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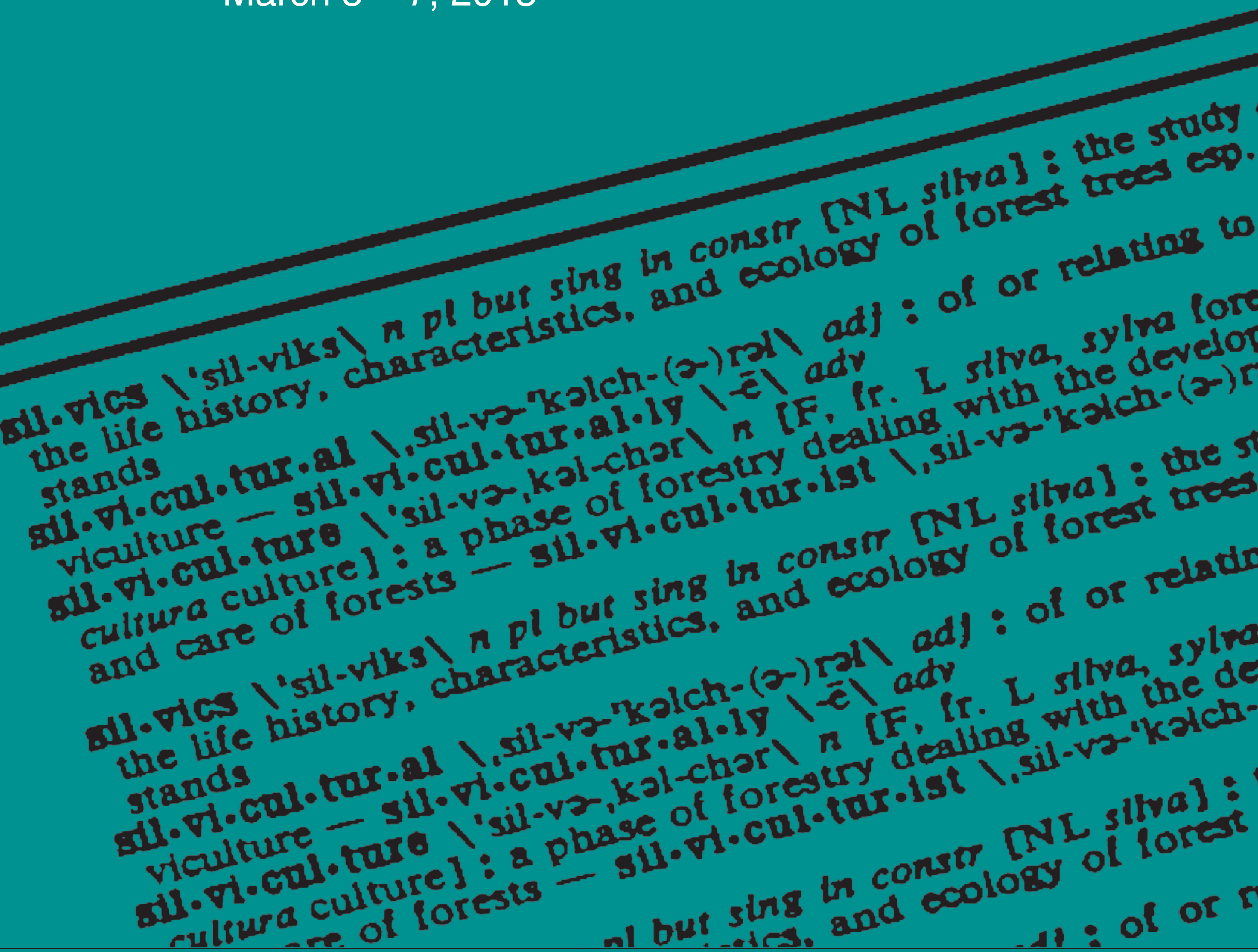
Proceedings of the 17th Biennial Southern Silvicultural Research Conference

Shreveport, Louisiana
March 5 – 7, 2013



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Proceedings of the 17th Biennial Southern Silvicultural Research Conference

Edited by

A. Gordon Holley, Kristina F. Connor, and James D. Haywood

Shreveport, Louisiana
March 5 – 7, 2013

Hosted by

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Preface

The 17th Biennial Southern Silvicultural Research Conference was held March 5-7, 2013 at Sam's Town Hotel & Casino, Shreveport, LA. This conference was the latest in a series of meetings designed to provide a forum for the exchange of research information among silviculturists, researchers, and managers. Presentations emphasized research in hardwood regeneration, best management practices for stream crossings, conservation, pine regeneration and genetics, fire effects, eco-physiology, nutrition, vegetation management, growth and development, methods, biometrics, biomass, and threats. Over 300 people registered for the conference. The conference included 104 oral and 50 poster presentations on March 5 and 6. Two field tours on March 7 covered Restoration and Conservation-Based Silviculture and Intensive Loblolly Pine Management. The online General Technical Report (e-GTR-SRS) was edited and compiled by the Chairs for publication by the Southern Research Station to document the proceedings. Many university, industry, and association partners worked with the USDA Forest Service to make this conference a success. The steering committee devoted numerous hours to reviewing abstracts, establishing the program for oral and poster presentations, and making all necessary arrangements for the conference. Steering committee members included:

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Partial funding for the many conference expenses was provided by the Southern Research Station, Louisiana Tech University, our sponsors, and exhibitors. The Southern Research Station is especially grateful to Louisiana Tech University for its diligence in handling the fiscal responsibilities of the meeting and the Alumni Association for helping with the registration process. We thank Erwin Chambliss for his editorial help with this proceedings.

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Hardwood Regeneration

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SURVIVAL AND INITIAL GROWTH ATTRIBUTES OF IMPROVED AND UNIMPROVED CHERRYBARK OAK IN SOUTH ARKANSAS

Joshua P. Adams, David Graves, Matthew H. Pelkki, Chris Stuhlinger, and Jon Barry[1]

Abstract--Thousands of acres are planted every year with genetically improved seedlings; but while pine continues to be extensively explored, the same is not true for hardwoods due to costs and rotation length. An improved cherrybark oak (*Quercus pagoda* Raf.) seed orchard exists in North Little Rock, AR, providing an opportunity to evaluate hardwood improvement. However, the cost and limited testing of these seedlings have been large limiting factors in their deployment. In February 2012, improved and woods-run seedlings were hand-planted at two sites in southern Arkansas including a site near Hope, AR, and one near Monticello, AR. The sites were treated with 2 ounces per acre of Oust XP® 2 weeks after tree planting with manual control of sumac (*Rhus* spp.) and sweetgum (*Liquidambar styraciflua* L.) shortly thereafter. A random sample of seedlings at the nursery confirmed that seedling undercutting effectively controlled root length which was statistically the same for both groups at 21.8 inches. However, root collar diameter of an improved seedling was on average 27 percent larger than an unimproved seedling. These trends were similar to those among planted seedlings in which improved seedlings were 9 percent and 8 percent greater in regards to ground line diameter and height, respectively. However, improved seedlings exhibited greater initial mortality, by 6.2 percent, in the first few months of their growing season. While initial mortality is often considered random, disparity between the two groups points to other causes, such as the larger root sizes, which may pose planting problems. In conclusion, these results indicate that genetic improvement has increased 1-0 seedling size, but survivability must be carefully monitored.

INTRODUCTION

Cherrybark oak (*Quercus pagoda* Raf.) is a red oak that is one of the most highly valued hardwood species in southern forests. Similar to other hardwood species, very few genetic improvement studies have been performed on it. Adams and others (2006) determined that trees grown from seeds collected in the central area of the cherrybark oak range, predominately on the loess hills along the Mississippi River, had superior volume production. Substantial variation among families within a geographical region was also observed by Greene and others (1990) and Adams and others (2006) leading to the recommendation that provenance and individual family considerations should be made in tandem during selection for future generations. This is similar to recommendations for water oak (*Q. nigra* L.) in a provenance study (Adams 1989) that overlapped geographically with the studies conducted on cherrybark oak.

Cherrybark oak does have high individual tree height heritability (i.e., the proportion of variability that can be explained by genetic variability) in these studies and is comparable with findings in Nuttall oak (*Quercus texana* Buckl.) which have a family heritability of 0.72 to 0.96 for height and 0.22 to 0.95 for diameter (Gwaze and others 2003). These genetic assessments of oaks indicate that genetic gain in volume production can be achieved through proper selections. While some commercial work has been undertaken, field tests of the resulting progeny are unique.

Successful hardwood stand establishment generally requires both high quality seedlings and costly silviculture treatments, including herbaceous weed control through herbicide. The high costs of hardwood seedlings and higher planting costs make initial growth and survival critical to justify the costs of stand establishment. Additional costs from using improved genetic seedlings makes deployment even more economically challenging. For instance, the Arkansas Forestry Commission (AFC) sells 2nd generation improved cherrybark oak seedlings for $400 per 1000 seedlings (http://forestry.arkansas.gov). While these seedlings are twice as expensive as the unimproved cherrybark oak, they are projected to achieve a 13 to 18 percent volume gain. At an 8- by 10-foot spacing, improved seedlings add an additional $109 per acre at establishment. At a 6 percent rate of return over a 50 year rotation, the added timber value must result in $2,007 per acre over the unimproved seedlings. Thus, added expenses must be carefully examined based solely on genetic volume gains from a progeny test. Regardless of the genetic nature of the seedlings, the cost of establishment is

[1]Assistant Professor, University of Arkansas, Arkansas Forest Resource Center, Monticello, AR 71656; Private Lands Biologist East Region, Arkansas Game and Fish Commission, Brinkley, AR, 72021; Professor and University System Forester, respectively, University of Arkansas, Arkansas Forest Resources Center, Monticello, AR 71656; and Extension Forester, Arkansas Forest Resources Center, Southwest Research and Extension Center, Hope, AR, 71801.

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high and competition control is needed to ensure adequate seedling survival. To ensure that survival and growth goals are met, further investments in herbaceous weed control are recommended (Dubois and others 2000, Ezell and Catchot 1998, Ezell and Hodges 2002, Ezell and Yeiser 2007, Schuler and others 2004). Competition control before planting and 1 year after planting have consistently shown increased survival to the degree that the added cost may be warranted. However, every application of herbicide (including labor) increases the cost by $30 to $50 an acre for a pre-emergent herbicide application and over $200 for direct spraying of herbicide after the first year of growth. If the added tree vigor, often quoted as an attribute of genetically improved seedlings, allows for adequate survival without extra herbicide treatments, then the costs savings would justify the added costs of buying the improved oak seedlings.

MATERIAL AND METHODS

Initial dimensions of seedlings were assessed directly from the nursery grounds. Seedlings (n = 400) were sampled from Arkansas Forestry Commission cool storage at the Baucum Nursery in North Little Rock, AR. Four random bags were pulled from storage, and 100 seedlings from each were measured for root length, defined as root collar to the end of the tap root, and root collar diameter. Data were analyzed with a general linear model to assess differences between the two seedling types.

A random selection of 840 seedlings, 420 improved and 420 unimproved, were planted at two locations in south Arkansas. The first location was an old-field site near Hope, AR at the University of Arkansas Southwest Research and Extension Center, and the second was a recently cleared pine forest on the University of Arkansas-Monticello School Teaching and Research Forest near Monticello, AR. Both sites had site indices over 80 feet at 50 years for oaks.

Each site was divided into two blocks, and seedlings were planted at an 8- by 10-foot spacing. A border row of oaks was planted around the outside of the study. After planting, a random sample of 480 seedlings (220 per seedling type) was assessed for initial height and ground line diameter (gld). After leaf-out in late April, all seedlings were assessed for survival.

A general linear model was used to assess differences in height and gld among sites and seedling types. All factors were considered random for the analysis. Interaction terms among the variables were pooled into the error term since there was no biological interaction immediately following planting and before growth initiated. Survival differences were assessed using a general linear mixed model (PROC GLIMMIX; SAS®) with a specification of a binomial distribution logit link function. Potential interaction effects were included in this model since there was a potential biological interaction. Variable means were calculated using a least square means estimation for each significant random source effect.

RESULTS AND DISCUSSION

Rot collar diameter was significantly affected by seedling type ($p = 0.02$) when seedlings taken directly from cold storage were sampled. Improved seedlings were 27 percent greater in root collar diameter (fig. 1). Conversely, root length was statistically the same ($p = 0.98$) for both seedling types (21.8 inches) indicating effective and homogenous undercutting prior to seedling harvest.

