 143. Columbus, OH: U.S. Department of Agriculture Forest Service, Central States Forest Experiment Station. 2 p.

McGee, C.E. 1975. Regeneration alternatives in mixed oak stands. Research Paper SE-125. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 8 p.

McNab, W.H.; Berg, E.C.; Oprean, III, T.M. 2013. Response to prescribed burning of five year old hardwood regeneration on a mesic site in the southern Appalachian Mountains. In: Guldin, J. ed. Proceedings of the 15th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-175. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 513-517.

Nesbitt, W.A. 1941. History of early settlement and land use on the Bent Creek Experimental Forest, Buncombe County, N.C. Unpublished report. On file with: Southern Research Station, Upland Hardwood Ecology and Management Research Work Unit, 1577 Brevard Road, Asheville, NC 28806. 78 p.

Olsen, Jr., D.F. 1969. Silvical characteristics of yellow-poplar (Liriodendron tulipifera L.). Research Paper SE-48 Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 16 p.

R Core Team. 2014. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Shearin, A.T., Bruner, M.H.; Goebel, N.B. 1972. Prescribed burning stimulates natural regeneration of yellow-poplar. Journal of Forestry. 70: 482-484.

Stephens, G.R., Jr. 1965. Yellow-poplar seedlings respond to soil temperature. Journal of Forestry. 63:7 01-703.

Van Wagner, C.E. 1973. Height of crown scorch in forest fires. Canadian Journal of Forest Research. 3: 373-378.

Whipple, S.D. 1968. Yellow poplar regeneration after seed tree cutting and site preparation. Bulletin 384. Auburn, AL: Auburn University, Agricultural Experiment Station. 15 p.

ESTIMATING CARBON STOCKS IN UNEVEN-AGED BOTTOMLAND HARDWOOD FOREST STANDS IN SOUTH LOUISIANA

Matthew K. Moerschbaecher, Richard F. Keim, and John W. Day[1]

Abstract—This study reports growth and carbon storage of four bottomland hardwood forest sites in the Lower Mississippi Alluvial Valley (LMAV) of southern Louisiana. Forest growth and carbon sequestration rates at the four sites were highly variable because of differences in stand composition, age, structure, and site hydrology. Mean annual carbon assimilation rates ranged from 1.9 - 3.4 Mg/ha/yr during a seven year period. The highest sequestration rates occurred in trees on the drier, ridge site. Remnant mature trees accounted for a large proportion of total carbon assimilated and stored and at each site. The carbon sequestration rates reported in this study are generally less than those reported by previous research of carbon storage capacity of bottomland hardwood forests in the LMAV.

INTRODUCTION

Bottomland hardwood (BLH) wetlands are riparian wetland forests located in river floodplains and seasonally wet areas throughout the southeastern U.S. (Hodges 1997). The typical location of BLH forests in floodplains causes seasonal flooding and sediment deposition that contributes to highly productive soils. The largest expanse of BLH forests in the southeastern US are in the Lower Mississippi Alluvial Valley (LMAV). Historically, the LMAV contained approximately 10 million hectares of BLH forests, but now only 2.8 million hectares remain because of widespread conversion to agriculture (King and Keeland 1999). The loss of BLH forests results in the loss of ecosystem service function values including carbon storage, water quality, wildlife habitat, and flood control (Jenkins and others 2010). Federal and state conservation programs have recently encouraged afforestation to restore BLH forest area mainly for restoration of wildlife habitat (Gardiner and Oliver 2005), and the economics of carbon sequestration have also attracted the interest of landowners and private investors (Shoch and others 2009).

Bottomland hardwood species distribution within floodplains is largely the result of floodplain microtopographic variation. These minor topographic variations have a large effect on site hydrology, which is the major allogenic force determining species presence and natural patterns of ecological succession and productivity (Hodges 1997). Although BLH forests are highly productive forest ecosystems (Messina and Conner 1998) with a high capacity to sequester carbon (Brinson 1990), there is a scarcity of published growth and yield studies, and conflicting results for those studies that do exist (Shoch and others 2009). For example, some research suggests high productivity in transitional BLH forests located between cypress-tupelo swamps and upland forests (Conner and Day 1976; Taylor and others 1990), while other studies have shown the physiological stresses of anaerobic soils and drought outweigh the benefits of periodic flooding (Mitsch and Rust 1984; Megonigal 1997).

Much of the published work with regard to the sequestration capacity of these forests has focused on afforested stands and plantation forests. However, cutover and naturally regenerated stands make up the majority of bottomland hardwood forests in the LMAV. These stands are not amenable to simple modeling by age due to their structural diversity (Smith and others 2006; Shoch and others 2009; Covey and others 2012). Several studies have documented carbon storage of even aged stands, forest plantations, and naturally regenerated stands of known ages (Smith and others 2006; Heath and others 2011) throughout the United Sates. Shoch and others (2009) examined carbon storage rates in BLH forests of northern reaches of the LMAV and concluded that the Smith and others (2006) table for the south-central U.S. region underestimates carbon storage in live-tree biomass of bottomland hardwood stands in the LMAV aged 20-90 years. A full understanding of forest growth and carbon storage in bottomland sites requires more information from natural stands.

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The purpose of this study was to measure carbon stocks and sequestration rates in bottomland hardwood stands in the LMAV in south Louisiana. We hypothesized that forest stands across varying micro topographical environments store carbon at different rates based on stand characteristics and site hydrology. We compare carbon stored in live aboveground woody biomass at the stands using two sets of allometric equations.

Field-Site Description

This research was carried out at two bottomland hardwood sites near Baton Rouge and St. Gabriel, Louisiana, owned by the Louisiana State University (LSU) Agricultural Center (fig. 1). Both sites are in the floodplain of the Mississippi River but have not been flooded by river water for more than a century because of flood control levees. Flooding at the research sites is by rainwater and stormwater runoff from the surrounding uplands.

The Baton Rouge site consists of approximately 300 acres of forest at the Ben Hur Agricultural and Forest research station, approximately 3 km south of the LSU main campus, and is bordered by agricultural fields and private residential developments. Two general forest habitat types—ridge and swale—exist at the site, with tree species present heavily influenced by local hydrology. The ridge habitat (BHR) is dominated by oaks, hickories, and sugarberry while the swale habitat (BHS) is dominated by cypress and tupelo. The sites are mixed-species, multicohort, uneven-aged stands in the understory reinitiation stage of stand development (Oliver and Larson 1996). Ridges and swales are well defined and occur successively approximately every 25-30 m so that that the swales are functional drainages while the ridges are rarely breached by floodwaters. Water enters the site in the form of precipitation and from the overflow of a drainage ditch that runs along the western edge of the forest adjacent to a suburban housing development. The swale site is composed of poorly-drained Schriever clay (very-fine, smectitic, hyperthermic Chromic Epiaquert) typical of meander scar landforms. Parent material is clayey alluvium to 152 cm and the typical depth to water table for this series is 0-61 cm (NRCS 2011). The ridge site is composed of Thibaut clays (clayey over loamy, smectitic over mixed, superactive, nonacid, hyperthermic Vertic Epiaquept) that are the dominant soil type of ridges formed from clayey over loamy alluvium parent material. Typical depth to water table for this series is 46-91 cm.

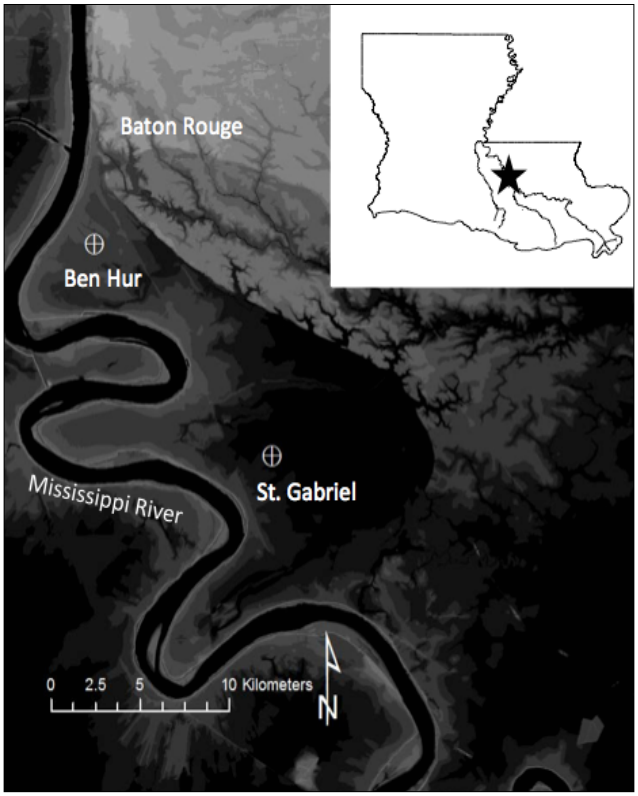


Figure 1—Map of study sites (crosses in circles) and surrounding area in Louisiana, USA. Background shading indicates elevation.

The St. Gabriel site is located approximately 18 km southeast of LSU main campus on the LSU AgCenter Reproductive Biology Center campus. The approximately 200 acre forest is composed of mixed bottomland hardwood species that has not been managed in recent years. The area was heavily logged or cleared for pasture in the middle part of last century and the study sites are composed of naturally regenerated multi-cohort stands in the stem exclusion stage of stand development (Oliver and Larson 1996). A number of spoil banks and canals affect hydrology and overall ecosystem function of the forest by impounding water during periods of high rainfall. As is the case at Ben Hur, species composition of the forest results from the frequency and duration of inundation. Impounded areas that are drier for longer periods, the St. Gabriel flats (SGF), are composed of mixed bottomland oaks (Quercus spp.), American elm (Ulmus americana), and sugarberry (Celtis leavigata), while the wetter areas, the St. Gabriel swale (SGS), are dominated by baldcypress (Taxodium distichum) and water tupelo (Nyssa aquatica). The micro topography at St. Gabriel is composed of much broader features than the well-defined ridges and swales at the Ben Hur site. The St. Gabriel sites are both classified as backswamp landforms composed of Schriever clay, but the swale site floods more frequently.

METHODS

Two pairs of 100 × 20 m plots were established at each site. Trees greater than 3cm in diameter were tagged and diameter at breast height (DBH) measured. Tree cores were taken from a random subsample of trees at each site to determine the ages of trees in the stand. The number of tree cores taken at each site was 14 (BHR), 18 (BHS), 16 (SGF), and 14 (SGS). Stem

production was estimated for 2009 and 2010 from annual changes in woody biomass calculated based on tree rings.

Tree biomass was estimated using allometric equations derived from Megonigal (1997) and Jenkins and others (2003). Carbon was calculated as 50 percent of tree biomass (Swift and others 1979). Two different sets of allometric equations were used because previous work has shown that differences in equation forms and species groupings may cause differences at small scales depending on tree size and forest species composition (Jenkins and others 2003). The equations used from Jenkins and others (2003) were for the mixed hardwoods species group and the oak/hickory species group.