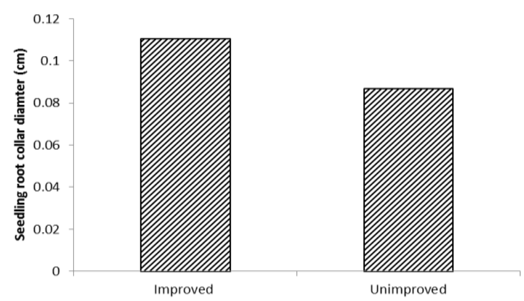

Figure 1–Average root collar diameters for random samples of improved and unimproved seedlings taken directly from the Baucum Nursery, North Little Rock, AR.

Planting height and gld were both significant and only affected by seedling type ($p < 0.001$ for both). The unimproved seedlings were 8 percent shorter in height than the improved seedlings (fig. 2a). While the difference is not as exaggerated as the root collar findings from the nursery or the height differences, the gld of unimproved seedlings was 9 percent less than

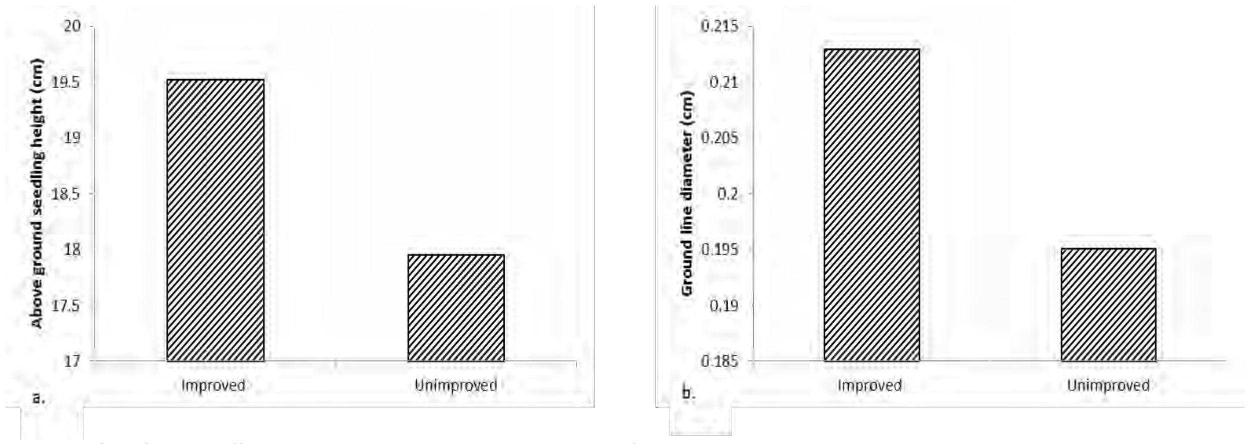

Figure 2–Significantly different (α = 0.05) seedling type averages of (a) above ground height and (b) ground line diameter across two sites in southern Arkansas immediately after planting.

the improved seedlings (fig. 2b). Gld and root collar diameter would be expected to be very similar from random samples. The smaller differences and overall smaller widths measured at ground line diameters probably demonstrate that many of the seedlings were planted with root collars slightly below ground level.

Overall, seedlings that come from the improved seedling stock are inherently larger than their unimproved counterparts in both above-ground height and gld. This could have a great impact on future needs, and costs, of herbicide application or manual weeding of herbaceous competition. Greater initial height of the improved seedlings may allow them to overtop competing vegetation more quickly and avoid negative impacts to growth and development (Kolb and Steiner 1990, Lorimer and others 1994). Potentially, this could lead to a reduction in the number of herbicide applications needed during establishment and offsetting some of the higher costs of the improved seedlings.

These initial planted seedling measurements support results from the independent samples taken at the AFC storage area, where improved seedling root collar diameter was significantly larger by an average of 0.06 cm. However, this larger improved seedling does not seem to lead towards better survival immediately following leaf-out. In the survival analysis, seedling type also significantly affected ($p = 0.01$) the probability of survival. The improved seedlings were 6 percent less likely to survive to 3 months after planting with 85.8 percent surviving opposed to 91.5 percent for the unimproved seedlings. This is surprising as the improved seedlings were quantitatively larger and their roots were visually more robust in lateral root size and length and larger in overall root biomass. Generally, more developed roots of 1-0 seedlings are associated with better survival (Thompson and Schultz 1995). In this study, the extreme horizontal root mass caused difficulties in planting. Planters were highly supervised and were aided by broad oak-style dibble bars. Still, many of the roots were so massive that proper orientation and protection of roots was marginal. Undercutting of the seedlings was conducted at the nursery regulating total length of the seedlings which has been shown in some cases to improve survivorship (Zaczek and others 1997). Still, no control of the width or mass of the roots is conducted which may pose problems with planting in normal field conditions.

In conclusion, the cherrybark oak genetic improvement efforts have produced larger seedlings than their woods-run counterparts. This potentially will alleviate costs of competition control because seedlings will have a chance to emerge from the understory more quickly with the initial head-start. At the same time survivorship was lower which is, in part, attributed to planting difficulties of large seedlings. Still, later data from this study, not yet published, is pointing toward a closure of the survivorship gap or better survival of the improved seedlings. Thus, these very initial results must be weighed with caution since the study is a long-term one.

ACKNOWLEDGMENTS

We would like to extend our appreciation to the Arkansas Forestry Commission for the donation of seedlings and time, notably David Bowling and Allan Murray. We also would like to acknowledge Tom Byram with the Western Gulf Tree Improvement Cooperative for advice on the study. Finally, we would like to acknowledge the many student workers who have helped conduct field work and the graduate student Nick Mustoe who also assisted and will be using this study for his future thesis. Funding for this project was provided by a UAM mini-grant.

LITERATURE CITED

Adams, J.C. 1989. Five year growth results of water oak (*Quercus nigra* L.) provenances. In: Proceedings of the 20th southern forest tree improvement conference. Sponsored Publ. No. 42: 310-314.

Adams, J.P.; Rousseau, R.J.; Adams, J.C. 2006. Genetic performance and maximizating genetic gain through direct and indirect selection in cherrybark oak. Silvae Genetica. 56: 80-87.

Dubois, M.R.; Chappe ka, A.H.; Robbins, E. [and others]. 2000. Tree shelters and weed control: effects on protection, survival and growth of cherrybark oak seedlings planted on a cutover site. New Forests. 20: 105-118.

Ezell, A.W.; Catchot, A.L., Jr. 1998 Competition control for hardwood plantation establishment. 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: 42-43.

Ezell, A.W.; Hodges, J.D. 2002. Herbaceous weed control improves survival of planted shumard oak seedlings. 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: 273-275.

Ezell, A.W.; Yeiser, J.L. 2007. Survival of planted oak seedlings is improved by herbaceous weed control. Weed Technology. 21: 175-178.

Greene, T.A.; Lowe, W.J.; Stine, M. 1991. Volume production of six cherrybark oak provenances in the Western Gulf region. 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: 736-743.

Gwaze, D.P.; Byram, T.D.; Raley, E.M. 2003. Performance of Nuttal oak (*Quercus texana* Buckl.) provenances in the western gulf region. Proceedings of the 27th southern forest tree improvement conference: 122-137.

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. 36: 34-44.

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: 227-237.

Schuler, J.L.; Robison, D.J.; Quicke, H.E. 2004. Assessing the use of chopper herbicide for establishing hardwood plantations on a cutover site. Southern Journal of Applied Forestry. 28: 163-170.

Thompson, J.R.; Schultz, R.C. 1995. Root system morphology of *Quercus rubra* L. planting stock and 3-year field performance in Iowa. New Forests. 9: 225-236.

Zaczek, J.; Steiner, K.; Bowersox, T. 1997. Northern red oak planting stock: 6-year results. New Forests. 13: 177-191.

EFFECTS OF LATE ROTATION THINNING ON LIGHT AVAILABILITY AND RED OAK REGENERATION WITHIN A MINOR STREAM BOTTOM IN MISSISSIPPI

Ellen M. Boerger, Brent R. Frey, Andrew W. Ezell, and Tracy Hawkins[1]

Abstract–Recent studies suggest a troubling decline in the abundance of red oak species (*Quercus* spp., Section *Erythrobalanus*) in bottomland forests of the southeastern United States. We assessed red oak advance regeneration and associated tree species in relation to light availability in a 77-year-old oak-dominated stand 5 years after late rotation thinning. Residual basal areas across four thinning treatments ranged between 48 and 69 square feet per acre and were compared to an unthinned control (108 square feet per acre). Available understory light was significantly greater in the most intensive treatment (48 square feet per acre) compared to the unthinned control. Red oak advance regeneration was significantly taller in thinned areas. For the tallest height class (48+ inches), the control area contained only 44 red oak seedlings per acre, while the highest thinning intensity treatment contained more than six times the number of large red oaks (272 red oak seedlings per acre). Relative height of red oak seedlings (the tallest red oak as a proportion on the tallest non-oak seedling) did not differ among treatments. Higher intensity late rotation thinning regimes appeared to be beneficial to increasing red oak advance regeneration vigor, which may aid their progress into the overstory canopy level upon a harvest release. Further monitoring through harvest is needed. Late rotation thinning may provide managers an effective means for enhancing the regeneration of red oak species in future regeneration treatments.

INTRODUCTION

Oaks are dominant components of minor stream bottom hardwood forests in the southeastern United States. Recent studies suggest a troubling decline in the abundance of red oak species in the Southeast (Oliver and others 2005). The presence of oak seedlings can be variable depending on factors such as mast production, flooding duration, light levels, and species present, among others. Once red oak seedlings become established, progressing these seedlings from the understory into the overstory canopy can be difficult, especially with a lack of available understory light.

Most oaks are classified as shade-intolerant, so their exposure to ample understory light is vital for their survival and growth in bottomland hardwood forests (Carvell and Tryon 1961, Logan 1965, McGee 1968). Hodges and Gardiner (1993) reported that low light availability may be the most limiting factor to oak regeneration, and the range of 27 to 53 percent light exposure resulted in maximum growth of these oak seedlings (Gardiner and Hodges 1998). Studies examining the effects of canopy gap conditions on regeneration have indicated that oak seedlings are found more often in larger (> 0.25 acre) gaps (Holladay and others 2006). As further evidence of light limitation of oak seedlings, silvicultural treatments such as midstory control have been used to increase light penetration to the understory to increase understory oak seedling growth (Ezell and others 1993, Loftis 2004, Peairs and others 2004). As a stand-alone treatment, midstory removal may not be sufficient to increase light levels to understory seedlings in closed canopy riparian forests (Cunningham and others 2011, Ostrom and Loewenstein 2006); however, studies have shown that a combination of midstory removal and partial overstory harvest can increase light levels and provide favorable conditions for oak regeneration (Cunningham and others 2011, Peairs 2003).

Bottomland hardwood forests typically contain variable spacing of desirable species and high-quality stems (Meadows and Skojac 2012). Therefore, bottomland thinning operations are more likely to create gaps than thinning operations in more uniform stands with frequently occurring crop trees (e.g. simple conifer stand types, plantations, or upland hardwoods) (Meadows and Skojac 2012). Thinning is an intermediate treatment that develops residual stems by regulating stand density and aims to improve growth and development, modify species composition, and improve overall bole quality of residual stems

[1]Graduate Assistant, Assistant Professor, and Professor, respectively, Mississippi State University, Forest and Wildlife Research Center, Mississippi State, MS 39762; and Research Ecologist, USDA Forest Service, Southern Research Station, Starkville, MS 39762.

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(Meadows 1996, Smith and others 1997). Although not the intent, thinning treatments may aid in establishment of advance regeneration that could be released upon final harvest (Lockhart and others 2004). Gaps created during thinning provide increased light availability, which is favorable to developing oak advance regeneration (Collins and Battaglia 2008, Lockhart and others 2004). Thus thinning may provide a critical opportunity for developing oak regeneration (Meadows and Stanturf 1997). Thinning may negatively impact oak regeneration due to machine traffic. However, Lockhart and others (2000) reported that damage during thinning operations would have a negligible effect on the development of the advance regeneration of red oaks prior to final harvest. At the same time, increased light conditions provided by thinning operations can also create opportunities for growth and development of competitors of red oak. These tradeoffs associated with thinning effects have not been well investigated.

An improved understanding of the impacts of thinning on regeneration of red oaks is needed, particularly in terms of relationships among residual basal area, available understory light and development of advance regeneration. The objectives of this study were: (1) to evaluate the light levels across different thinning intensities, and (2) to evaluate the effects of thinning on the development of bottomland hardwood red oak and associated species. In this way, this study should help managers determine the efficacy of late-rotation thinning for enhancing the regeneration of red oak species in future regeneration treatments.

MATERIALS AND METHODS

Study Area

This study was conducted within a 77-year-old minor stream bottom hardwood stand within the Samuel D. Hamilton Noxubee National Wildlife Refuge (Noxubee NWR) in Noxubee County, in east-central Mississippi (N33°15' W88°43', elevation ca. 250 feet). Noxubee NWR received 60.75 inches precipitation in 2012, with the highest monthly average occurring in July (National Climatic Data Center 2013). The predominant soil type in this area is the Urbo-Mantachie association (NRCS 2012). This area is adjacent to the Noxubee River and is subject to periodic flooding events, especially through the winter months. The site supports an approximately 70-year-old mixed species hardwood stand, dominated by red oaks (*Quercus* spp.) and sweetgum (*Liquidambar styraciflua* L.). The red oak species represented 51 percent of stand basal area prior to treatment and were primarily comprised of cherrybark (*Q. pagoda* Raf.) and water oak (*Q. nigra* L.), with a lesser component of willow oak (*Q. phellos* L.). Sweetgum comprised 23 percent of stand basal area prior to treatment, with hickory (*Carya* spp.), green ash (*Fraxinus pennsylvanica* Marsh.), swamp chestnut oak (*Q. michauxii* Nutt.), overcup oak (*Q. lyrata* Walt.), and American elm (*Ulmus americana* L.) comprising the remaining basal area (Meadows and Skojac 2012).

Experimental Design

This study area was established in 2007 to investigate effects of late rotation thinning on stand development and mast production (Meadows and Skojac 2012). The study was established as a completely randomized block design containing four thinning intensities (48, 57, 61, and 69 square feet per acre residual basal area) and an uncut control (108 square feet per acre residual basal area) with three replications of each totaling 15 treatment areas of 264 feet by 330 feet each (table 1). Sites were thinned using Stand Quality Management specifications, which favor the retention of the highest quality trees (species and form) regardless of spacing (Meadows and Skojac 2012).

Regeneration Sampling

Each experimental unit was sampled using a series of subplots, centered off a randomly located point halfway between the bole and dripline of individual red oak canopy trees. Seedlings were sampled at 53 individual red oak stems within the study area, averaging 2 to 3 sample points per treatment. At each sampling

Table 1—Basal area (BA) by treatment pre-and post-thinning in a minor stream bottom hardwood forest in east-central Mississippi: initial, residual, and year 3 BA (modified from Meadows and Skojac 2012)

Treatment	Initial BA Oct 2007	Residual BA Apr-May 2008	Year 3 BA Jan 2011
	----------------*square feet per acre*-------------------		
Control	113	108	112
Acceptable/superior	122	69	68
Acceptable/no pole	108	61	64
Desirable/superior	126	57	59
Desirable/no pole	112	48	51

point, four circular 0.002-acre (5.3-foot radius) subplots were positioned at 90° angles at a distance of 19.7 feet from each randomly selected point. In each subplot, individual seedlings were identified by species, and seedling heights were recorded.

Understory Light Sampling

Hemispherical photography was used to estimate light availability based on global site factor (GSF) indices derived from processed digital images. GSF is the proportion of global radiation (direct plus diffuse radiation) at a specific location (i.e. under the forest canopy) in relation to that in the open condition (Anderson 1964, Delta-T Devices 1999). Hemispherical photographs were taken at the center of each regeneration plot at the study area in July and August 2012. Photographs were taken in the morning before sunrise using a 3.34 megapixel Nikon Coolpix 990 camera with a Nikon LC-ER1 fisheye lens attachment (Nikon, Melville, NY). The camera system was placed into a self-leveling mount (Delta-T Devices, Cambridge, United Kingdom) attached to a camera tripod. The camera position was 5 feet above ground level, oriented north, and the timer setting was used. Hemispherical canopy images were analyzed with HemiView Analysis 2.1 software (Delta-T Devices 1999) to calculate GSF. Threshold values were determined for individual photographs (Anderson 1964, Rich 1990).

Data Analysis

Data were analyzed using analysis of variance (ANOVA) by PROC GLM procedure in SAS 9.3 (SAS Institute Inc., Cary, NC). Significance was evaluated at α = 0.05. We compared light availability (GSF), seedling height, and red oak relative height (tallest red oak as percent of tallest non-oak) across treatments (averaged at the plot level, n = 3 per treatment). GSF and seedling height values were square root transformed to meet ANOVA assumptions. Tukey's HSD test was used for multiple comparisons.

RESULTS

Light

Across the treatments, GSF was significantly different (P = 0.0409). The highest intensity thinning treatment (48 square feet per acre; mean GSF = 0.2896; SE = 0.094) provided over twice the amount of light to the understory than the control (108 square feet per acre; mean GSF = 0.1324; SE = 0.010) (fig. 1). There were no differences among the other thinning treatments and the control.

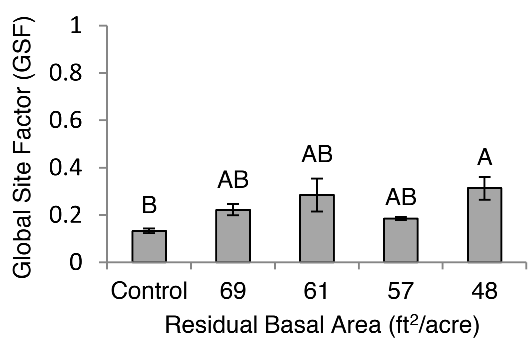

Figure 1—Mean global site factor (GSF) by treatment (shown as residual basal area square feet per acre) 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Letters signify differences at α = 0.05, using Tukey's HSD for multiple comparisons. Error bars represent ± standard error of the mean.

Composition and Frequency

Across the entire site, frequency of red oak advance regeneration was between 20 to 35 percent, depending upon species (fig. 2).

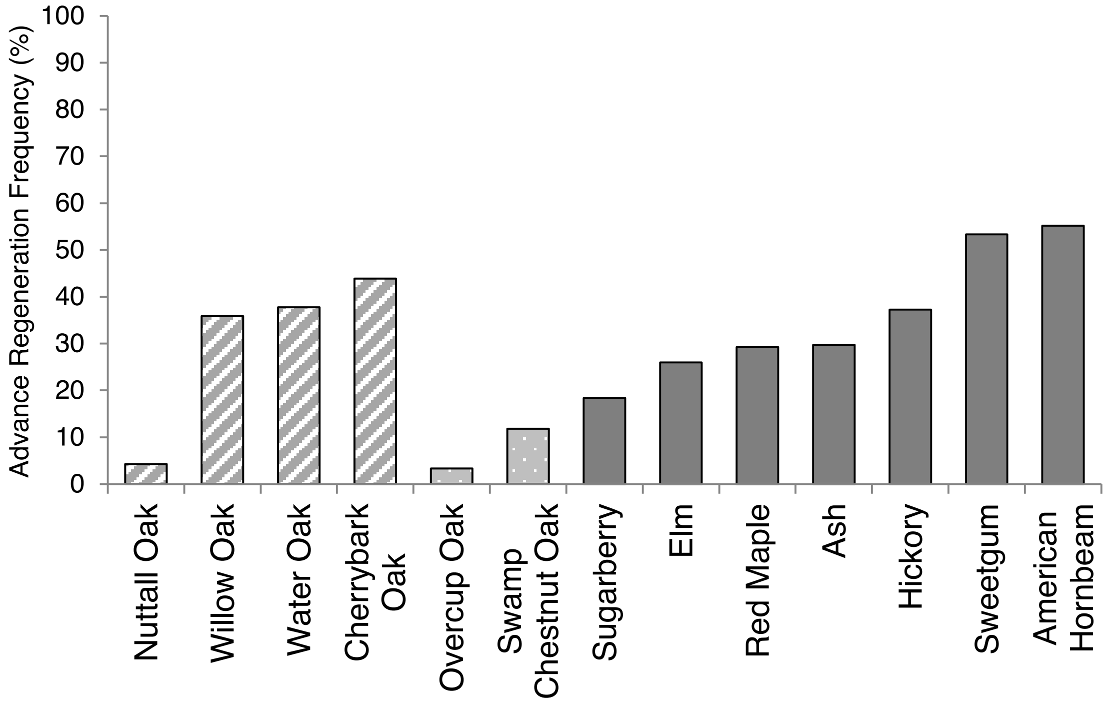

Figure 2—Frequency (percent) of advance regeneration by species 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Red oaks are indicated by striped bars, white oaks by dotted bars, and non-oak species by solid bars.

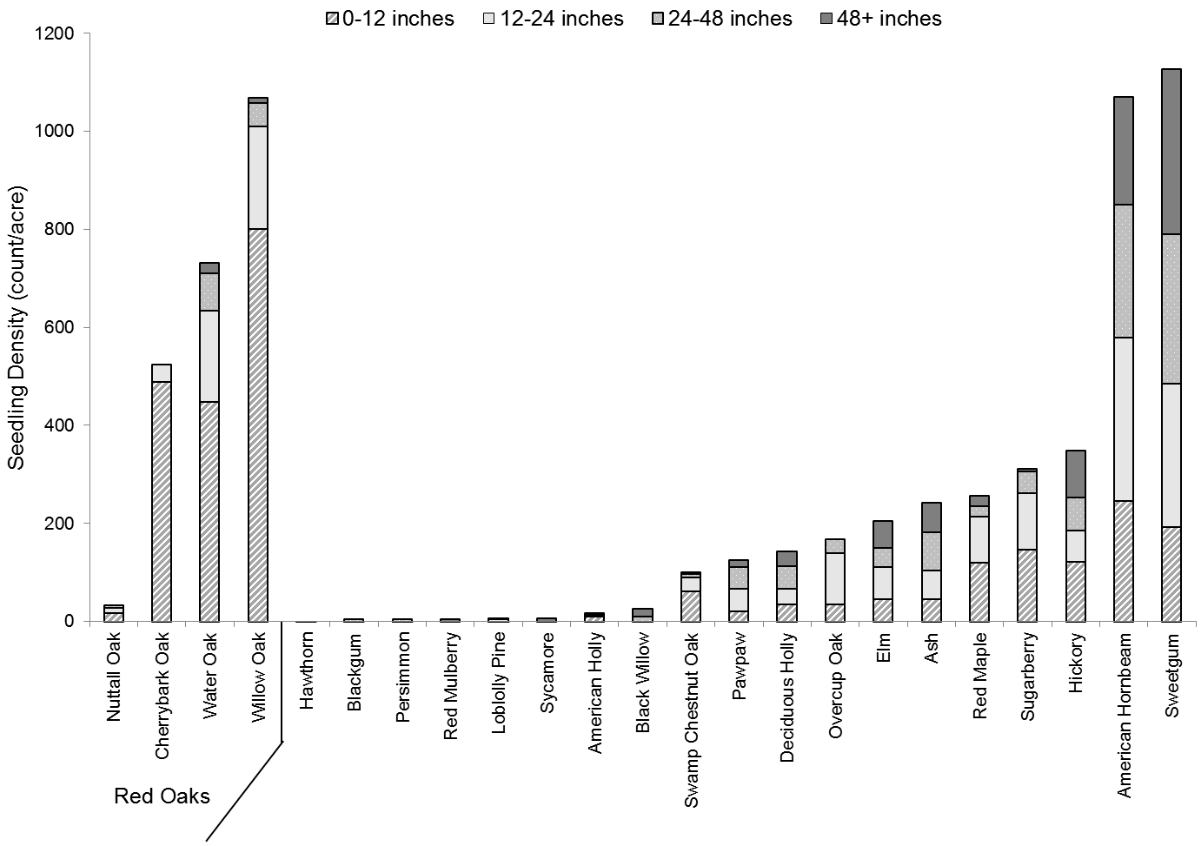

Figure 3—Seedling density per acre of advance regeneration by species and height class (0-12, 12-24, 24-48, and 48+ inches) 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Species not previously identified in the text: hawthorn (*Crataegus* spp.), black gum, (*Nyssa sylvatica* Marsh.), persimmon (*Diospyros virginiana* L.), red mulberry (*Morus rubra* L.), loblolly pine (*Pinus taeda* L.), sycamore (*Platanus occidentalis* L.), American holly (*Ilex opaca* Aiton), black willow (*Salix nigra* Marsh.), pawpaw [*Asimina triloba* (L.) Dunal], and deciduous holly (*Ilex decidua* Walt.).