Trees were cored with a 5.15-millimeter diameter increment borer to estimate recent growth rates. Cores were taken at breast height, dried at 60 °C, mounted on wooden frames, and sanded to expose tree rings. Ring width was measured to the nearest 0.01 mm on a sliding stage, and converted to basal area increment. Carbon storage was estimated based on accumulated basal area. The study did not account for tree mortality or other carbon pools such as litter, dead wood, non-woody vegetation, and soils.

RESULTS

Stand Structure

Study sites were composed of mixed bottomland hardwood communities typical of the LMAV. Diameter distributions were generally dominated by small trees (the "inverse J" shape; fig. 2) befitting natural stands with complex disturbance history.

Tree cores from each of the sites provided an estimate of the relative stand ages. The Ben Hur ridge site contained older trees compared to the other sites, several of which dated back to the early 20th century from 1899-1926. One tree at the Ben Hur swale site dated to 1876 and another to 1927 but the majority dated from the 1960s. The St. Gabriel flat site is composed of younger trees dating back to the 1970s with several older oaks dating back to the early 1960s. The St. Gabriel swale site consists of overstory trees that date back to the 1940s and 1950s.
Basal area at the Ben Hur sites increased by 1.3 m^2/ha from 2009 to 2010, while basal area increased by approximately half this amount at the St. Gabriel sites (table 1).

Carbon Storage

Total carbon storage in tree biomass increased from 2009 to 2010 at all sites (table 1). A greater increase

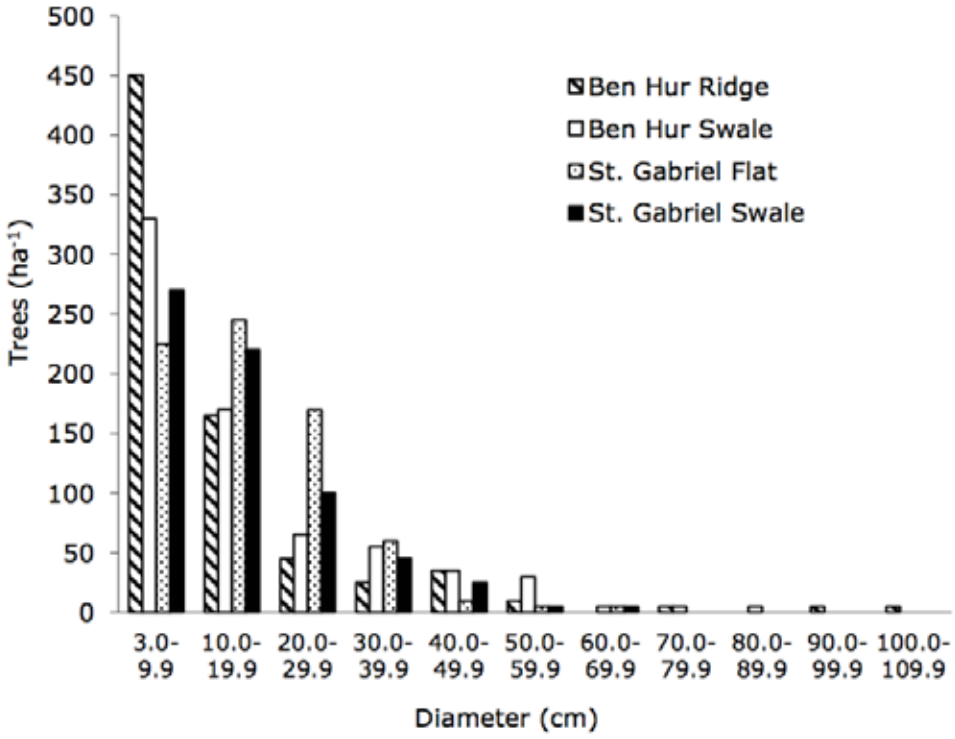


Figure 2—Distribution of trees by diameter at breast height for each site.

Table 1—Basal area and carbon storage at each of the study sites using allometric equations by Megonigal and others (1997) and Jenkins and others (2003)

	Basal Area		Megonigal-C			Jenkins-C		
	(m^3/ha)[1]		(Mg/ha)[1]		(Mg/ha/yr)[2]	(Mg/ha)[1]		(Mg/ha/yr)[2]
Site	2009	2010	2009	2010	2003-2010	2009	2010	2003-2010
Ben Hur Ridge	31.1	32.4	133	140	3.4	131	136	3.0
Ben Hur Swale	31.2	32.5	91	96	2.4	104	109	2.7
St. Gabriel Flat	23.6	24.3	74	76	2.3	72	73	2.3
St. Gabriel Swale	21.2	21.8	63	63	1.9	64	65	1.9

[1]Caclulated from diameter measurements
[2]Calculated from tree rings

in carbon storage was measured at the Ben Hur sites than at the St. Gabriel sites. Carbon storage in woody biomass was greater at the Ben Hur site than at the St. Gabriel site. The Ben Hur ridge site contained more carbon than the Ben Hur swale site. The St. Gabriel flat site contained more carbon stored in woody biomass than the St. Gabriel swale site.

The application of the Jenkins and others (2003) allometric equations led to a slightly higher estimate of carbon storage for the swale sites while estimated carbon storage at the Ben Hur ridge site and St. Gabriel flat site was higher when equations from Megonigal (1997) were applied. At all sites, carbon stored in the ten largest trees was greater than 50 percent of all live woody biomass carbon. The ten largest trees at the Ben Hur ridge site contained 73 percent of live woody biomass while the ten largest at the Ben Hur swale contained 54 percent. The proportion of carbon stored in the ten largest trees at the St. Gabriel flat site was 89 percent and 54 percent at the St. Gabriel swale site. The higher percentages of carbon in the ten largest trees at the Ben Hur ridge site and St. Gabriel flat site are a result of remnant mature trees present in the multi-cohort stands.

The uneven-aged stands examined in this study-sequestered carbon at variable mean annual rates ranging from 1.9 MgC/ha/yr to 3.4 MgC/ha/yr from 2003-2010. The ridge site sequestered carbon at the highest mean annual rate. Carbon sequestration rates at all the sites were within 1 MgC/ha of each other, on average, during the study period. Carbon sequestration rate variability within sites ranged from 0.2 – 0.6 MgC/ha/yr. Carbon sequestration and storage at the sites was the combined result of variable species composition within sites, differences in stand age and density across sites, and estimates varied depending on the allometric equation used.

Trends in BAI 2003-2009 were highly variable across sites by diameter class (fig. 3). However, similar patterns in tree growth size classes within sites occurred. In general, BAI was higher at BHR for all tree size classes than at any other site. The larger trees in the stand are generally growing at a faster rate than the smaller trees and are storing carbon in larger quantities and assimilating at a greater rate.

Estimated carbon storage rate in live woody biomass was greater using the Megonigal (1997) equations at the BHR and SGF site while the Jenkins and others (2003) equations produced greater estimates of carbon storage at BHS and SGS (fig. 3). The differences in these calculations were the result of species composition and tree sizes at the study sites. The Megonigal equations produced greater estimates for most species with the exceptions of bald cypress, water tupelo, Quercus species (oaks), and Carya species (hickories). The presence of bald cypress and overcup oak at the SGS site accounted for slightly higher carbon estimates when using the Jenkins equations. The presence of a greater number of large bald cypress and water tupelo at BHS led to higher estimates using the equations from Jenkins and others. (2003). The two sets of equations applied to the BHR site yielded approximately equal estimates of carbon storage.

Changes in carbon storage occurred at different rates across each of the sites (table 1). The mean annual rate of carbon storage was greatest at BHR, as was total change in carbon storage from 2003-2010. BHR was the only site where the Megonigal equation produced greater estimated rates of change and total change in carbon storage over time. Mean annual rates of carbon accumulation were within 0.5 MgC/ha at all sites for both sets of allometric equations. Total estimated carbon accumulation across the 7-year period at the Ben Hur site was higher using the Megonigal equation (23.4 MgC/ha) as opposed to the Jenkins equation (20.8 MgC/ha).

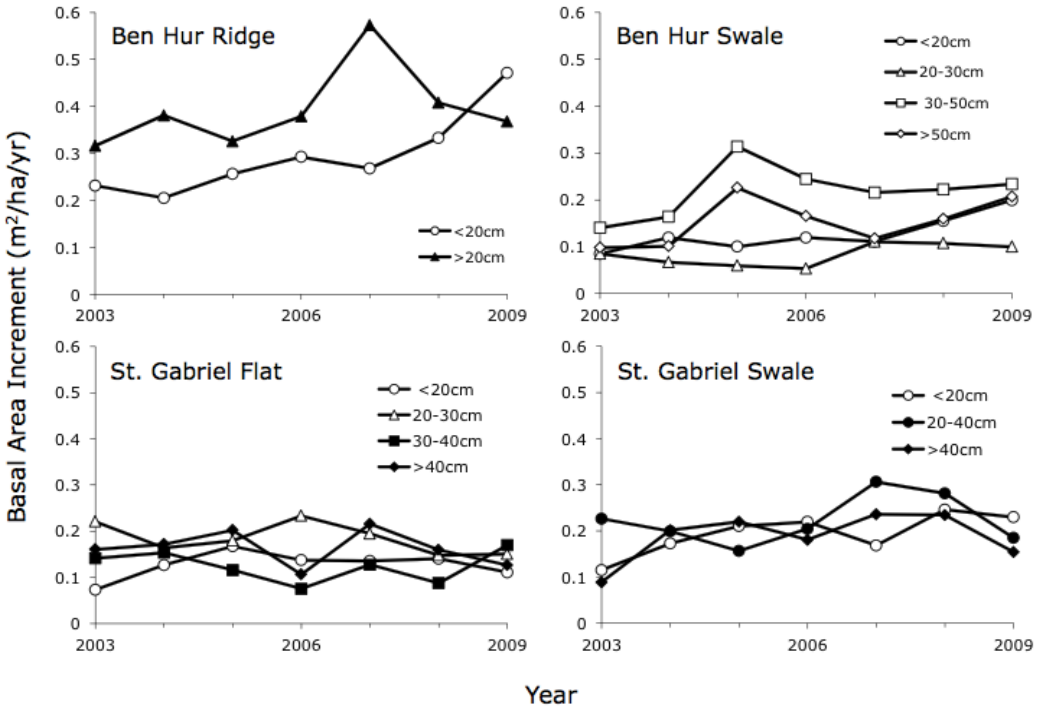


Figure 3—Basal area increment for each site reconstructed from tree rings.

DISCUSSION

There was no evidence in bias between the Megonigal and Jenkins equations, but results varied with respect to species composition at each of the sites. Variability of forest carbon sequestration rates is a result of several factors, including stand age, composition, density, and hydrology (Megonigal 1997; Ryan and others 2010). Based on this study, one hectare of bottomland hardwood forest stores between 63-140 MgC in live aboveground woody biomass.

Species composition and micro topographic variations were typical of the LMAV (Hodges 1997). Results of this study are consistent with previous studies, in that changing elevation by <2 meters had a marked difference on site quality, species occurrence, and stand development (Hodges and Switzer 1979; Wharton and others 1982; Hodges 1997).