Willow oak, water oak, and cherrybark oak were the most abundant red oaks; Nuttall oak (*Q. texana* Buckl.) seedlings were rare on the site in any size class. The most frequent non-oak species were American hornbeam (*Carpinus caroliniana* Walt.) and sweetgum (50 to 60 percent); however, other frequent non-red oak species included hickory (*Carya spp.*), ash (*Fraxinus spp.*), red maple (*Acer rubrum* L.), elm (*Ulmus spp.*), and sugarberry (*Celtis laevigata* Willd.) (fig. 2). Across the entire site, most red oaks fell within the 0 to 12 inches height class and had few stems within the largest height classes (fig. 3). Most of the advance regeneration in the largest height classes (24 to 48 inches and 48+ inches) included sweetgum, American hornbeam, hickory, ash, and elm (fig. 3).

Seedling Density and Height

There were no treatment effects on red oak mean seedling density (P = 0.8441) or aggregate height (P = 0.8721). When red oak seedling density was separated into height classes (0-12, 12-24, 24-48, and 48+ inches) little variation in seedling density was apparent in the shorter (0-12, 12-24 inch) height classes. However, there were more seedlings in the taller height classes (48+ inches) in the higher intensity thinning treatments (especially 48 square feet per acre) than in the control (108 square feet per acre) (fig. 4). For the tallest seedlings (48+ inches), the control contained only 44 red oak seedlings per acre, while the most intensive thinning treatment (48 square feet per acre) contained more than six times the number of red oaks (272 red oak seedlings per acre). Red oak mean height was also significantly (P = 0.0238) greater in the highest intensity thinning treatment (48 residual square feet per acre) compared to the control (108 residual square feet per acre) (fig. 5).

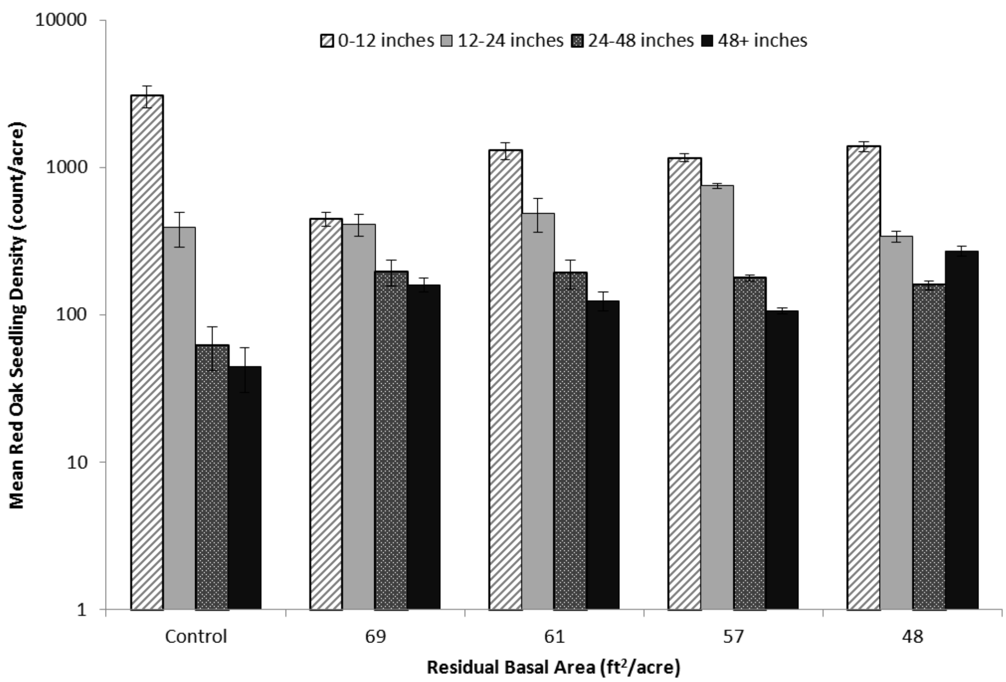

Figure 4—Density of red oak advance regeneration per acre by treatment (residual basal area; square feet per acre) and height class (0-12, 12-24, 24-48, 48+ inches) 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Error bars represent ± standard error of the mean. Note: seedling density is presented on a logarithmic scale

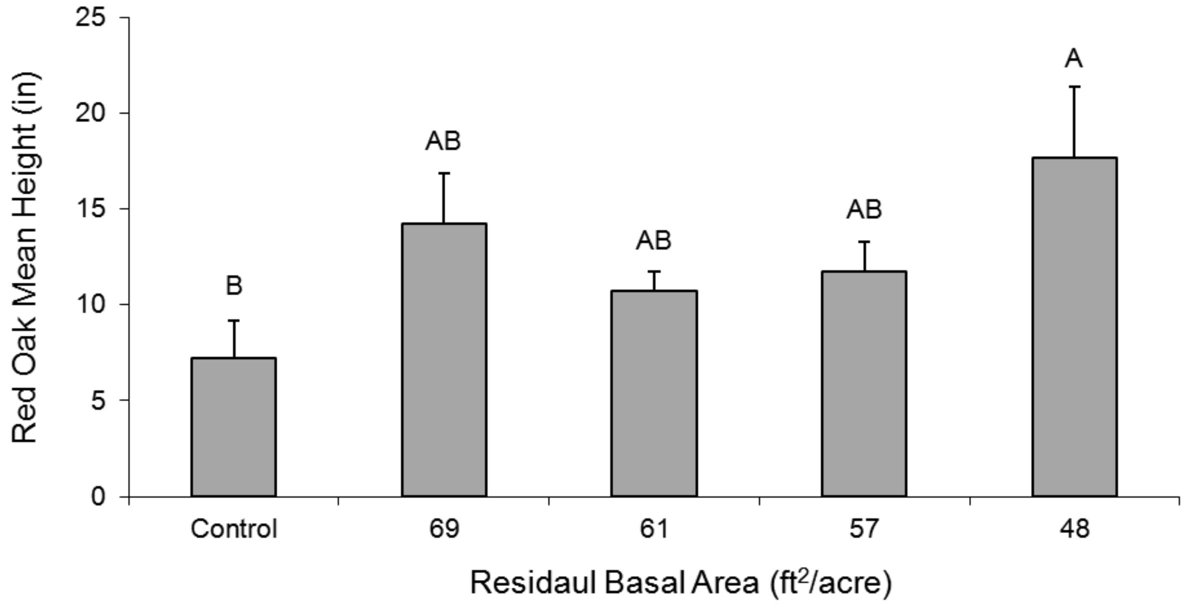

Figure 5—Mean height (inches) of red oak advance regeneration by treatment (shown as residual basal area square feet per acre) 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Letters signify differences at α = 0.05, using Tukey's HSD for multiple comparisons. Error bars represent ± standard error of the mean

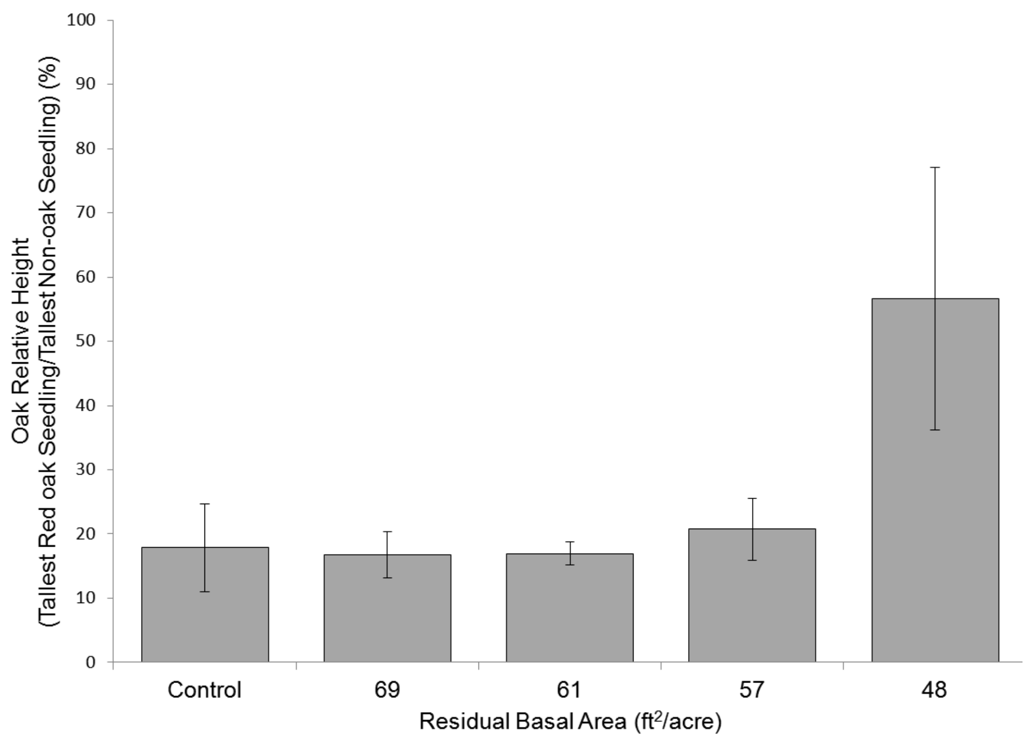

Figure 6— Relative height of oak seedlings as a percent of the tallest non-oak seedling by treatment (residual basal area; square feet per acre) 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Error bars represent ± standard error of the mean.

Competition

The difference between the single tallest non-oak seedling and single tallest red oak seedling was determined at 53 individual overstory stems. The oak relative height (tallest red oak seedling divided by tallest non-oak seedling) was not significantly different by treatment (P = 0.0731) (fig. 6). The overall average difference in height of these tallest seedlings across the entire site was 9.2 feet. The most common tallest red oak species were water oak and willow oak, while the most common tallest non-oak species included sweetgum, ash, and American hornbeam. The mean height of the most common species varied by species (fig. 7)

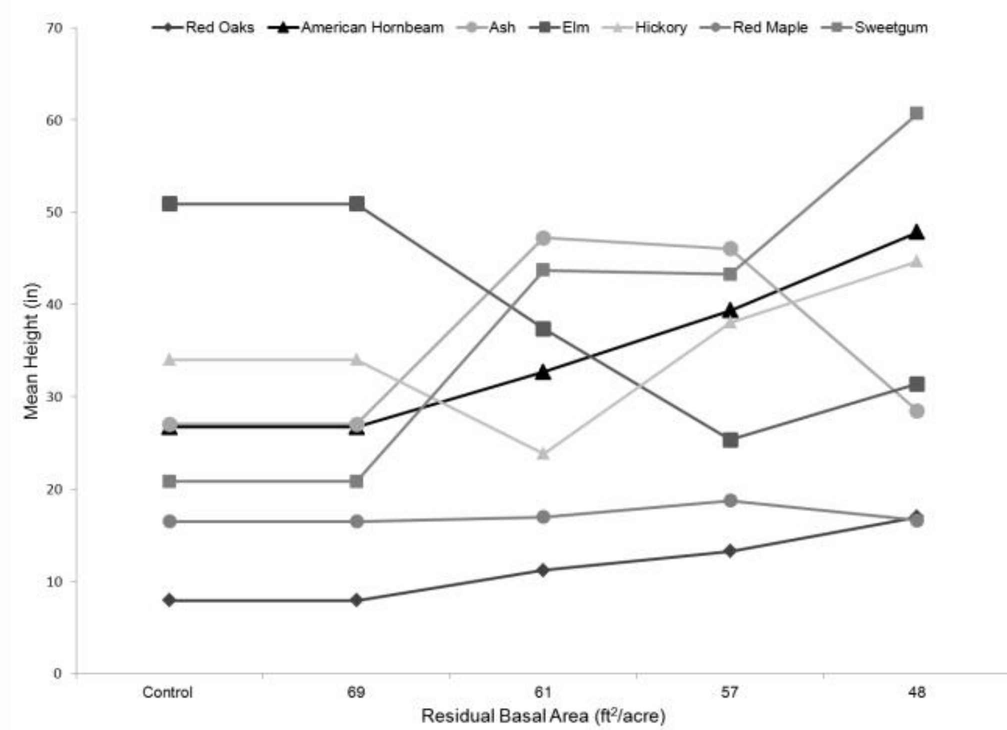

Figure 7— Mean height (inches) of red oaks, American hornbeam, ash, elm, hickory, red maple, and sweetgum species by treatment (residual basal area; square feet per acre) 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi.

DISCUSSION

Five years after thinning, available understory light increased with thinning intensity in a minor bottom hardwood forest. Light regimes are a critical determinant of red oak seedling survival and growth, especially after the first year when oak seedlings are no longer reliant on seed reserves (Crow 1988). Although it is possible for red oak seedlings to persist at very low light levels, such as those in the control treatments, the optimal range of light reported for red oak seedling growth is typically moderate levels between 20 to 53 percent (Gardiner and Hodges 1998, Gottschalk 1994, Guo and others 2001, Phares 1971). Light levels recorded in unthinned controls in this study were comparable to those found in midstory removal in closed canopy forests (14 percent; Cunningham and others 2011), uninjected greentree reservoir control areas (12.5 to 15.3 percent; Guttery and others 2011), and slightly higher than the untreated bottomland stands in the region (8.8 percent; Peairs 2003). As for thinned areas, the most intensively thinned areas in this study were similar in range (22.47 to 44.7 percent) for midstory injection reported by Guttery and others (2011) Significantly higher numbers of tall red oak seedlings occurred in thinning treatments compared to the control, indicating an increase in seedling vigor. The mean red oak seedling height was significantly greater in the most intensive thinning treatment (48 square feet per acre), which suggests that intensive late-rotation thinning treatments aid in increasing the vigor of red oak advance regeneration 5 years after thinning. Larsen and others (1997) also reported overstory basal areas > 87.1 square feet per acre failed to meet the standards necessary for adequate oak regeneration size and density. Although the intent of thinning regimes is solely to increase the quality of the residual timber, the simultaneous opening of the canopy can provide additional benefits by increasing the vigor of advance regeneration (Lockhart and others 2004). These benefits gained through thinning treatments should be kept in mind for stand management plans and could potentially reduce the need for further management to obtain adequate oak regeneration or reduce undesirable competition after final harvest (Meadows and Stanturf 1997).

CONCLUSIONS

Five years after thinning, light availability increased with increasing thinning intensity and provided favorable conditions for red oak advance regeneration. Significantly higher numbers of tall red oak seedlings occurred in thinning treatments than in the control, indicating an increase in seedling vigor. Consequently, late rotation thinning treatments can aid in development of red oak advance regeneration to increase their stature and place them in a more competitive position for advancement upon release by final harvest. Future observations of advance regeneration through harvest are needed. Additionally, species composition and competition are important factors controlling regeneration, and evaluation of these dynamics will improve our knowledge of oak regeneration.

ACKNOWLEDGEMENTS

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LITERATURE CITED

Anderson, M.C. 1964. Studies of the woodland light climate I. The photographic computation of light condition. Journal of Ecology. 52(1): 27-41.

Carvell, K.L.; Tryon, E.H. 1961. The effect of environmental factors on the abundance of oak regeneration beneath mature stands. Forest Science. 7: 98-105.

Collins, B.; Battaglia, L.L. 2008. Oak regeneration in southeastern bottomland hardwood forest. Forest Ecology and Management. 255(7): 3026-3034.

Crow, T.R. 1988. Reproductive mode and mechanisms for self-replacement of northern red oak. Quercus rubra - a review. Forest Science. 34(1): 19-40.

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.

Delta-T Devices. 1999. HemiView canopy analysis software. User manual version 2.1. Cambridge, UK: Delta-T Devices Ltd. 79 p.

Ezell, A.W.; Nelson, L.R; Schoenholtz, S.H.; Williams, J.G. 1993. Controlling undesirable stems using low intensity injections of imazapyr. In: Brisette, J.C., ed., Proceedings of the 7th biennial southern silvicultural research conference. Gen. Tech. Rep. SO-93. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 455-459.

Gardiner, E.S.; Hodges, J.D. 1998. Growth and biomass distribution of cherrybark oak (Quercus pagoda Raf.) seedlings as influenced by light availability. Forest Ecology and Management. 108(1-2): 127-134.

Gottschalk, K.W. 1994. Shade, leaf growth and crown development of Quercus rubra, Quercus velutina, Prunus serotina and Acer rubrum seedlings. Tree Physiology. 14(7-9): 735-749.

Guo, Y.; Shelton, M.G.; Lockhart, B.R. 2001. Effects of light regime on the growth of cherrybark oak seedlings. Forest Science. 47(2): 270-277.

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(3): 187-203.

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. Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 54-65.

Holladay, C.A.; Kwit, C.; Collins, B. 2006. Woody regeneration in and around aging southern bottomland

PERFORMANCE OF TWO OAK SPECIES AND THREE PLANTING STOCKS ON LANDS DAMAGED BY HURRICANE KATRINA

John A. Conrad III, Andrew W. Ezell, Emily B. Schultz, and John D. Hodges[1]

Abstract--Hurricane Katrina had a devastating impact on bottomland hardwood forests in 2005. Artificial regeneration was considered the most appropriate method for reforesting these areas, but few studies have evaluated methods for artificially regenerating oaks on clear cut sites in the southern United States. First-year survival and growth of two oak species, live oak (*Quercus virginiana* Mill.) and Nuttall oak (*Quercus texana* Buckl.), and three planting stocks [1-0 bareroot, conventional containerized, and Root Production Method (RPM)™ seedlings] were compared. Seedlings were established using recommended methods for each respective planting stock. Conventional containerized live oak and bareroot Nuttall oak seedlings exhibited the greatest survival. RPM™ seedlings exhibited the lowest survival in both species. Conventional containerized seedlings exhibited greater groundline-diameter (GLD) growth and twice as much height growth as bareroot seedlings, regardless of species. RPM™ seedlings exhibited similar GLD growth compared to bareroot seedlings but the least height growth of all planting stocks, regardless of species.

INTRODUCTION

Bottomland hardwood forests were severely impacted by Hurricane Katrina in 2005 (Chapman and others 2008). Prompted by uncertainty about the future of these stands and the impact the storm had on non-industrial private landowners, Congress developed a cost share program to provide an incentive for landowners to regenerate hardwoods. Wildlife habitat and timber production were priority objectives of most landowners, and many desired to develop stands abundant in oaks. Although natural regeneration is the preferred method for regenerating oaks, artificial regeneration was considered more appropriate given the unplanned circumstance.

Artificial regeneration of bottomland hardwoods has traditionally been accomplished using 1-0 bareroot seedlings, but success has often been impeded by competing vegetation, drought, flooding, and herbivory (Stanturf and others 2004). Recent decades have been marked by efforts to reduce the impact of these factors by improving nursery, site preparation, and competition control techniques. The use of high-quality seedlings combined with herbaceous weed control has greatly improved survival of bareroot seedlings (Ezell and Catchot 1998, Ezell and Hodges 2002, Ezell and others 2007), and based on projections made by Grebner and others (2003), the practice is likely to be cost effective. Concurrently, techniques for containerized seedling production have improved, and some sources indicate that interest in using containerized seedlings is increasing (Dey and others 2008).

Compared to bareroot seedlings, containerized seedlings have a more fibrous root system which is better protected from damage and less vulnerable to desiccation prior to and during planting. Several studies have documented greater survival and growth of containerized seedlings compared to bareroot seedlings (Humphrey 1994, Rathfon and others 1995, Self and others 2010, Williams and Craft 1998). Others have also shown improved survival and growth with increasing container size (Howell and Harrington 2002). A relatively new method, the Root Production Method™ (RPM™), is a system of nursery techniques that has resulted in the development of large containerized seedlings with extensive root systems, characteristics which could potentially make them more resilient to drought, flooding, and terminal shoot-removal by white-tailed deer (*Odocoileus virginianus*) (Dey and others 2006). Some studies have reported greater early survival and growth of RPM™ seedlings compared to 1-0 bareroot seedlings (Dey and others 2003) and conventional-sized containerized seedlings (Alkire 2011). Although large containerized seedlings may have potential survival and growth advantages compared to smaller seedling types, they are extremely costly to purchase and plant (Shaw and others 2003).

[1]Graduate Research Assistant, Professor, and Professor, respectively, Mississippi State University, Forest and Wildlife Research Center, Mississippi State, MS 39759; and Professor Emeritus, Ashland, MS 39576.

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Comparative evaluations of planting stock performance are needed to help foresters make well-informed decisions about regeneration. While results of previous studies comparing oak planting stocks are available in the literature, most studies have been conducted on retired agricultural lands in major river bottoms and only involve the evaluation of a few key oak species (Dey and others 2008). Information comparing the performance of planting stocks on cutover or storm-damaged lands is lacking. The objective of this study was to compare the survival and growth performance of two oak species, live oak (*Quercus virginiana* Mill.) and Nuttall oak (*Quercus texana* Buckl.), and three planting stocks: 1-0 bareroot seedlings, conventional containerized seedlings, and RPM™ seedlings.

MATERIALS AND METHODS

Research was conducted on two privately owned tracts near Hattiesburg, MS. Both sites were positioned on minor stream bottoms. Average annual precipitation for the area is 158.7 cm. The Guiles tract is located in Perry County, approximately 16 km east of Hattiesburg. The soil series was Trebloc silt loam, and soil sample analysis indicated that the pH was 4.6. Following salvage, residual stems were injected with a 20 percent aqueous solution of Arsenal® AC (imazapyr), and the site was mechanically prepared using a roller drum chopper attached to a bulldozer. The site received a direct foliar application of 2 percent Accord® XRT solution in November 2010 to control winged sumac (*Rhus copallinum* L.). Herbicide was applied with a Solo® 11.4-L piston-pump backpack sprayer.

The second site, the Morgan Tract, is located in Forrest County, MS approximately 14 km northeast of Hattiesburg. The soil present was a complex of Bibb and Jena soil series. Analysis of soil samples indicated that the soil was a sandy-loam with a pH of 5.4. This site was sheared, raked, burned, and cleared with a bulldozer to prepare the site for planting. The intensity of mechanical site preparation abated the need for chemical site preparation on this site.

Two oak species, Nuttall oak and live oak, and three planting stocks [high-quality 1-0 bareroot, 0.24-L conventional containerized, and 11.4-L RPM™ seedlings] were selected for evaluation. Bareroot seedlings were purchased from Molpus Woodlands Group™ in Elberta, AL.

Conventional containerized seedlings were purchased from Rennerwood Incorporated™ in Tennessee Colony, TX. RPM® seedlings were purchased from RPM Ecosystems™ in Ocean Springs, MS.

Three hundred seedlings per species and planting stock combination were planted on each site. Bareroot seedlings and conventional containerized seedlings were planted by Mississippi State University personnel on the first two weekends of February 2011. Bareroot seedlings were planted with OST dibble bars. Conventional containerized seedlings were planted with planting shovels. RPM™ seedlings were planted by a RPM Ecosystems™ planting crew on February 19, 2011 using planting shovels. Seedlings were planted next to pre-marked pin flags to ensure uniform spacing.

RPM™ seedlings were established according to the company's "plant and walk away" approach, and thus did not receive herbaceous weed control. Bareroot and conventional containerized seedlings were treated with a post-plant, pre-bud break application of Oust® XP (140 g/ sprayed ha) in February 2009. An 11.4-L Solo® piston-pump backpack sprayer equipped with a TeeJet 8003 Visiflo® nozzle, specially designed to minimize wind drift, was used to apply the herbicide as a 1.5-m band over the top of seedlings. Herbicide was applied in the morning when wind was minimal as a primary precaution to avoid herbicide drift into untreated plots.

Survival and seedling parameter measurements were recorded at the conclusion of the first growing season. The cambial layer was nicked to affirm suspected mortality during each survival evaluation. Groundline diameter (GLD) and height measurements were recorded 2 weeks after seedlings were planted and at the conclusion of the first growing season. GLD measurements were recorded to the nearest tenth of a mm using Mititoyo® digital calipers. Height measurements were taken at the top of the living portion of the stem in the advent that a seedling exhibited dieback. Height of bareroot and conventional containerized seedlings was measured to the nearest cm using a meter stick. Height of RPM® seedlings was measured to the nearest tenth of a foot using a telescopic height pole.