Although it is not possible to determine effects on forest growth from one year of hydrologic data, the wettest site—SGS—had the lowest carbon storage in the standing crop of woody biomass. There were many downed and broken trees at this site, which suggests that the site may be transitioning toward more water tolerant species. However, forest community response to environmental change often occurs over a number of years (Conner and others 2011), thus more comprehensive, longer term monitoring of the sites is needed to accurately describe the relationship between hydrology and productivity.

Our study focused only on carbon stocks of living trees, but woody debris is an important storage component in many forest ecosystems (Harmon and others 1986). There are few published estimates of carbon in dead wood (i.e., standing dead, understory snags, and forest floor litter) in LMAV bottomland-hardwood forests. The best published estimates have been by Smith and others (2006) and Cochran (2008), which suggest that the volume of coarse woody debris may have ranged from 7-23 m^3/ha while fine woody debris ranged from 2-5 m^3/ha, and that dry sites have less woody debris than wet sites (Cochran 2008). Extrapolating from those values and results of our study suggests that living trees generally account for 69-94 percent of the total woody biomass in natural BLH stands.

Stand structure is an important determinant of carbon storage. This is especially the case with regard to uneven-aged cutover forests with remnant mature trees, as has been demonstrated in this study. These large trees are the most important loci of carbon storage; more than 50 percent of the carbon stored in live woody biomass occurred in the ten largest trees at all sites in this study. This concurs with Ryan and others (2010), who reported that large diameter trees tend to account for a large proportion of the aboveground biomass in mature forests. Thus, foresters managing similar forests for carbon storage should consider the standing crop of remnant mature trees.

The carbon storage rates reported in this study are low in comparison to previous research conducted to assess the carbon storage capacity of bottomland hardwood forests in the LMAV (fig. 4). Previous research by Shoch and others (2009) focused on even aged stands that followed a sigmoid growth curve in regards to age. Carbon storage of the uneven-aged trees in our study sites may be better represented by the estimates of carbon storage rates of oak-gum-cypress stands in the south-central U.S., as suggested by Smith and others (2006).

ACKNOWLEDGMENTS

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LITERATURE CITED

Brinson, M. 1990. Riverine forests. In: Lugo, A.E.; Brinson, M.; Brown, S., eds. Ecosystems of the World 15: Forested Wetlands. New York: Elsevier: 87-141.

Cochran, J.W. 2008. Coarse woody debris characteristics of managed and unmanaged bottomland hardwood forests. Baton Rouge, LA: Louisiana State University. 107 p. M.S. thesis.

Conner, W.H.; Day, J.W., Jr. 1976. Productivity and composition of a baldcypress water tupelo site and a bottomland hardwood site in a Louisiana swamp. American Journal of Botany. 63: 1354-1364.

Conner, W.H.; Sog, B.; Williams, T.M. [and others]. 2011. Long-term tree productivity of a South Carolina coastal plain forest across a hydrology gradient. Journal of Plant Ecology. 4: 67-76.

Covey, K.R.; Orefice, J.; Lee, X. 2012. The physiological ecology of carbon science in forest stands. In: Ashton, M.S.; Tyrrell, M.L.; Spalding, D. Gentry, B., eds. Managing forest carbon in a changing climate. Dordrecht, Netherlands: Springer: 31-49.

Gardiner, E.; Oliver, J. 2005. Restoration of bottomland hardwood forests in the Lower Mississippi Alluvial Valley, U.S.A. In: Stanturf, J.; Madsen, P., eds. Restoration of boreal and temperate forests. Boca Raton, FL: CRC Press: 235-251.

Harmon, M.E.; Franklin, J.F.; Swanson, F.J. [and others]. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research. 15: 133-302.

Heath, L.S.; Smith, J.E.; Skog, K.E. [and others]. 2011. Managed forest carbon estimates for the US greenhouse gas inventory, 1990–2008. Journal of Forestry. 109: 167-173.

Hodges, J.D.; Switzer, G.L. 1979. Some aspects of the ecology of southern bottomland hardwoods. In: North America's forests: gateway to opportunity, proceedings, 1978 conference of the Society of American Foresters and the Canadian Institute of Forestry. Washington, DC: Society of American Foresters: 360-365

Hodges. J.D. 1997 Development and ecology of bottomland hardwood sites. Forest Ecology and Management. 90: 117-125.

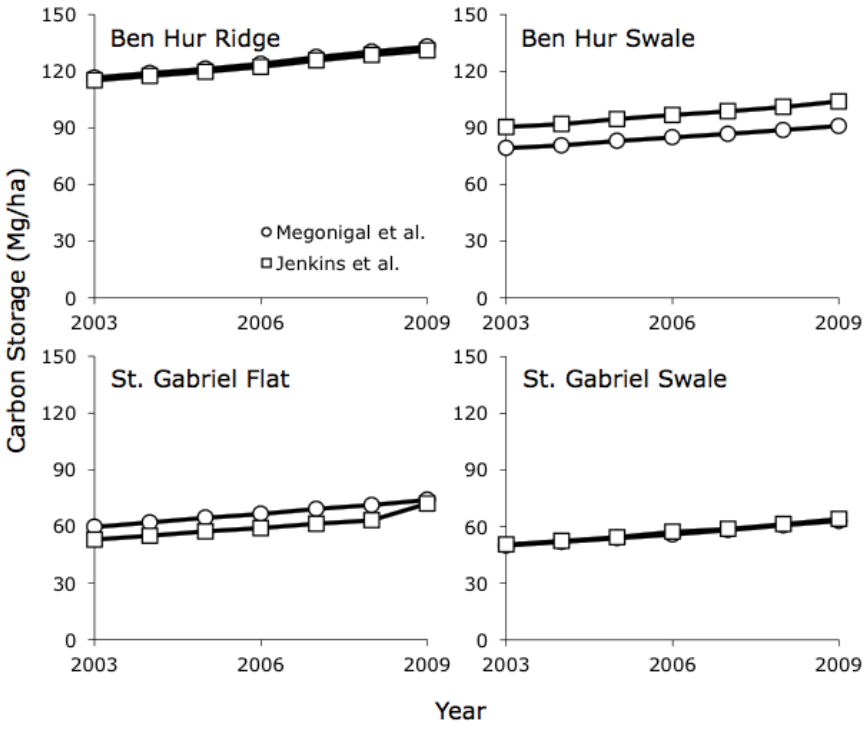


Figure 4—Carbon storage in live woody biomass at each site estimated using allometric equations by Megonigal and others (1997) and Jenkins and others (2003).

Jenkins, J.C.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A. 2003. National-scale biomass estimators for United States tree species. Forest Science. 49: 12-35.

Megonigal, J.P.; Conner, W.H.; Kroeger, S.; Sharitz, R.R. 1997. Aboveground production in Southeastern floodplain forests: a test of the subsidy-stress hypothesis. Ecology. 78: 370-384.

Messina, M.G.; Conner, W.H. 1998. Southern forested wetlands: ecology and management. Boca Raton, FL: Lewis Publishers, CRC Press LLC. 616 p.

Mitsch, W.J.; Rust, W.G. 1984. Tree growth responses to flooding in a bottomland forest in northeastern Illinois. Forest Science. 30: 499-510.

Oliver, C.D.; Larson, B.C. 1996. Forest Stand Dynamics. Update ed. New York: John Wiley and Sons, Inc. 520 p.

Ryan, M.G.; Harmon, M.E.; Birdsey, R.A. [and others]. 2010. A Synthesis of the science on forests and carbon for U.S. forests. Ecological Society of America: Issues in Ecology. 13: 1-16.

Shoch, D.T.; Kaster, G.; Hohl, A.; Souter, R. 2009. Carbon storage of bottomland hardwood afforestation in the Lower Mississippi Valley, USA. Wetlands. 29: 535-542.

Smith, J.E.; Heath, L.S.; Skog, K.E.; Birdsey, R.A. 2006. Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States. Gen. Tech. Rep. NE 343. Newtown Square, PA. U.S. Department of Agriculture Forest Service, Northeastern Research Station. 216 p.

Swift, M.J.; Heal, O.W.; Anderson, J.M. 1979. Decomposition in terrestrial ecosystems. Berkley, CA: University of California Press. 372 p.

Taylor, J.R.; Cardamone, M.A.; Mitsch, W.J. 1990 Bottomland hardwood forests: their functions and values. In: Gosselink, J.G.; Lee, L.C.; Muir, T.A., eds. Ecological processes and cumulative impacts: illustrated by bottomland hardwood wetland ecosystems. New York: Lewis Publishers: 5-86.

Wharton, C.H.; Kitchens, W.M.; Pendleton, E.C.; Sipe, T.W. 1982. The ecology of bottomland hardwood swamps of the Southeast: a community profile. FWS/OBS 81/37. Washington, DC: U. S. Department of the Interior Fish and Wildlife Service, Office of Biological Services. 133 p.

GROWTH ANALYSIS OF OAK TREES IN BOTTOMLAND HARDWOOD RESTORATION PLANTINGS IN THE LOWER MISSISSIPPI ALLUVIAL VALLEY

Rodrigo V. Leite, Brent R. Frey, and Jonathan D. Stoll[1]

Abstract—The Lower Mississippi Alluvial Valley (LMAV) once had the largest expanse of bottomland forest cover in the United States, but these diverse forests have been heavily cleared for agricultural purposes. In recent decades significant forest restoration efforts have been attempting to restore bottomland forest cover and the environmental services it provides (e.g. water quality, soil conservation, wildlife habitat). One of the most commonly planted groups of species in these afforestation efforts are different oak (*Quercus* spp.) species However, information about their growth and yield are lacking. The objective of this study is to collect information related to oak growth and yield in bottomland hardwood restoration plantings by using a complete stem analysis approach. Different oak species, ranging in age from 8 to 20 years, were felled and tree discs were removed from the base and at regular intervals along the tree stem. Each tree disc was dried, sanded and then their growth rings measured. Overall, mean height increment varied over time, but generally increased with age. There was substantial height increment variation among individual trees, with increment ranging from less than 0.5m/year, to 2m/year or more in some years. Height growth trajectories differed widely among sample trees, likely reflecting differences in site quality, genetic differences, seedling stock quality, and/or vegetation competition. The information about height, diameter and volume growth over the years acquired by a complete stem analysis can be used in the development of site index curves, employed to classify and compare the growth rates of different sites. This information will increase our knowledge about oak growth and yield in afforestation stands, and improve bottomland hardwood restoration efforts in the LMAV.

INTRODUCTION

The Lower-Mississippi Alluvial Valley (LMAV) once had the largest expanse of bottomland forest cover in the United States but extensive conversion to agriculture has diminished forest cover to less than 25% of its original extent. Forest restoration in the LMAV has been implemented over the past several decades, supported in large part by federal incentive programs aimed at restoring forest cover for the enhancement of wildlife habitat, soil conservation, and water quality. Oaks (*Quercus* spp.) are the most widely planted species, due to their wildlife and timber value. Many afforestation stands are reaching 20 years of age and yet there is limited information about stand development and growth performance of these species following afforestation.

MATERIALS AND METHODS

This study is evaluating planted hardwood stands throughout the Lower Mississippi Alluvial Valley. Sites include private lands registered in federal incentive programs such as the Conservation or Wetland Reserve Programs (CRP, WRP), state Wildlife Management Areas (WMAs), and federal lands such as US Army Corps of Engineers Mitigation lands.