The experimental design was randomized complete block. Each species and planting stock combination was replicated three times per site. Analysis of variance (ANOVA) was used to determine significant differences between species and among planting stocks. Mean separation was performed using the GLIMMIX procedure of Statistical Analysis Software (SAS) version 9.2®. Differences were considered significant when $P < 0.05$. Survival percentages were arcsine transformed prior to analysis, but actual means are presented for the purpose of interpretation.

RESULTS

Survival

All species and planting stocks exhibited lower survival on the Guiles site compared to the Morgan site (table 1). On the Guiles site, survival of bareroot live oak, conventional containerized Nuttall oak seedlings, and RPM™ seedlings of both species declined below 80 percent, which was considered a critical level (Ezell and Hodges 2002). Survival of conventional containerized live oak and bareroot Nuttall oak remained high at 86.0 and 87.7 percent, respectively. On the Morgan site, survival of conventional containerized live oak and all Nuttall oak planting stocks exceeded 95 percent. Lower survival on the Guiles site was attributed to greater competition. The Morgan site, which received more intensive site preparation, was mostly void of vegetation during May and June. It is thought that the lack of competition on the Morgan site during these early months allowed seedlings a better opportunity to overcome transplant stress.

Planting-stock comparisons were performed separately by species. For live oak, conventional containerized seedlings exhibited greatest survival (92.3 percent overall), and bareroot and RPM™ live oak seedlings exhibited similar survival (77.8 and 76.6 percent overall, respectively). Wind damage was common in live oak RPM™ seedlings. Many RPM™ live oak seedlings were either leaning or bent, and some were completely laid over with their root systems exposed. In contrast, bareroot seedlings exhibited the greatest survival in Nuttall oak (93.8 percent overall), and conventional containerized and RPM™ seedlings exhibited similar survival (87.5 and 84.1 percent overall, respectively). Wind damage was less prevalent in RPM™ Nuttall oak seedlings, which did not have their leaves when planted.

Table 1--Percent survival by species, planting stock, and site after the first growing season

Planting stock[a]	Guiles site[bc]	Morgan site[bc]	Overall[bc]
	percent survival		
Live oak			
BRT	68.0 Bb	87.7 Bb	77.8 Bb
CC	86.0 Aa	98.7 Aa	92.3 Aa
RPM™	70.6 Bb	82.6 Bb	76.6 Bb
Nuttall oak			
BRT	87.7 Aa	100.0 Aa	93.8 Aa
CC	77.3 Bb	97.7 Bb	87.5 Bb
RPM™	73.2 Ba	95.1 Ba	84.1 Ba

[a]BRT = bareroot, CC = conventional containerized, RPM™ = Root Production Method™.
[b]Means followed by the same uppercase letter in a column within a species do not differ significantly at α = 0.05.
[c]For each respective planting stock, means followed by the same lowercase letter in a column do not differ signficantly between species at α = 0.05.

GLD and Height Growth

Similar to survival, all species and planting stocks exhibited lower GLD and height growth on the Guiles site compared to the Morgan site (table 2). On both sites, Nuttall oak exhibited significantly greater GLD growth than live oak in all planting stocks. Conventional containerized seedlings exhibited the greatest overall GLD growth in live oak and Nuttall oak (5.7 and 7.9 mm overall, respectively). RPM™ seedlings exhibited slightly greater GLD growth than bareroot seedlings overall, but the difference was not significant in live oak (3.7 and 3.3 mm overall, respectively) or Nuttall oak (6.2 and 5.7 mm overall, respectively).

Height growth of live oak was comparable to Nuttall oak in both bareroot (9.7 and 11.2 cm overall, respectively) and conventional containerized planting stocks (18.9 and 20.3 cm overall, respectively) (table 2). In both species, conventional containerized seedlings exhibited

Table 2--Mean initial groundline diameter (GLD) and height plus first-year growth by species, planting stock, and site

Planting stock[c]	GLD (*mm*)[ab] Initial growth	First-year growth Guiles site	First-year growth Morgan site	Overall	Height (*cm*)[ab] Initial growth	First-year growth Guiles site	First-year growth Morgan site	Overall
Live oak								
BRT	6.1	1.6 ABb	4.5 Bb	3.3 Bb	43.6	2.2 Aa	15.3 Ba	9.7 Aa
CC	5.4	2.3 Ab	8.6 Ab	5.7 Ab	73.5	6.4 Aa	29.4 Aa	18.9 Aa
RPM™	16.5	1.4 Bb	5.3 Bb	3.7 Bb	201.8	-32.1 Bb	-41.9 Cb	-37.9 Bb
Nuttall oak								
BRT	7.2	3.6 Ba	7.5 Ba	5.7 Ba	50.2	10.2 Ba	12.1 Ba	11.2Ba
CC	5.8	4.3 Aa	10.6 Aa	7.9 Aa	50.0	13.8 Aa	25.2 Aa	20.3 Aa
RPM™	20.0	3.9 Ba	8.1 Ba	6.2 Ba	196.7	3.8 Ca	7.4 Ca	5.8 Ca

[a]Means followed by the same uppercase letter in a column within a species do not differ significantly ($\alpha = 0.05$).
[b]For each respective planting stock, means followed by the same lowercase letter in a column do not differ significantly between species ($\alpha = 0.05$).
[c]BRT = bareroot, CC = conventional containerized, RPM™ = Root Production Method™

greater mean height growth than bareroot seedlings, but the difference between these planting stocks in live oak was not significant on the Guiles site. RPM™ seedlings exhibited the least height growth in both species. The overall mean height growth of RPM™ Nuttall oak seedlings was 5.8 cm, but live oak RPM™ seedlings decreased in height by 37.9 cm.

DISCUSSION

Although seedlings are subject to a variety of detrimental factors during the first few growing seasons, competing vegetation is the most consistent factor limiting survival and growth (Ezell and others 2007, Russell and others 1998). In this study, sites differed appreciably in the level of competition, and this probably led to differences in survival and growth (tables 1 and 2). Although seedling survival and growth differed between sites, species and planting stock comparisons were, for the most part, consistent between sites.

Survival and growth of RPM™ seedlings was less than expected. In both species, RPM™ seedlings exhibited lower overall survival than bareroot or conventional containerized seedlings. This was primarily attributed to wind damage. Seedlings are known to increase diameter growth near the stress point in response to wind damage (Close and others 2010). The severity of damage was not reflected in GLD growth. However, it was reflected in relatively lower height growth. RPM™ live oak seedlings exhibited a 20 percent decrease in mean height (table 2). Dieback due to wind damage was presumably more severe in RPM™ live oak seedlings because they retained their leaves when planted. RPM™ Nuttall oak seedlings exhibited a slight increase in mean height (5.8 cm), but it was approximately two and three times less than that exhibited by bareroot and conventional containerized Nuttall oak seedlings [11.2 and 20.3 cm, respectively (table 2)]. In contrast, Alkire (2011) reported 23.8 cm height growth for RPM™ Nuttall oak seedlings after the first growing season, which was more than twice as much exhibited by bareroot or containerized seedlings during the first growing season.

In two preceding trials (Alkire 2011, Hollis 2011), bareroot seedlings exhibited greater than or comparable first-year survival and growth to conventional containerized seedlings when only bareroot seedlings received herbaceous weed control (HWC). In this trial, when both planting stocks received HWC, conventional containerized seedlings exhibited no clear advantage with respect to survival. Live oak conventional containerized seedlings exhibited appreciably greater survival than bareroot seedlings (92.3 and 77.8 percent, overall), but Nuttall oak bareroot seedlings exhibited the highest survival of all species and planting stocks (93.8 percent) (table 1). In both species, however, conventional containerized seedlings exhibited significantly greater GLD and height growth than bareroot seedlings. This was expected, because containerized seedlings have been shown to be less susceptible to transplant shock (Johnson and others 1984, Wilson and others 2007). An unexpected result, however, was that in bareroot and conventional containerized stock, height growth for live oak was comparable to Nuttall oak, which is known to exhibit more rapid juvenile growth than other bottomland oaks (Allen 1990, Miwa and others 1992).

Few trials have evaluated oak planting stock performance past the second growing season, but results of more long-term studies have been inconsistent. Teclaw and Isebrands (1993) reported greater height growth of containerized seedlings compared to bareroot seedlings after the third growing season, but survival of both planting stocks was high (94 and 98 percent, respectively). Howell and Harrington (2004) reported that height growth increased with increasing container size after 5 years, but survival was high (> 90 percent) for all seedling types. In contrast, Burkett and others (2005) reported that after seedlings were subjected to severe herbivory, bareroot seedlings exhibited similar or greater survival than containerized seedlings and had surpassed them in height after the fifth growing season. In a comparison of bareroot and RPM™ seedlings, Dey and others (2006) reported greater survival of RPM™ seedlings, but both planting stocks exhibited a decline in mean height after the third growing season due to severe herbivory by eastern cottontail rabbits (*Sylvilagus floridanus*). Mullins and others (1998) reported no significant differences in height or diameter between bareroot, conventional containerized seedlings, or large, greenhouse-forced seedlings by the end of the fourth growing season. Moreover, use of tree shelters to protect seedlings from herbivory did not increase height growth. In an 8-year trial, Henderson and others (2009) reported higher relative growth rates of bareroot seedlings compared to RPM™ seedlings and no significant differences in survival between the two planting stocks.

CONCLUSIONS

More long-term evaluations are needed to determine if early survival and growth differences among planting stocks persist. Based on first-year results, it appears that 1-0 bareroot seedlings, the least costly seedling type, can exhibit acceptable survival when competition is controlled for at least part of the first growing season (Ezell and Hodges 2002). Conventional containerized seedlings may be the more suitable choice when herbaceous weed control cannot be implemented, or for landowners willing to spend more for establishment. They are also recommended when plantings must be conducted during the growing season (Stanturf and others 2004). Large containerized seedlings may be best suited to situations when management objectives require fewer seedlings per acre, whereby more intensive management techniques could be financially feasible.

LITERATURE CITED

Alkire, D.K. 2011. Artificial regeneration of bottomland hardwoods in southern Mississippi on lands damaged by Hurricane Katrina. Mississippi State, MS: Mississippi State University. 62 p. M.S. thesis.

Allen, J.A. 1990. Establishment of bottomland oak plantations on the Yazoo National Wildlife Refuge Complex. Southern Journal of Applied Forestry. 14(4): 206-210.

Burkett, V.R.; Draugelis-Dale, R.O.; Williams, H.M.; Shoenholtz, S.H. 2005. Effects of flooding regime and seedlings treatment on early survival and growth of Nuttall oak. Restoration Ecology. 13(3): 471-479.

Chapman, E.L.; Chambers, J.Q.; Ribbeck, K.F. [and others]. 2008. Hurricane Katrina impacts on forest trees of Louisiana's Pearl River basin. Forest Ecology and Management. 256(5): 833-889.

Close, D.C.; Beadle, C.L.; Brown, P.H. 2005. The physiological basis of containerised tree seedling 'transplant shock': a review. Australian Forestry. 68(2): 112-120.

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.; Kabrick, J.M.; Gold, M.A. 2003. Tree establishment in floodplain agroforestry practices. In: Sharrow, S.H., ed. Agroforestry and riparian buffers for land productivity and environmental stability. Proceedings of the 8th North American agroforestry conference. Corvallis, OR: Oregon State University: 102-115.

Dey, D.C.; Kabrick, J.M.; Gold, M.A. 2006. The role of large container seedlings in afforesting oaks in bottomlands. 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: 218-223.

Ezell, A.W.; Cachot, A.L., Jr. 1998. Competition control for hardwood plantation establishment. 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: 42-43.

Ezell, A.W.; Hodges, J.D. 2002. Herbaceous weed control improves survival of planted shumard oak seedlings. 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: 273-275.

Ezell, A.W.; Yeiser, J.W.; Nelson, L.R. 2007. Survival of planted oak seedlings is improved by herbaceous weed control. Weed Technology. 21: 175-178.

Grebner, D.L.; Ezell, A.W.; Gaddis, D.A.; Bullard, S.H. 2003. Impacts of southern oak seedling survival on investment returns in Mississippi. Journal of Sustainable Forestry. 17(3): 1–19.

Henderson, D.E.; Botch, P; Cussimanio, J. [and others]. 2009. Growth and mortality of pin oak and pecan reforestation in a constructed wetland: analysis with management implications. Science and Management Tech. Series. No. 1. Jefferson City, MO: Missouri Department of Conservation. 19 p.