The trees were felled and sectioned in discs taken at ground level, 0.3m, 1.3m, and then meter by meter until the top of the tree. Tree discs were dried in the oven prior to preparing. The discs were sanded to allow identification and measurement of individual growth rings. Tree discs were sanded in several stages, first using a belt sander (80 grit), and then sanding with an orbital sander (120 grit and 220 grit). The discs were analyzed in two orthogonal positions for the growth rings count, with the first measuring line drawn 22.5° to the largest diameter. Rings were identified and marked with the aid of a machinist ruler and dissecting microscope. Each growth ring represents one year of growth. The radius of each ring was measured in each direction of the orthogonal lines based on its distance from the pith. Average ring width was calculated as the average of the four measurements.

RESULTS AND DISCUSSION

The information obtained from ring measurements can be used to create a stem profile graph for all ages of the tree, each line corresponding to one growth year of the tree. The total height for the respective ages was

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estimated using Carmean's regression approach (Dyer and Bailey 1987). In order to evaluate growth increment over time, the rings series were normalized by age. Overall, mean height increment varied over time, but generally increased with age. There was substantial height increment variation among individual trees, with increment ranging from less than 0.5m/year, to 2m/year or more in some years. Height growth trajectories differed widely among sample trees, likely reflecting differences in site quality, genetic differences, seedling stock quality, and/or vegetation competition (fig. 1).

The information about height, diameter and volume growth over the years acquired by a complete stem analysis can be used in the development of site index curves, employed to classify and compare the growth rates of different sites. We expect that this study will increase our knowledge of oak growth and yield for afforestation stands in the LMAV. This information will be important to developing management approaches for afforestation stands, particularly as they mature.

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LITERATURE CITED

Dyer, M.; Bailey, R. L. 1987. A test of six methods for estimating true heights from stem analysis data. Forest Science. 33: 3-13.



Figure 1—Height growth trajectories over time for sampled trees.

ROTATION-LENGTH EFFECTS OF DIVERSE LEVELS OF COMPETITION CONTROL AND PRE-COMMERCIAL THINNING ON STAND DEVELOPMENT AND FINANCIAL PERFORMANCE OF LOBLOLLY PINE IN CENTRAL LOUISIANA

Michael A. Blazier, A. Gordon Holley, Shaun M. Tanger, Terry R. Clason, and Eric L. Taylor[1]

Long-term productivity of loblolly pine (*Pinus taeda L.*) plantations can be increased by early suppression of herbaceous and woody competing vegetation (Zutter and others 1986, Haywood 1994, Miller and others 2003a). The USDA Forest Service's Competition Omission Monitoring Project (COMP) was designed to isolate influences of two major competition groups (herbaceous and woody plants) at a range of sites in the Southeast U.S. A broad array of results from the COMP study improved understanding of loblolly pine growth (Miller and others 1991, 1995a, 1995b, 2003a), soil and foliage nutrients (Miller and others 2006), and understory and mid-story vegetation development (Miller and others 2003b). The potential for intensive early competition control to induce earlier onset of intraspecific competition and the differences between loblolly pine growth and development in response to more operational, one-time applications of herbicides relative to the sustained herbaceous and woody suppression of the COMP study treatments have received less attention. This study was designed to complement the COMP study by adding treatments that consisted of several operational mixtures of herbicides and pre-commercial thinning to the COMP study treatments.

The study was conducted at the USDA Forest Service Kisatchie National Forest in central Louisiana, near the town of Winnfield. The site's soil was classified as USDA Natural Resource Conservation Service Bellwood series, which is a poorly-drained loam. The site was planted with loblolly pine in December 1984 at 640 trees per acre following a mechanical site preparation in May 1984 and site preparation burn in October 1984. In July 2002, the entire site was commercially thinned to 160 trees per acre.

The following herbicide treatments were conducted at the site: (1) CONTROL, (2) HERB-COMP, (3) WOOD-COMP, (4) TOTAL-COMP, (5) LOTOT-OP, (6) MEDTOT-OP, and (7) HITOT-OP. The CONTROL treatment received no herbicide. Three treatments were COMP study treatments conducted annually from 1985 through 1989 as described by Miller and others (2003b). The HERB-COMP treatment was a suppression of herbaceous vegetation, the WOOD-COMP treatment was a suppression of woody vegetation, and the TOTAL-COMP treatment consisted of control of herbaceous and woody vegetation.

Three herbicide treatments (LOTOT-OP, MEDTOT-OP, HITOT-OP) were relatively low-, medium-, and high-cost herbicide mixtures conducted to approximate an operational, one-time application of herbicides that would provide suppression of herbaceous and woody vegetation. Woody vegetation control herbicides for the operational treatments were applied prior to planting in 1984. For the LOTOT-OP treatment, a mixture of picloram and triclopyr at 1.5 and 0.5 gallons per acre, respectively, was used for woody vegetation control. Woody vegetation control of the MEDTOT-OP treatment was conducted with a mixture of a blended formula of 2,4 D and dicamba and triclopyr at 1.0 and 0.13 gallons per acre, respectively. The HITOT-OP treatment consisted of hexazinone applied at 3.0 gallons per acre for woody vegetation control. Herbaceous vegetation control herbicides for the operational treatments were applied in 1985; all operational treatments had a mixture of sulfometuron methyl and sethoxydim at 4.0 oz. per acre and 1.5 pt. per acre, respectively, for herbaceous vegetation control. Four replications of all treatments were conducted; experimental units were 0.25-acre plots.

In 1991, a pre-commercial thinning treatment was superimposed as a split-plot treatment across all treatments. Two levels of a pre-commercial split-plot treatment were conducted: no pre-commercial thinning,

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and pre-commercial thinning to 300 trees per acre. Pre-commercial thinning was conducted by chainsaw-felling trees.

In 1987, 1991, 1994, 1999, 2001, 2005, and 2013, stand basal area was measured for all plots to determine treatment effects on stand-level growth. To determine the financial performance of all treatments, net present value, land expectation value, and estimated annual income were determined for all treatment combinations at a 5 percent interest rate. Silvicultural costs used in the financial analysis were drawn from Southeast U.S. silviculture cost surveys (Dooley and Barlow 2013). Revenue estimates in the financial analysis were derived from values of pulpwood, chip-and-saw, and sawtimber from the 2002 thinning and for standing timber value in 2013 (Louisiana Department of Agriculture and Forestry 2014).

There were significant herbicide and pre-commercial thinning effects on stand basal area. There were differences among herbicide treatments in all years observed except 2005 (fig. 1A). The TOTAL-COMP treatment had the highest basal area and the CONTROL treatment had the lowest basal area of all treatments in 1991. The TOTAL-COMP and CONTROL treatments had among the highest and lowest basal areas, respectively, of all treatments in all years in which significant

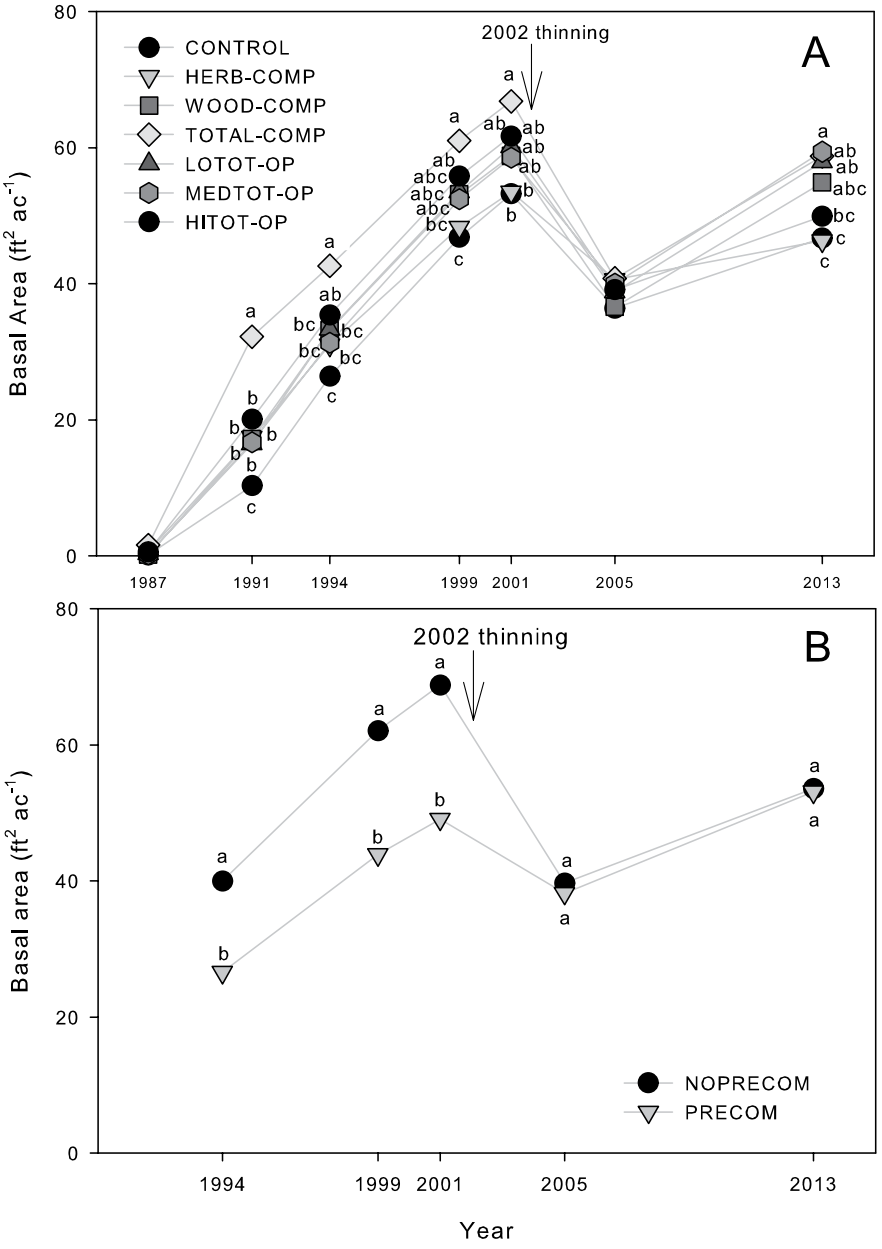


Figure 1—Stand basal area in response to (A) early-rotation herbicide treatments and (B) pre-commercial thinning treatments in a loblolly pine plantation in central Louisiana. For each year, treatment symbols noted with different letters differ at $P < 0.05$.

differences were observed. Pre-commercial thinning led to greater stand basal area in all years observed (fig. 1B). Among all treatment combinations, the HERB-COMP treatment with pre-commercial thinning had the highest net present value, land expectation value, and estimated annual income at the 5 percent interest rate tested (table 1). These findings suggest that intensive herbaceous vegetation control provided the greatest productivity benefit for the expense. Furthermore, although pre-commercial thinning increased silvicultural costs its long-term growth benefit enhanced financial performance. A management implication for these results is that early-rotation stand density is an important consideration when intensive vegetation control is conducted, with lower early-rotation stand density likely preferable in order to reduce lost productivity to intraspecific competition.

LITERATURE CITED

Dooley, E.; Barlow, R. 2013. Costs and cost trends of southern forestry practices in 2012. Alabama Cooperative Extension System Bulletin FOR-2001. Auburn, AL: Alabama Cooperative Extension System. [Not paged].

Haywood, J.D. 1994. Tenth-year results of herbaceous weed control in a loblolly pine plantation. Southern Journal of Applied Forestry. 18(3): 105-109.

Louisiana Department of Agriculture and Forestry. 2014. Louisiana pulpwood and timber production report. Available online at: www.ldaf.louisiana.gov/portal/Offices/Forestry/ForestryReports/tabid/449/Default.aspx [Date accessed: March 14, 2015].

Miller, J.H.; Zutter, B.R.; Zedaker, S.M. [and others]. 1991. A regional study on the influence of woody and herbaceous competition on early loblolly pine growth. Southern Journal of Applied Forestry. 15(4): 169-179.

Miller, J.H.; Zutter, B.R.; Zedaker, S.M. [and others]. 1995a. Early plant succession in loblolly pine plantations as affected by vegetation management. Southern Journal of Applied Forestry. 19(3): 109-126.

Miller, J.H.; Zutter, B.R.; Zedaker, S.M. [and others]. 1995b. A regional framework of early growth response for loblolly pine relative to herbaceous, woody, and complete competition control: the COMP project. Gen. Tech. Rep. SO-117. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Research Experiment Station. 48 p.

Miller, J.H.; Zutter, B.R.; Zedaker, S.M. [and others]. 2003a. Growth and yield relative to competition for loblolly pine plantations to midrotation: a southeastern United States regional study. Southern Journal of Applied Forestry. 27(4): 237-252.

Miller, J.H.; Zutter, B.R.; Newbold, R.A. [and others]. 2003b. Stand dynamics and plant associates of loblolly pine plantations to midrotation after early intensive vegetation management: a southeastern United States regional study. Southern Journal of Applied Forestry. 27(4): 1-16.

Miller, J.H.; Allen, H.L.; Zutter, B.R. [and others]. 2006. Soil and pine foliage nutrient responses 15 years after competing-vegetation control and their correlation with growth for 13 loblolly pine plantations in the southern United States. Canadian Journal of Forest Research. 36: 2412-2425.

Zutter, B.R.; Glover, G.R.; Gjerstad, D.H. 1986. Effects of herbaceous weed control using herbicides on a young loblolly pine plantation. Forest Science. 32(4): 882-899.

Table 1—Net present value (NPV), land expectation value (LEV), and estimated annual income (EAI) at a 5% interest rate for a 29-year rotation in response to herbicide and pre-commercial thinning treatments at a site in central Louisiana

Treatment	NPV	LEV	EAI
No Pre-commercial thinning			
Control	3877.3	1167.2	58.4
Herb-comp	3640.0	1095.7	54.8
Wood-comp	2302.5	693.1	34.7
Total-comp	3967.6	1194.4	59.7
Otot-op	4840.3	1457.1	72.9
Medtot-op	5092.6	1533.0	76.7
Hitot-op	5048.9	1519.9	76.0
Pre-commercial thinning			
Control	4795.7	1443.7	72.2
Herb-comp	5440.1	1637.6	81.9
Wood-comp	-667.9	-201.0	-10.1
Total-comp	-60.0	-18.1	-0.9
Lotot-op	4104.4	1234.6	47.2
Medtot-op	3133.5	943.3	67.1
Hitot-op	3310.6	996.6	49.8

USING MULT-SPECTRAL LANDSAT IMAGERY TO EXAMINE FOREST HEALTH TRENDS AT FORT BENNING, GEORGIA

Shawna L. Reid, Joan L. Walker, and Abigail Schaaf[1]

Assessing vegetation health attributes like canopy density or live crown ratio and ecological processes such as growth or succession ultimately requires direct measures of plant communities. However, on-the-ground sampling is labor and time intensive, effectively limiting the amount of forest that can be evaluated. Radiometric data collected with a variety of sensors from satellite platforms provide a partial solution to this challenge. Because plant function via photosynthesis is directly tied to electromagnetic energy, vegetation function has been successfully related to radiometric data (Lawley and others In press). Various indices have been developed to interpret vegetative functions or conditions including basal area, species composition, moisture stress, and damage from insects or disease (Liew and others 2008; Bannari and others 1995). The normalized difference vegetation index (NDVI), based on reflectance in the red (R) and near infrared (NIR) bands of the electromagnetic spectrum (NDVI = (NIR - R) / (NIR + R); range:-1 to 1), has been shown to be highly correlated with photosynthetic capacity, net primary productivity, leaf area index, and evapotranspiration. Further, time-series of NDVI have proven useful for evaluating such functions as canopy growth rates, and phenological events like the onset of spring (Pettorelli 2013).

Landsat TM data is one of the most useful types of radiometric data for interpreting vegetation. It has a moderate spatial resolution (30 m), high temporal resolution (images acquired every 16 days), and a long-term data archive from 1982 through present. The NDVI is readily calculated from Landsat TM data, adding to the desirability of these data for ecological applications. Finally, Landsat data are available at no charge, in part, ensuring their wide use in research and increasing the comparability among studies.

We conducted this study at Fort Benning, Georgia, where the upland landscape is managed to create an open pine forest that both meets military training needs and supports management of sensitive wildlife species. Management includes frequent prescribed burning to promote herbaceous groundcover and to control hardwood mid-story development, and thinning to achieve and maintain desired pine basal area. As early as 2005 Fort Benning forest managers voiced concerns about unexpected mortality in older loblolly pines. It was unclear if this phenomenon was novel, or if it was typical of historic patterns. To answer that question, in part, we proposed to investigate historic patterns of forest productivity using Landsat TM data. Here, we describe our approach to evaluating trends in pine forest productivity on the Fort Benning landscape. This project was conducted in partnership with the U.S. Forest Service Remote Sensing Applications Center (RSAC).

We used NDVI calculated from Landsat TM data as a general indicator of forest productivity, assuming that "greener" (higher NDVI, more productive) canopies are healthier than low productivity forests on similar sites. We used the USGS Earth Explorer website archive for identifying and downloading suitable scenes. We selected Landsat 5 scenes except in 2012 and 2013 when Landsat 7 scenes were the only ones available. With few exceptions we were able to identify scenes taken during leaf-off (December-March) and with minimal cloud cover. We downloaded Level 1 Product Generation System L1T Standard Terrain Correction ortho-corrected scenes. These were processed to Top-of-Atmosphere (Chandler and others 2009) and surface reflectance (Chavez 1988, Zhu and Woodcock 2012) using scripts developed by RSAC personnel and run within the Python and ERDAS Imagine software. The conversion from raw digital numbers to surface reflectance was done to minimize atmospheric effects (Song and others 2001). Two scenes were needed to cover the study area; they were joined within ERDAS. To calculate NDVI we used an RSAC-developed script that rescaled the index to integers between 0 and 200. Finally, we used the ARCINFO raster tool to extract values from each NDVI image for each of the 88 sample plots in which we had assessed pine canopy health following U.S. Forest Service Forest Health Monitoring protocols. Using these plots for which we have direct

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canopy health measures allowed us to relate spectral data to actual forest health conditions (results not reported here).

We examined the general trend in NDVI from 1984 to 2013 (fig. 1). Based on simple linear regression, the mean performance of all plots through time was negative (fig. 1).Though the trend in NDVI was negative (slope = -0.4241) the relationship was weak (r^2= 0.1052).

To better understand the variation among plots, we attempted to account for two components of the temporal trend: the NDVI level and slope. We first standardized NDVI values across all plots within each year, thereby assigning a standardized (z-transformed) NDVI score for each plot in each year. We calculated an average standardized NDVI score for each plot across all years. Then we calculated the slope of NDVI through time for each plot, and calculated a standardized slope score. Using both the mean standardized NDVI and standardized slope, we divided the plots into sixteen classes (table 1) defined by all combinations of four NDVI levels and four slope categories (strongly positive, flat, decreasing, and strongly decreasing). Levels were determined by natural breaks in the distributions of plots across the standardized NDVI and slope values, respectively.

Most sample plots were declining in greenness through time, consistent with the overall NDVI trend (fig. 1). The classification identified some plots that were both low NDVI but strongly increasing and high NDVI but strongly declining, combinations that were not intuitively easy to interpret. Based on an examination of a series of aerial photos, we determined that the former included relatively young fast-growing plantations and the latter occurred where management activities (e.g. fire, harvest) had reduced canopy cover abruptly.

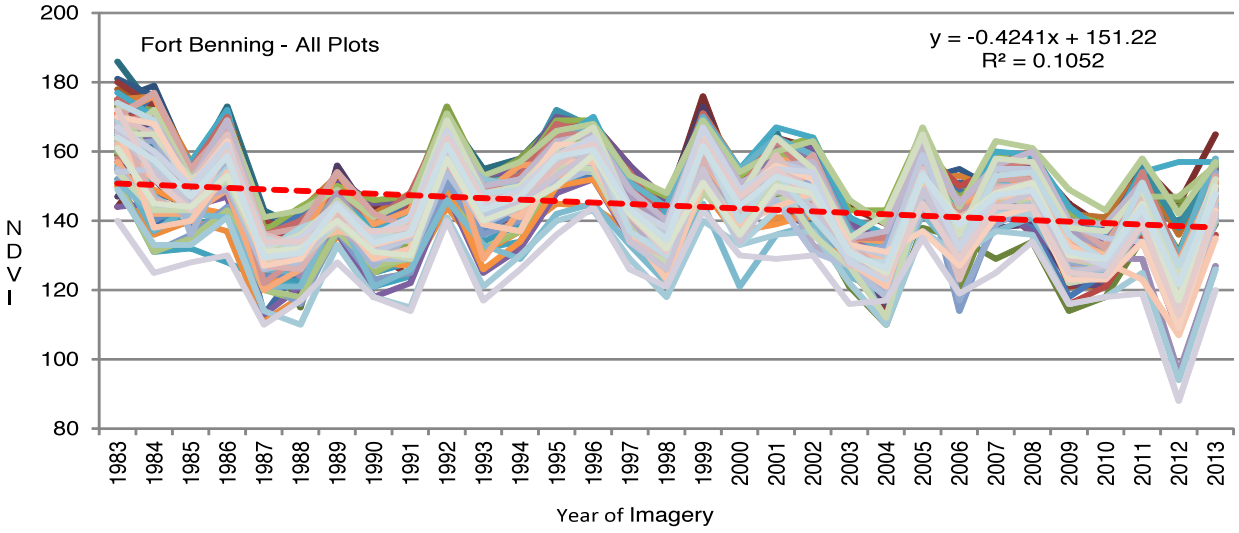


Figure 1—Trend analysis graph showing the NDVI values for each plot over the 30 year time series and the trend line (in red) which shows the slope of mean plot performance against year (linear regression).