Hollis, D.B. 2011. Survival and growth of three oak planting stocks on Hurricane Katrina disturbed lands. Mississippi State, MS: Mississippi State University. 59 p. M.S. thesis.

Howell, K.D.; Harrington, T.B. 2004. Nursery practices influence seedling morphology, field performance, and cost efficiency of containerized cherrybark oak. Southern Journal of Applied Forestry. 28(3): 152-162.

Humphrey, M. 1994. The influence of planting date on the performance of bare-root, container grown, and direct seeded *Quercus nuttallii* Nuttall oak on Sharkey soil. Lorman, MS: Alcorn State University. 52 p. M.S. thesis.

Johnson, P.S.; Novinger, S.L.; Mares, W.G. 1984. Root, shoot, and leaf area growth potentials of northern red oak planting stock. Forest Science. 30(4): 1017-1026.

Miwa, M.; Schoenholtz, S.H.; Hodges, J.D.; Kennedy, H.E., Jr. 1992. First-year results of bottomland oak reestablishment in alluvial soils of the lower Mississippi valley. In: Brissette, J.C., ed. Proceedings of the 7th biennial southern silvicultural research conference. Gen Tech. Rep. SO-93. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 73-79.

Mullins, J.A.; Buckner, E.R.; Hopper, G.; Percell, G.G. 1998. Early growth of sheltered and unsheltered cherrybark oak established by planting 1-0 bareroot and 1-0 containerized seedlings, and by direct seeding. In: Waldrop, T.A., ed. Proceedings of the 9th biennial southern silvicultural conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 234-237.

Rathfon, R.A.; Kaczmarek, D.J.; Pope, P.E. 1995. Site preparation for red oak plantation establishment on old field sites in southern Indiana. In: Gottschalk, K.W.; Fosbroke, S.L.C., eds. Proceedings of the 10th central hardwood conference. Gen. Tech. Rep. NE-197. Radnor, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station: 349-362.

Russell, D.R., Jr.; Hodges, J.D.; Ezell, A.W. 1998. An evaluation of hardwood reforestation methods on previously farmed lands in central Alabama. In: In: Waldrop, T.A., ed. Proceedings of the 9th biennial southern silvicultural conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 272-276.

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 a retired agricultural site. 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: 159-163.

Shaw, G.W.; Dey, D.C.; Kabrick, J. [and others]. 2003. Comparison of site preparation methods and stock types for artificial regeneration of oaks in bottomlands. In: Van Sambeek, J.W.; Dawson, J.O.; Ponder, F., Jr. [and others], eds. Proceedings of the 13th central hardwood forest conference. Gen. Tech. Rep. NC-234. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Research Station: 186-198.

Stanturf, J.A.; Conner, W.H.; Gardiner, E.S. [and others]. 2004. Recognizing and overcoming difficult site conditions for afforestation of bottomland hardwoods. Ecological Restoration. 22(3): 183-193.

Teclaw, R.M.; Isebrands, J.G. 1993. Artificial regeneration of northern red oak in the Lake States with a light shelterwood: a departure from tradition. In: Gillespie, A.R.; Parker, G.R.; Pope, P.E.; Rink, G., eds. Proceedings of the 9th central hardwood forest conference. Gen. Tech. Rep. NC-161. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station: 185-194.

Williams, H.M.; Craft, M.N. 1998. First-year survival and growth of bare-root, container, and direct-seeded Nuttall oak planted on flood-prone agricultural fields. 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: 300-303.

Wilson, E.R.; Vitols, K.C.; Park, A. 2007. Root characteristics and growth potential of container and bare-root seedlings of red oak (*Quercus rubra* L.) in Ontario, Canada. New Forests. 34: 163–176.

SIX-YEAR EFFECT OF MIDSTORY REMOVAL ON WHITE OAK GROWTH AND BIOMASS DISTRIBUTION AND SEEDLING RESPONSE ONE YEAR POST-CLIPPING

Jared M. Craig, John M. Lhotka, and Jeffrey W. Stringer[1]

Increasing abundance of shade tolerant trees in oak- (*Quercus* spp.) dominated forests has led to enhanced concern regarding the recruitment of oak into the overstory following disturbance. The key to future oak success is through developing large advance reproduction. One proposed method with demonstrated success is midstory removal. This technique increases light to the understory, which may allow oak to gain a competitive advantage over shade-tolerant species (Lockhart and others 2000, Lorimer and others 1994, Parrott and others 2012). Basal shoot clipping has also been suggested as a way to develop more vigorous oak seedlings (Lockhart and others 2000). What is still unknown, however, is what effect a midstory removal treatment has on oak seedling biomass and whether any increases in size provide an advantage to seedlings following clipping. Our first objective was to document biomass accumulation and allocation trends of white oak (*Q. alba* L.) advance reproduction 6 years following midstory removal. We also examined the effect of basal shoot clipping 6 years following midstory removal on white oak sprout growth through one growing season.

The study was established in three stands of intermediate site quality (average site index of 19.5 m) located on Berea College Forest in Madison County, KY. Stand overstories were primarily composed of oak and hickory (*Carya* spp.), while understories were dominated by red maple (*Acer rubrum* L.) and American beech (*Fagus grandifolia* Ehrh.). In 2005, two 0.4-ha plots were established at each site, and the plots were randomly assigned midstory removal and control treatments. The midstory removal treatment reduced basal area by 20 percent and removed overtopped and intermediate crown class trees. Pretreatment data indicated that oak seedling size did not differ between treatments, and mean oak seedling height and ground line diameter (gld) were 9.7 cm and 3.0 mm, respectively.

Six years after midstory removal, 30 white oak seedlings were selected from each plot, and height and gld were recorded. Seedling leaf area was estimated using midrib length and leaf area relationships presented in Parrott and others (2012). In the winter prior to the seventh growing season, the selected white oak seedlings were clipped at their base using hand shears. Seedling tops were oven dried and weighted to determine above-ground biomass. Eighty additional white oak seedlings were excavated, and oven-dried root weights from these seedlings were used to develop allometric root biomass equations. Equations were then applied to height and gld data from the clipped seedlings to estimate below-ground biomass. One growing season after basal clipping, height and gld of the seedling sprouts were measured.

Results indicate that midstory removal significantly increased white oak seedling size and above- and below-ground biomass accumulation after six growing seasons (table 1). Stem and root biomass were more than three times larger in the midstory removal treatment than in the control. However, biomass allocation fractions between stem and root did not differ between treatments. The sprout probability following basal clipping was similar between the midstory removal (86 percent) and control (85 percent) treatments. Sprout height and gld one growing season following basal clipping was significantly larger in the midstory removal treatment than in the control (table 1). However, no treatment effect was found for the relative growth response by clipped seedlings. Among treatments, 66 percent of pre-clipping height and 60 percent of pre-clipping gld were achieved one growing season following clipping.

[1]Forester, Ohio Department of Natural Resources, Shawnee State Forest, Portsmouth, OH 45663; and Assistant Professor and Professor, respectively, University of Kentucky, Department of Forestry, Lexington, KY 40546.

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Table 1--Mean (±SE) comparisons between control and midstory removal treatments for white oak seedling measurements with associated t-test p-values

Variable	Control	Midstory removal	p-value
Six-year seedling characteristics (pre-clipping)			
Height (cm)	24.30 ± 1.17	36.00 ± 2.30	<0.001
Ground line diameter (mm)	2.99 ± 0.14	4.87 ± 0.30	<0.001
Total leaf area (cm^2)	239.30 ± 56.74	1087.10 ± 229.20	<0.001
Above-ground biomass (g)	1.39 ± 0.21	6.02 ± 1.35	0.008
Below-ground biomass (g)	4.20 ± 0.44	13.13 ± 2.05	<0.001
Total biomass (g)	5.21 ± 0.65	18.46 ± 2.92	<0.001
Seedling characteristics 1 year post-clipping			
Sprout height (cm)	15.20 ± 0.75	21.33 ± 1.33	<0.001
Sprout ground line diameter (mm)	1.75 ± 0.10	2.68 ± 0.17	<0.001

Study findings suggest that midstory removal increased biomass accumulation of white oak advance reproduction but did not alter the allocation of biomass between above- and below-ground portions. Results also suggest that short-term response to basal clipping was linked to seedling size and not conditions inherent to the midstory removal treatment. Although results are from one growing season following clipping, it appears that the combination of midstory removal and subsequent basal clipping may aid in promoting larger white oak advance reproduction. Subsequent monitoring will provide insight into the long-term response of clipped seedlings developing in the two treatments evaluated by this study.

LITERATURE CITED

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(1): 45-50.

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(2): 227-237.

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. Northern Journal of Applied Forestry. 29(4): 182-190.

THIRD-YEAR RESPONSE OF OAK NATURAL REPRODUCTION TO A SHELTERWOOD HARVEST AND MIDSTORY COMPETITION CONTROL IN THE ARKANSAS OZARKS

K. Kyle Cunningham[1]

Abstract--A study evaluating the response of oak natural reproduction to a shelterwood harvest and midstory competition control in an upland hardwood stand within the Ozark Highlands of Arkansas was conducted. The study site was located in the dissected Springfield Plateau physiographic region on the University of Arkansas, Division of Agriculture Livestock and Forestry Research Station near Batesville, AR. Five-acre treatment plots were established within a 140-acre shelterwood harvest on north-facing slopes (site indices 65 to 75 for oaks) in a 110-year-old upland hardwood stand. The overstory was dominated by white oak (*Quercus alba* L.), black oak (*Q. velutina* Lam.), and northern red oak (*Q. rubra* L.). Treatments include: (1) shelterwood harvest to 50 square feet per acre (BA50); (2) shelterwood harvest to 50 square feet per acre plus injection of non-oak midstory trees (1 to 5 inches d.b.h.; BA50+MR); and (3) non-harvested control (NHC). Initial mean oak seedlings per acre (SPA) were 308 (± 49.8); 613 (± 123.8); and 548 (± 72.0) for BA50, BA50+MR, and NHC, respectively. Year 3 post-treatment mean oak SPAs were 1,698 (± 365); 3,070 (± 807.8); and 3,340 (± 990.3), respectively. A significant difference was detected between initial versus year 3 values for all three treatments ($p = 0.003$, 0.008, and 0.007 respectively; student's t-test, $\alpha = 0.05$). No differences in total oak SPA were detected between treatments within year 3. Significant differences were detected between initial and year 3 oak SPA by height class for BA50 and BA50+MR. BA50+MR exhibited the greatest changes in oak seedling height growth.

INTRODUCTION

Throughout the hardwood forests of North America, regenerating oak stands on productive upland sites presents a major problem to resource managers (Brose and others 1998). The physiological and morphological adaptations of oak seedlings often narrow the environmental conditions in which they survive and grow. A basic assumption is that success in survival and growth is influenced by: (1) microclimate and edaphic factors, (2) morphological and physiological characteristics of a particular species, and (3) interaction between the two (Hodges and Gardiner 1993). Understanding these relationships is important to understanding management strategies for perpetuating oaks into new forests. Most hardwood stands do not have adequate sunlight penetration through the upper canopy for the development of oak seedlings. Undisturbed, "closed" canopies often result in sunlight canopy penetration to the ground level of less than 10 percent (Canham and others 1990, Cunningham and others 2011) Low understory light levels in hardwood stands may be the most limiting factor to the establishment and growth of oak reproduction (Hodges and Gardiner 1993). Battaglia and others (2000) stated that environmental factors such as light and soil moisture may have independent or interacting influence on hardwood seedling survival and growth. Quero and others (2008) found that irradiance levels have greater impact on seedling growth than water supply.

An increase of sunlight aids in promoting both the successful establishment and subsequent growth of advance oak reproduction in hardwood stands. However, too much light in the initial stages of development may hinder oak seedlings by favoring faster growing, more shade-intolerant, tree species and herbaceous vegetation (Hodges and Janzen 1986). Hodges and Gardener (1993) detected that sufficient sunlight levels for growth and survival for cherrybark oak (*Q. pagoda* Raf.), under controlled conditions, occurred at 27 percent of total available photosynthetically active radiation (PAR) and optimal growth conditions occur at 53 percent of total available PAR. Recent advances in applied research for oak natural regeneration have established combinations of partial overstory harvests and midstory competition control to improve environmental conditions for developing oak reproduction (Cunningham and others 2011; Larsen and Johnson 1998; Loftis 1990, 1993). Sources for hardwood regeneration include seedlings, seedling sprouts, and stump sprouts. Seedlings present prior to harvest are known as advance reproduction (Rogers and others 1993). The level of partial overstory removal may affect the amount of advance

[1]Extension Forestry Instructor, University of Arkansas, Division of Agriculture, Arkansas Forest Resources Center, Little Rock, AR 72203.