Table 1—Distribution of 88 sample plots across trend classes defined by combining the NDVI standardized (z-transformed) within each year and averaged across all years with the magnitude of the slope of NDVI regressed on year

Class	Strong decline	Declining	Flat	Improving
Very healthy	5	12	0	1
Healthy	13	3	4	1
Unhealthy	5	13	1	1
Very unhealthy	5	15	6	3

Our results indicate the complexity of interpreting temporal trends in productivity on a managed landscape. Though overall trends appeared negative, management actions, especially those that cause abrupt changes in canopy cover, may influence trends. The use of Landsat based NDVI trends to interpret changes in forest health are complicated by image resolution that may not show isolated small patches of dying trees. At the very least, interpretations must be corroborated by other data, for example, information about stand management or natural disturbance.

LITERATURE CITED

Bannari, A.; Morin, D.; Bonn, F.; Huete, A.R. 1995. A review of vegetation indices. Remote Sensing Reviews. 131: 95-120.

Chandler, G.; Markham, B.L.; Helder, D.L. 2009. Summary of current radiometric calibration coefficients for Landsat MSS, TM, ETM+, and EO-1 ALI sensors. Remote Sensing of Environment. 113(5): 893-903.

Chavez, P.S. 1988. An improved dark-object subtraction technique for atmospheric scattering correction of multispectral data. Remote Sensing of Environment. 24(3): 459-479.

Lawley, V.; Lewis, M.; Clarke, K.; Ostendorf, B. 2016. Site-based and remote sensing methods for monitoring indicators of vegetation condition: An Australian review. Ecological Indicators. 60: 1273-1283.

Liew, O.W.; Chong, P.C.J.; Li, B.; Asundi, A.K. 2008. Signature optical cues: emerging technologies for monitoring plant health. Sensors (Basel). 8(5): 3205-3239. doi:10.3390/s8053205.

Pettorelli, N. 2013. The Normalized Difference Vegetation Index. Oxford, United Kingdom: Oxford University Press. 44 p.

Song, C.; Woodcock, C.E.; Seto, K.C. [and others]. 2001. Classification and change detection using Landsat TM data: when and how to correct atmospheric effects. Remote Sensing of Environment. 75(2): 230-244.

Zhu, Z.; Woodcock, C.E. 2012. Object-based cloud and cloud shadow detection in Landsat imagery. Remote Sensing of Environment. 118: 83-94. doi:10.1016/j.rse.2011.10.028.

CHANGES IN BARK COMPOSITION FROM LONG-TERM ELEVATED CO_2 TREATMENT: IMPLICATIONS FOR THE MANAGEMENT OF SWEETGUM AS A WOOD ENERGY CROP

Thomas L. Eberhardt, Chi-Leung So, Daniel J. Leduc, Nicole Labbé and Jeffrey M. Warren[1]

Tree bark is comprised of living inner bark (phloem) that transports the products of photosynthesis and dead outer bark that protects the living tissues and seals in moisture. Active and passive defenses against destructive agents (e.g., insects, fungal pathogens) are provided by the inner bark and outer bark, respectively (Eberhardt 2013). For sweetgum (*Liquidambar styraciflua* L.), the bark comprises 23 percent by mass of the sapling-sized stems and 31 percent of the branches (Koch 1985); the percentage of bark in larger trees would be less. Trees debarked during processing by the forest products industry sector generate significant amounts of bark residue for conversion to mulch products or for fuel value recovery. The chemical composition of tree bark is important as it has the potential to significantly impact the utilization of bark, particularly for biofuel applications.

Sweetgum is a medium to large tree that grows best on moist sites, but can tolerate low moisture and nutrient availabilities. Adaptability to a wide range of site conditions makes it the preferred hardwood for bioenergy production despite its slower growth rate than other hardwoods such as cottonwood and sycamore (Kline and Coleman 2010). The effects of long-term elevated-CO_2 treatment on the inner and outer bark chemistry of sweetgum trees were investigated by the authors in a recent study (Eberhardt and others 2015). In a continuation of this work, ash, carbon, and hydrogen analyses are presented in the context of biofuels, and nitrogen content is presented since the demand for this macronutrient is relevant to the management of sweetgum as a wood energy crop. Furthermore, the application of principal component analysis (PCA) has been included as a technique to detect differences in spectroscopic data collected from the bark tissues both before and after extraction.

Inner and outer bark was collected from sweetgum trees harvested at the Oak Ridge National Laboratory sweetgum Free-Air CO_2 Enrichment (FACE) study site in Roane County, TN. Elevated-CO_2 treatment (*ca.* 550 ppm) was applied during the daytime, April through October, for more than a decade. Ash content was determined by combustion (525 °C, 6h). Carbon, hydrogen, and nitrogen concentrations were determined using a CHNS/O analyzer. One-way ANOVA of plot means used PROC GLM in SAS/STAT®9.3 ($P_\alpha < 0.1$ being significant). Multivariate analysis was performed on Fourier transform infrared (FTIR) spectroscopic data using Unscrambler 8.0 software.

Similar to that observed in sweetgum wood from a corresponding FACE study (Kim and others 2015), a higher ash content was observed for sweetgum inner bark with the elevated-CO_2 treatment (table 1); no difference was observed for the outer bark. Increases in ash content are disconcerting due to the negative impact on the utilization of woody biomass. The presence of ash can lead to excessive tool wear during lumber production (Porankiewicz and others 2006), and the formation of deposits during pulp and paper processing operations (Biermann 1996). As for biofuel applications, ash sintering during combustion causes problems in boilers (Lestander and others 2012), poisons catalysts used for biofuel conversions/syntheses (López-González and others 2014), and leads to the accumulation of heavy metals in biofuel byproducts thereby causing disposal issues (Liu and others 2015). To date, little is known about the effects of elevated CO_2 on biomass energy content in forest plantations (Calfapietra and others 2009). We found that the elevated-CO_2 treatment was associated with lower contents of carbon and hydrogen in the inner bark (table 1). Compared to fossil fuels, the higher heating value for biomass is lower due to relatively high hydrogen/carbon and oxygen/carbon ratios

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Table 1—Ash, carbon, hydrogen, and nitrogen analyses of sweetgum inner and outer bark samples from a FACE study site in Tennessee

		Ash	Carbon	Hydrogen	Nitrogen
		---------- % ----------			
Inner bark	Ambient CO_2	8.1 ± 0.1	40.1 ± 0.3	5.1 ± 0.2	0.32 ± 0.01
	Elevated CO_2	10.8 ± 1.0	35.9 ± 1.5	4.4 ± 0.3	0.24 ± 0.04
	P_α value	0.062	0.064	0.092	0.149
Outer bark	Ambient CO_2	7.1 ± 0.2	46.2 ± 0.2	5.2 ± 0.2	0.42 ± 0.00
	Elevated CO_2	7.9 ± 0.8	46.1 ± 0.5	5.2 ± 0.1	0.35 ± 0.01
	P_α value	0.336	0.856	0.542	0.016

(Basu 2010). Overall, these results suggest that the elevated-CO_2 treatment lowered the quality of bark for thermochemical conversions (combustion, pyrolysis, gasification, liquefaction) by an increased mineral content coinciding with decreased carbon levels.

While nitrogen concentrations were not different for the inner bark from the two CO_2 treatments, they were lower for the outer bark receiving the elevated-CO_2 treatment (table 1). Sweetgum with elevated CO_2 has shown increased productivity (Norby and others 2002) and increased root biomass/distribution (Iversen 2010) for nutrient extraction from the soil (Garten and others 2011). Decreased foliar nitrogen content (Norby and Iverson 2006), and also decreased nitrogen content of the outer bark are indicative of the constraints placed on nitrogen availability. Increased management intensity (e.g., weed control, weed control with irrigation, etc.) affords increased productivity in sweetgum stands and increased soil carbon and nitrogen by accelerated litter and root formation (Johnsen and others 2013). With intensively-managed hardwoods there can be timing issues whereby the time that the nitrogen is available in the soil is not the same as when it is needed by the tree (Scott and others 2004). Balancing nitrogen supply and demand for short-rotation sweetgum plantations will likely warrant greater attention to offset any exacerbation higher nitrogen demand from higher atmospheric CO_2 concentrations (Grant 2013).

Biochemical routes (e.g., fermentation) to generate biofuels from biomass are dependent upon the relative proportion of basic chemical constituents, those being cellulose, lignin, hemicelluloses, along with non-structural biomolecules that can be isolated by extraction with organic solvents, the so-called extractives. A discussion of these constituents is beyond the scope of this report; however, PCA of FTIR spectroscopic data from the inner and outer bark before and after extraction are provided to demonstrate the potential magnitude of chemical changes from the elevated-CO_2 treatment. The PCA scores plot (fig. 1) shows that the technique was able to differentiate between both sample processing (unextracted vs. extracted) and bark type (inner vs. outer bark); however it was unable to differentiate between elevated- and ambient-CO_2 treatments. Accordingly, while there may be chemical changes in the inner or outer bark tissues in response to the elevated-CO_2 treatment, it is unlikely that those changes would be significant enough to impact biochemical routes to generate biofuels.

This work would not have been possible without the efforts of Karen Reed (USDA Forest Service, Southern Research Station) and Keonhee Kim (Center for Renewable Carbon, University of Tennessee).

LITERATURE CITED

Basu, P. 2010. Biomass gasification and pyrolysis: practical design and theory. Burlington, MA: Academic Press. 365 p.

Biermann, C.J. 1996. Handbook of pulping and papermaking, 2nd ed. San Diego: Academic Press. 754 p.

Calfapietra, C.; Ainsworth, E.A.; Beier, C. [and others]. 2009. Challenges in elevated CO_2 experiments on forests. Trends in Plant Science. 15(1): 5-10.

Eberhardt, T.L. 2013. Longleaf pine inner bark and outer bark thicknesses: measurement and relevance. Southern Journal of Applied Forestry. 37(3): 177-180.

Eberhardt, T.L.; Labbé, N.; So, C.-L. [and others]. 2015. Effects of long-term elevated CO_2 treatment on the inner and outer bark chemistry of sweetgum (*Liquidambar styraciflua* L.) trees. Trees-Structure and Function. 29: 1735-1747.

Garten, C.T.; Iversen, C.M.; Norby, R.J. 2011. Litterfall ^{15}N abundance indicates declining soil nitrogen availability in a free air CO_2-enrichment experiment. Ecology. 92: 133-139.

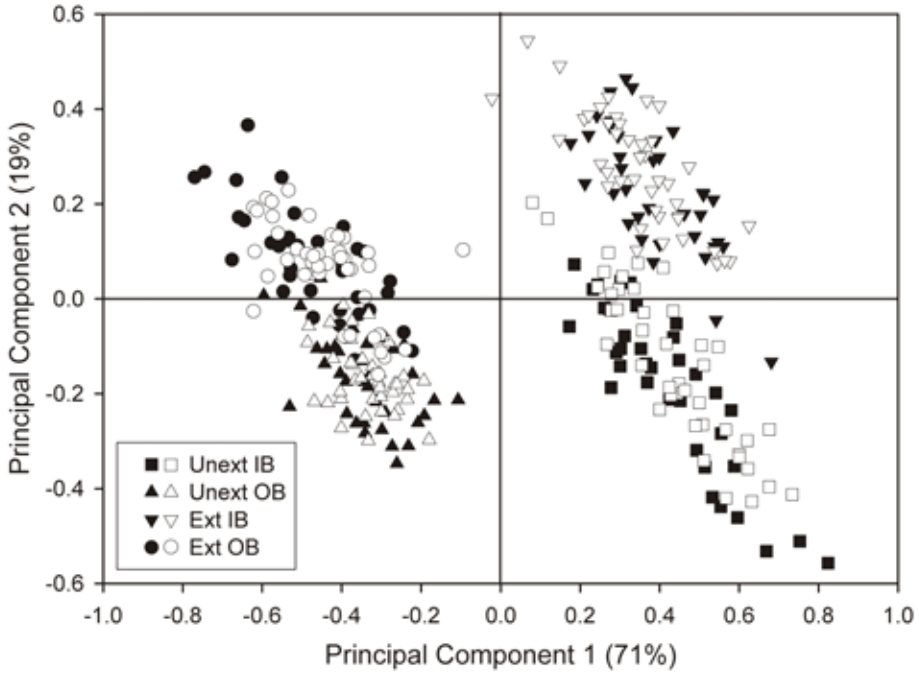


Figure 1—PCA scores plot of FTIR spectra from both unextracted (Unext) and extracted (Ext) inner (IB) and outer bark (OB) undergoing elevated- (filled shapes) and ambient-CO_2 (open shapes) treatments.

Grant, R.F. 2013. Modelling changes in nitrogen cycling to sustain increases in forest productivity under elevated atmospheric CO_2 and contrasting site conditions. Biogeosciences. 10: 7703-7721.

Iversen, C.M. 2010. Digging deeper: fine-root responses to rising atmospheric CO_2 concentration in forested ecosystems. New Phytologist. 186: 346-357.

Johnsen, K.H.; Samuelson, L.J.; Sanchez, F.G.; Eaton, R.J. 2013. Soil carbon and nitrogen content and stabilization in mid-rotation, intensively managed sweetgum and loblolly pine stands. Forest Ecology and Management. 302: 144-153.

Kim, K.; Labbé, N.; Warren, J.M. [and others]. 2015. Chemical and anatomical changes in *Liquidambar styraciflua* L. xylem after long term exposure to elevated CO_2. Environmental Pollution. 198: 179-185.

Kline, K.L.; Coleman, M.D. 2010. Woody energy crops in the southeastern United States: two centuries of practitioner experience. Biomass and Bioenergy. 34: 1655-1666.

Koch, P. 1985. Utilization of Hardwoods Growing on Southern Pine Sites. Vol. 3. Agric. Handb. 605. Washington, DC, U.S. Department of Agriculture: 2543-3710.

Lestander, T.A.; Lundström, A.; Finell, M. 2012. Assessment of biomass functions for calculating bark proportions and ash contents of refined biomass fuels derived from major boreal tree species. Canadian Journal of Forest Research. 42: 59-66.

Liu, N.; Nielsen, H.K.; Jorgensen, U.; Laerke, P.E. 2015. Sampling procedure in a willow plantation for chemical elements important for biomass combustion quality. Fuel. 142: 283-288.

López-González, D.; Fernandez-Lopez, M.; Valverde, J.L.; Sanchez-Silva, L. 2014. Gasification of lignocellulosic biomass char obtained from pyrolysis: kinetic and evolved gas analyses. Energy. 71: 456-467.

Norby, R.J.; Hanson, P.J.; O'Neill, E.G. [and others]. 2002. Net primary productivity of a CO_2-enriched deciduous forest and the implications for carbon storage. Ecological Applications. 12(5): 1261-1266.

Norby, R.J.; Iversen, C.M. 2006. Nitrogen uptake, distribution, turnover, and efficiency of use in a CO_2-enriched sweetgum forest. Ecology. 87(1): 5-14.

Porankiewicz, B.; Iskra, P.; Sandak, J. [and others]. 2006. High-speed steel tool wear during wood cutting in the presence of high-temperature corrosion and mineral contamination. Wood Science and Technology. 40: 673-682.

Scott, D.A.; Burger, J.A.; Kaczmarek, D.J.; Kane, M.B. 2004. Nitrogen supply and demand in short-rotation sweetgum plantations. Forest Ecology and Management. 189: 331-343.

AN ASSESSMENT OF COARSE WOODY DEBRIS DYNAMICS IN AN URBAN FOREST

Michael K. Crosby, Helen Petre, Justin Sims and Rachel Butler [1]

Abstract—Determining the amount of coarse woody debris (CWD) in an urban forest is essential to developing management strategies to maintain ecosystem function while minimizing hazards to local residents. It is also an essential variable used for the assessment and monitoring of carbon dynamics and fire fuel loads in forests. Plots were established and CWD measured in Marshall Forest, an approximately 300 acre urban forest in Rome, GA, managed by the Nature Conservancy, to determine the amount and the spatial distribution of CWD. The volume of CWD is 863 ft^3/ac and has a carbon content of 9.1 tons/ac and varies in its distribution throughout the forest. This area contains a large amount of debris that, while serving local wildlife, could pose a hazard for wildfire that could spread to adjacent neighborhoods. Forest managers should be aware of CWD levels and take steps to mitigate hazards.

INTRODUCTION

Marshall Forest Preserve (MFP) is 120 ha of forested natural area in northwest Georgia dominated by old growth pine and oak. Old growth forest remnants are rare in the eastern United States and are stands with trees that are old for their species in forests with a high degree of complexity (Petruccelli and others 2014). MFP is in the city of Rome (34° 15' N, 85° 12' W) and is one of the only remaining old growth areas in city limits in eastern North America (DeSelm 1984). Commercial logging has not been reported in MFP (DeSelm 1984) however, we frequently encountered evidence of timber harvest, especially on the west side near Mount Alto. MFP was designated a National Natural Landmark by the United States government in 1966 and is currently managed by the Nature Conservancy.

MFP is dominated by shortleaf pine *(Pinus echinata)* and chestnut oak (*Quercus montana)*. In the understory, red maple (*Acer rubra)* and mockernut hickory *(Carya tomentosa)* are common, with occasional dogwood (*Cornus florida),* loblolly pine *(Pinus taeda),* and black cherry *(Prunus serotina*). There is also a small population of montane longleaf pine (*Pinus palustris*) within the forest, although none were encountered during this study.

There are no fires documented in MFP since 1920 (Petruccelli and others 2014), which is a potential cause for the lack of pine and oak regeneration, as shade tolerant species are currently common in the understory. Active suppression of fires is often implicated in the decline of pine and oak regeneration and the succession to red maple dominated understory (Abrams 1992, Nowacki and Abrams 2008). It is unlikely that fire can be used at MFP due to the highly urbanized surrounding areas.

STUDY SITE

MFP is in the southern portion of the Ridge and Valley physiographic province in the Appalachian Mountains, characterized by folded layers of sedimentary rock forming long parallel ridges of sandstone and limestone (DeSelm 1984). The study site is on Horseleg Mountain, a ridge underlain by Armuchee chert and Conasauga shale (DeSelm 1984). Soil is shallow, well-drained silty loam formed from shale (Web soil survey 2010) and elevations range from 200-300 m (656 – 984 feet). The study area has a humid continental climate with mild winters and hot summers. Mean monthly temperatures range from 5° C (41° F) in January to 25° C (77° F) in July. Mean annual precipitation is approximately 1100 mm (43 inches) evenly distributed throughout the year (PRISM 2015).

METHODS

We randomly generated 30 plots within the boundaries of MFP using ArcMap (fig. 1). We navigated to the plots using Global Positioning Systems (GPS) and established 24 foot (7.31 m) transects from each plot center at 30, 150, and 270° azimuths (fig. 2; Woodall and Williams 2005). We included CWD pieces in the sample if the centerline intersected any of the three 24 foot transects emanating from the center of each plot at azimuths of 30, 150, and 270° (fig 2). CWD includes dead and downed pieces or portions of pieces of wood that have

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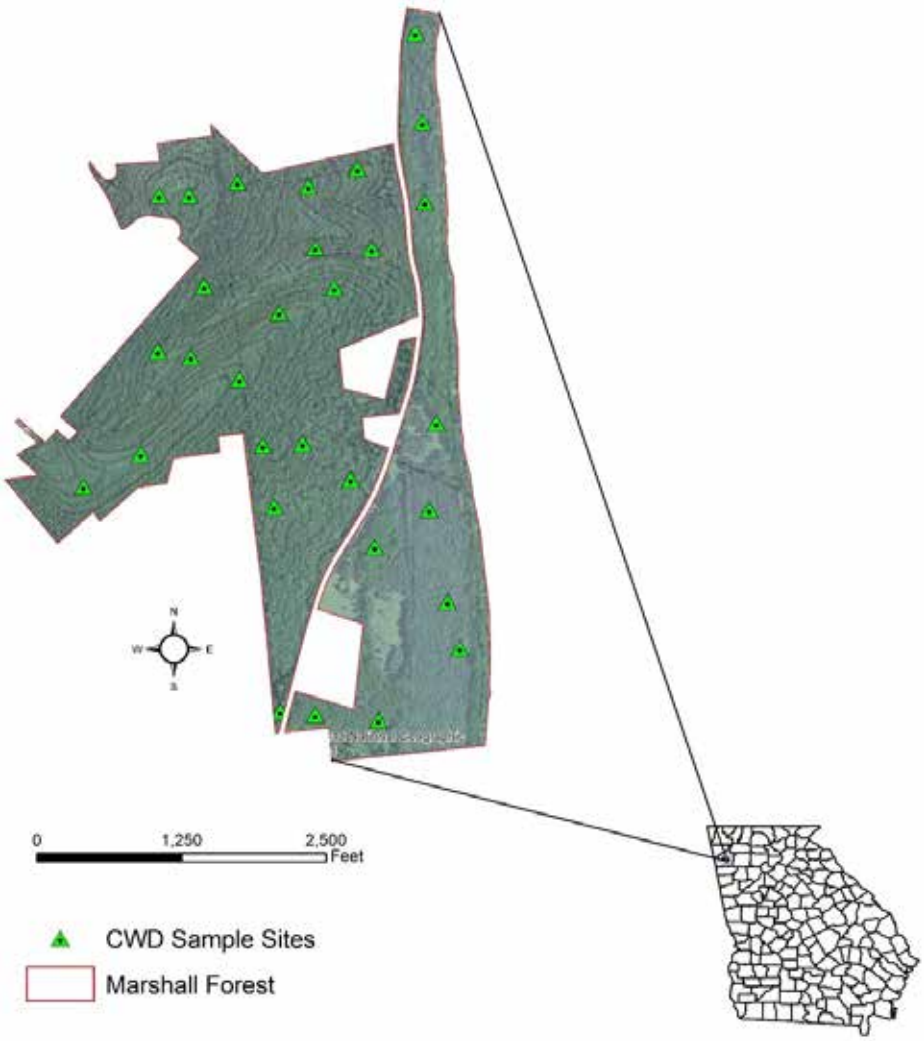


Figure 1—CWD sample locations in Marshall Forest Preserve, Rome, GA.

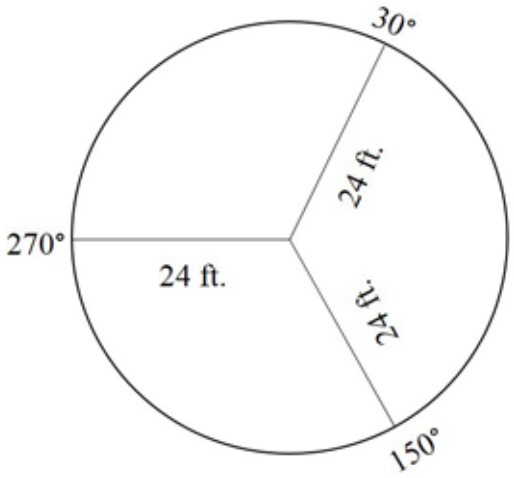


Figure 2—Example of a plot showing azimuths for each transect (after Woodall and Monleon 2008).

a diameter of at least 3 inches along a length of at least 3 feet and are in decay classes one to four (Woodall and Monleon 2008).

We rated the decay class according to a five-class scale (Sollins 1982) (table 1). For pieces in decay classes one through four we recorded the diameter at the point of transect intersection, large and small end diameter, species if possible (or pine or hardwood) decay class and length. We calculated volumes for each CWD piece using Smalian's formula (after Woodall and Monleon 2008). We calculated the estimated per acre (ft^3/ac) volume on each plot (Equation 1)

$$\overline{Y}_{CWD} = \frac{1}{k_c} \sum_{j=1}^{k_c} \overline{y}_j \ ,$$

where $\bar{y}_j$ = ft^3/ac volume on the jth transect and k_c= the number of transects per plot, and interpolated (via kriging) plot estimates for MFP.

RESULTS

MFP is dominated by shortleaf pine and chestnut oak with red maple abundant in the understory. Hardwood species comprised a majority of CWD sampled within MFP. There is very little CWD in decay class one, likely owing to time since the last major disturbance. Most CWD was in decay classes two and three for both pine and hardwood (fig. 3), although there was generally more pine in higher states of decay than hardwood.

The estimated CWD volume is 837 ft^3/ac, which represents a carbon content of 9.1 tons/ac. The distribution of CWD varies throughout the forest; however, the highest volumes are at the lower elevations along the north sections of MFP. This is the flood plain of the Coosa River, so it is reasonable that CWD washes up during flood events (fig. 4). Areas of increasing slope also have a high concentration of CWD as do areas along the MFP property boundary.

DISCUSSION

MFP is a disturbance-driven forest, with the most recent disturbances occurring in 1993 (ice/snow) and 2011 (wind) (Petruccelli and others 2014). The result of these disturbances is a high amount of CWD. Compared to Forest Inventory Analysis estimates, MFP has approximately four times the amount of CWD than the average for the State of Georgia (Woodall and Monleon 2008). Coupled with fire exclusion, disturbance has resulted in a significant amount of CWD in the MFP that could serve as a fuel source for wildfire. As the MFP is within the boundary of a city, prescribed fire is difficult to accomplish. Given that CWD falls within the 1,000+ hour fuel category, a wildfire would be catastrophic for the old-growth forest of MFP and negatively impact the city of Rome, GA. It is interesting that there was only one CWD piece in decay class one which points to the time elapsed since the last major disturbance. It would be informative to determine the decay rate of CWD by species within MFP. In terms of fire fuel modelling and monitoring, it would also be useful to obtain estimates for litter and duff within the forest. Future research within MFP will focus on these efforts with the goal of determining fire risk in MFP, which could be extended to other natural forests in the southeastern United States.

Table 1—Attributes of CWD pieces used to determine decay class (Woodall and Monleon 2008)

Decay class	Structural integrity	Texture of rotten portions	Color	Invading roots	Branches
1	Sound, freshly fallen	Intact	Original	Absent	Present, fine twigs are present
2	sound	Mostly intact	Original	Absent	Present, fine twigs are absent
3	Heartwood sound	Hard	Reddish brown	Sapwood	Branch stubs will not pull out
4	heartwood rotten	Soft, small blocky pieces	Reddish	Throughout	Branch stubs pull out
5	None	Soft; powdery when dry	Red brown	Throughout	Branch stubs and pitch pockets have usually rotted down

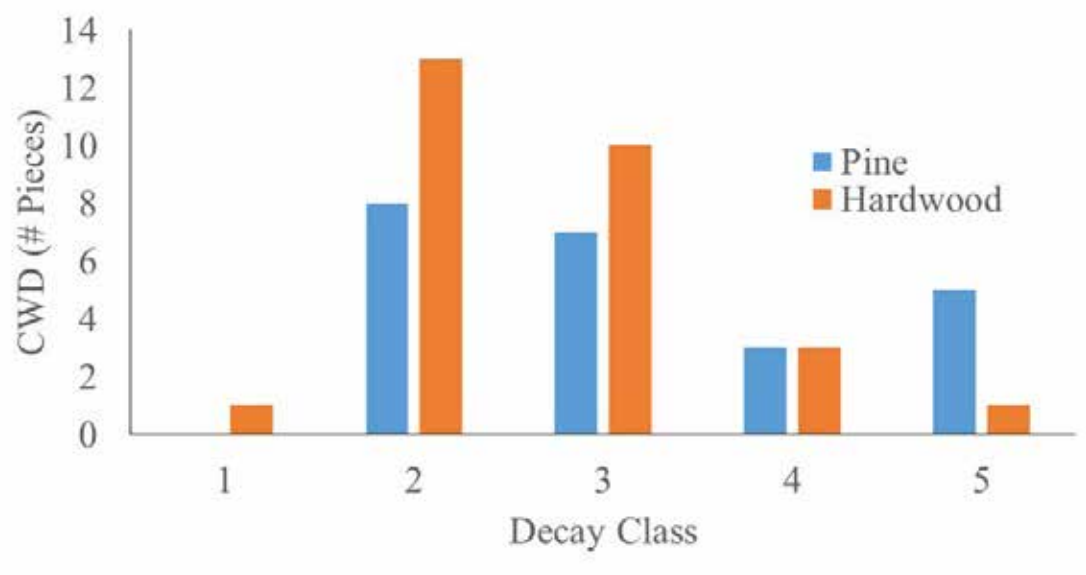


Figure 3—Distribution of CWD, for pine and hardwood, as a function of decay class.

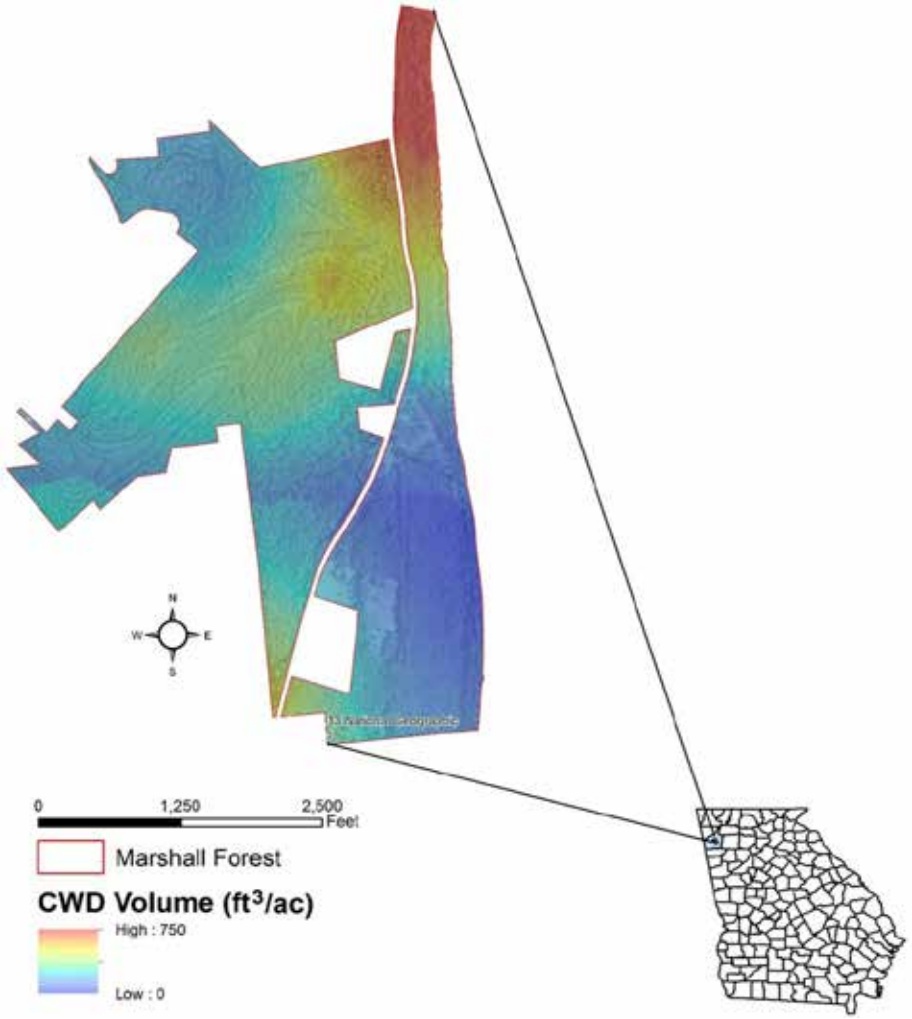


Figure 4—Spatial distribution of CWD (ft^3/ac) throughout Marshall Forest.

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We wish to thank Dr. Christopher Mullins and Mr. Clint Helms at Shorter University for their constructive reviews of this manuscript.

LITERATURE CITED

Abrams, M. 1992. Fire and the development of oak forest. Bioscience. 42(5): 346-353.

DeSelm, H. 1984. Potential national natural landmarks of the Appalachian ranges natural regions: Ecological report. Knoxville, TN: University of Tennessee: 277-279.

Nowacki, G.; Abrams, M. 2008. The demise of fire and "mesophication" of forests in the eastern United States. Bioscience. 58(2): 123-138.

Petruccelli, C.; Sakulich, J.; Harley, G.; Grissino-Mayer, H. 2014. Structure and dynamics of an old growth pine oak community of the southern Appalachians, Georgia, United States of America. Southeastern Geographer. 54 (2): 161-182.

PRISM Climate Group. 2015. Northwest Alliance for Computational Science & Engineering (NACSE). Corvallis, OR: Oregon State University. R. http://prism.oregonstate.edu/normals. [Date accessed: March 10, 2015].

Sollins, P. 1982. Input and decay of coarse woody debris in coniferous stands in western Oregon and Washington. Canadian Journal of Forest Research. 12(1): 18-28.

Web Soil Survey. 2010. Web soil surveys (Floyd County, Georgia). United States Department of Agriculture, Natural Resource Conservation Service. http://websoilsurvey.nrcs.usda.gov/app/. [Date accessed: March 10, 2015].

Woodall, C.; Monleon, V. 2008. Sampling protocol, estimation and analysis procedures for the down woody materials indicator of the forest inventory and analysis program. Gen. Tech. Rep NRS-22. Newton Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 68 p.

Woodall, C; Williams, M. 2005 Sampling protocol, estimation, and analysis procedures for the down woody materials indicator of the FIA program. Gen. Tech. Rep. NC-256. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Research Station. 47 p.

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Keywords: Disturbance, fertilization, forests, silviculture, regeneration.

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