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reproduction present following harvesting activity as it impacts both the amount of site disturbance and resulting available sunlight.

Shelterwood harvests may present the most flexible option for naturally regenerating desirable species such as oaks. A shelterwood harvest is a management system that promotes a standing crop of reproduction through a series of partial removals of the overstory (Smith and others 1996). An alternate version of the classical approach to shelterwood harvests may be required for desirable oak species on more productive sites. Combining herbicide treatments and/or prescribed fire along with the shelterwood has been evaluated by many researchers (Hicks and others 2001).

Although there are no universal prescriptions for the hardwood regeneration problem, modified shelterwood systems that remove canopy and sub-canopy individuals prior to overstory removal to increase light reaching the forest floor can increase seedling dominance and survival for desirable species such as oaks. This study attempts to further supplement our knowledge of oak natural regeneration by evaluating irradiance effects and hardwood reproduction response to two shelterwood methods.

MATERIALS AND METHODS

The study site is 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. 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 on broad ranges of slope positions. The areas selected for this study were only on north aspects, which potentially had somewhat higher organic matter, higher moisture content, and are 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

Three treatments with four replicates were incorporated into a completely randomized design. Treatments included: (1) shelterwood harvest to basal area (BA) 50 (BA50); (2) shelterwood harvest to BA 50 plus injection of non-oak stems between 1 and 5 inches diameter at breast height (d.b.h.) (BA50+MR); and (3) non-harvested control (NHC). Midstory removal treatments were applied from November 2008 to February 2009. Follow-up treatments were applied in July 2009. Midstory removal was performed using herbicide injection. A 0.03-ounce aqueous solution of 25 percent imazapyr and 75 percent water was injected for every 3 inches of diameter around tree trunks. A partial overstory harvest operation was applied to the BA50 and BA50+MR from October 2009 through March 2010. The target residual basal area was 50 square feet per acre. Desirable residual tree characteristics were: (1) oak species and (2) large vigorous crowns.

Each treatment replicate contained twelve 0.01-acre circular regeneration sample plots spaced on a grid along the slope gradient. Stand-level reproduction measurements at each plot included species and height class (< 1 foot, 1 to 3 feet, and > 3 feet). Overstory measurements were taken from two (one upper slope and one lower slope) 0.20-acre circular plots. Overstory measurements included species, d.b.h., merchantable height, log grade, damage, and number of epicormic branches. Midstory measurements were taken from two 0.05-acre circular plots. Midstory measurements included species and total height. Initial overstory, midstory and understory measurements were taken in summer 2009.

Photosynthetic Photon Flux Density (PPFD) was measured at each of the 12 regeneration plots per replicate. PPFD was measured at plot center using a microquantum sensor attached to a Mini-PAM 2000 (WALZ, Inc.). Mini-PAM light readings were calibrated against an additional quantum light sensor (Delta OHM LP 471, λ400 to 700 nm) for accuracy. The sensor was mounted to a leveled tripod at each measurement point. Plot center light measurements were taken in September of each growing season.

Statistical Analyses

All statistical analyses were performed in SAS 9.2 (SAS Inc., Cary, NC). Normality tests were performed through the PROC UNIVARIATE procedure utilizing the Shapiro-Wilks W-test. Data were also tested for equal variances. When necessary, regeneration data were square root transformed to meet required assumptions. Sunlight level and regeneration response were analyzed for treatment differences using analysis of variance (ANOVA) using PROC GLM. Individual means separation was conducted using Student Newman-Kuels (SNK) tests.

RESULTS

Initial overstory mean BA of treatment replicates was 93.8 square feet per acre (± 8.5 square feet), representing a fully stocked to slightly overstocked stand. Initial overstory species composition was dominated by approximately 75 percent oak species. Initial mean midstory density was 310 trees per acre (TPA) and dominated by non-oak, shade tolerant species. An appreciable portion of the midstory consisted of large crowned flowering dogwood (*Cornus florida* L.). Mean understory density included 475 oak seedlings per acre (SPA) (± 147) and 2,532 non-oak SPA (± 366). Species composition in the understory was dominated by shade-tolerant species. Red maple (*Acer rubrum* L.), winged elm (*Ulmus alata* Michx.), and hickory (*Carya* spp.) comprised 48 percent of understory, while oaks comprised 15 percent.

Treatment applications had appreciable impact on residual overstory and midstory conditions. For the BA50 and BA50+MR, post-treatment BAs ranged from 45 to 55 square feet per acre, resulting in approximately a 55 percent reduction in overstory density. The NHC remained at initial BA levels (approximately 93.8 square feet per acre). The BA50 midstory density was approximately 125 TPA for upper-slope plots and 190 TPA for lower-slope plots. The mean post-treatment, midstory density for BA50 was 157.5 TPA, resulting in approximately a 51 percent reduction from initial midstory density. The BA50+MR midstory trees were approximately 100 percent removed (approximately 310 TPA). The NHC midstory densities were approximately the same as initial stand conditions.

Irradiance

Mean mid-day PPFDs following treatment applications were 507, 744.5 and 72.1 μmol m^{-2} s^{-1}, respectively for BA50, BA50+MR, and NHC (fig. 1a). Sunlight canopy penetration to ground level was 28, 42, and 5 percent of full sunlight for BA50, BA50+MR, and NHC, respectively. A one-way ANOVA detected significant differences to exist ($p = 0.003$) for treatment effects. A SNK means analysis detected significant differences to exist between all three treatments. Sunlight levels increased from lower slope to upper slope positions for all treatments (fig. 1b). A significant difference ($p = 0.02$) was detected between upper slope (527.2 μmol m^{-2} s^{-1}) and lower slope (355.3 μmol m^{-2} s^{-1}).

Regeneration

Year 3 mean oak SPA for BA50, BA50+MR, and NHC were 1,698; 3,070; and 3,340, respectively (fig. 2a). Final oak SPA numbers represented an increase from initial numbers of 1,389; 2,456; and 2,792 for BA50, BA50+MR, and NHC, respectively (table 1). Paired t-test comparisons detected a significant difference between initial versus year 3 oak SPA for BA50, BA50+MR, and NHC ($p = 0.003$, 0.008, and 0.007, student's t-test, $\alpha = 0.05$). A one-way ANOVA for year by treatment detected significant differences between years (square root transformed; $p < 0.001$). SNK means separation procedure detected significant differences between year 3 versus year 2, year 1, and initial oak SPA and year 2 versus year 1 and initial oak SPA. No differences were detected between year 1 and initial oak SPA. A one-way ANOVA established differences among treatments across years (square root transformed; $p = 0.003$). A SNK means analysis detected significant differences between year 3 BA50+MR and NHC versus all treatments by year.

Year 3, mean oak SPA for BA50 were 1,354; 279; and 65 for height classes 1, 2, and 3, respectively. This was an increase of 1,062; 148; and 38 over initial values for height classes 1, 2, and 3. Paired t-test detected significant differences between year 3 versus initial oak SPA for all three height classes in BA50 (square root transformed; $p = 0.01$, 0.04, and 0.007, $\alpha = 0.05$). Year 3 mean oak SPA for BA50+MR were 2,105; 746; and 218 for height classes 1, 2, and 3, respectively. This was an increase of 1,807; 492; and 158 over initial values for BA50+MR oak SPA (fig. 3). Paired t-test detected significant differences for all height classes between initial and year 3 values (square root transformed; $p = 0.04$, 0.03, and 0.04, $\alpha = 0.05$).

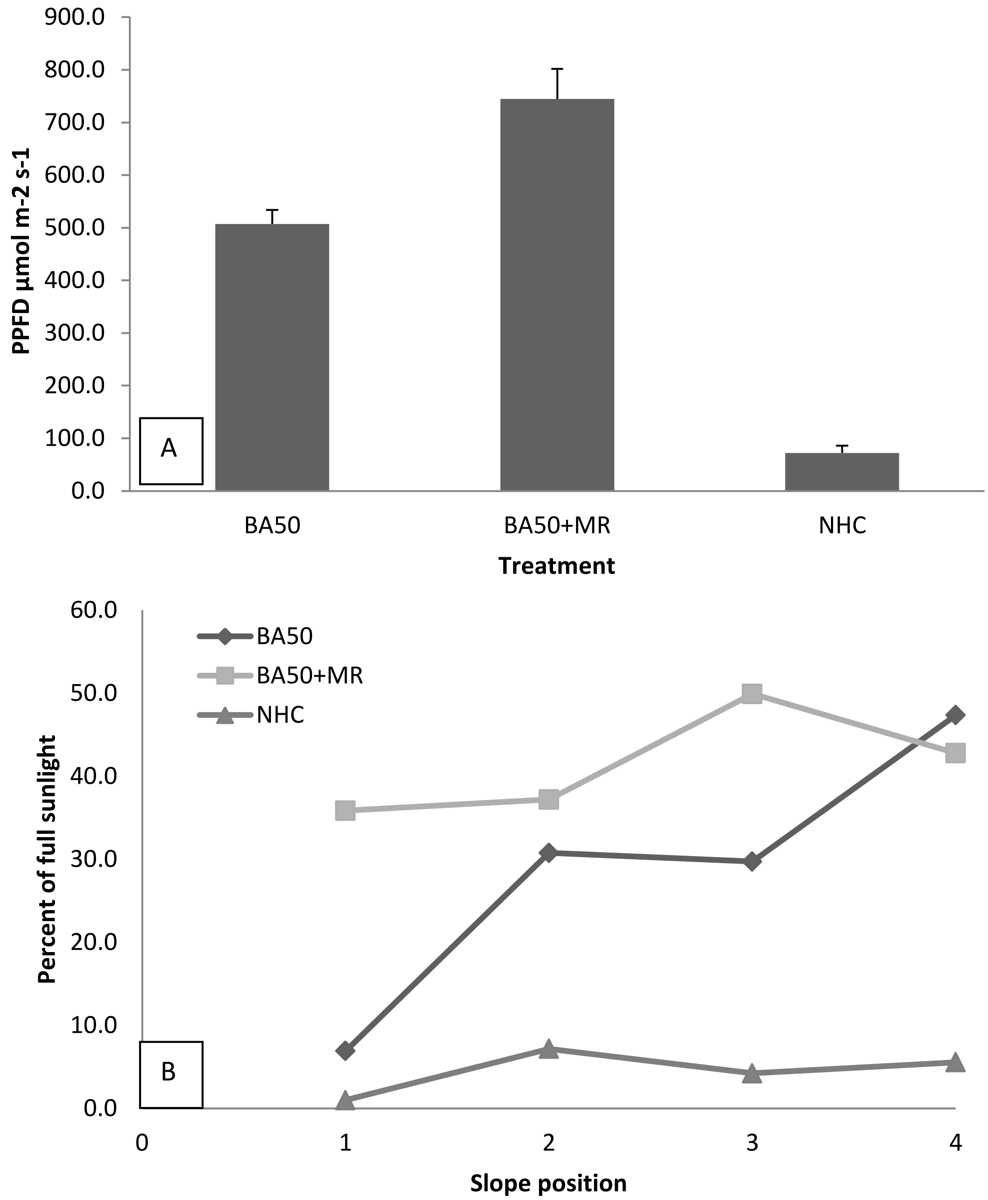

Figure 1--(A) Sunlight levels at ground level by treatment, and (B) sunlight levels at ground level by topographic position and treatment (means followed by same letter do not significantly differ at α = 0.05 using Student Newman-Kuels test

Table 1--Year 3 post-treatment versus initial oak seedlings per acre by treatment and height class. Numbers in brackets are ± SE; p values for paired t-tests at α = 0.05 are in parentheses

	-----------------BA50-----------------		------------BA50+MR------------		------------------NHC-----------------	
Height class	Initial	Final	Initial	Final	Initial	Final
	--*seedlings per acre*--					
1	292 [59]	1,354 [367] (0.01)	298 [98]	2,105 [724] (0.04)	367 [73]	3,109 [1,018] (0.07)
2	131 [22]	279 [68] (0.04)	254 [61]	746 [164] (0.03)	119 [29]	227 [48] (0.11)
3	27 [3]	65 [34] (0.007)	60 [23]	218 [57] (0.04)	6 [4]	4 [4] (0.82)

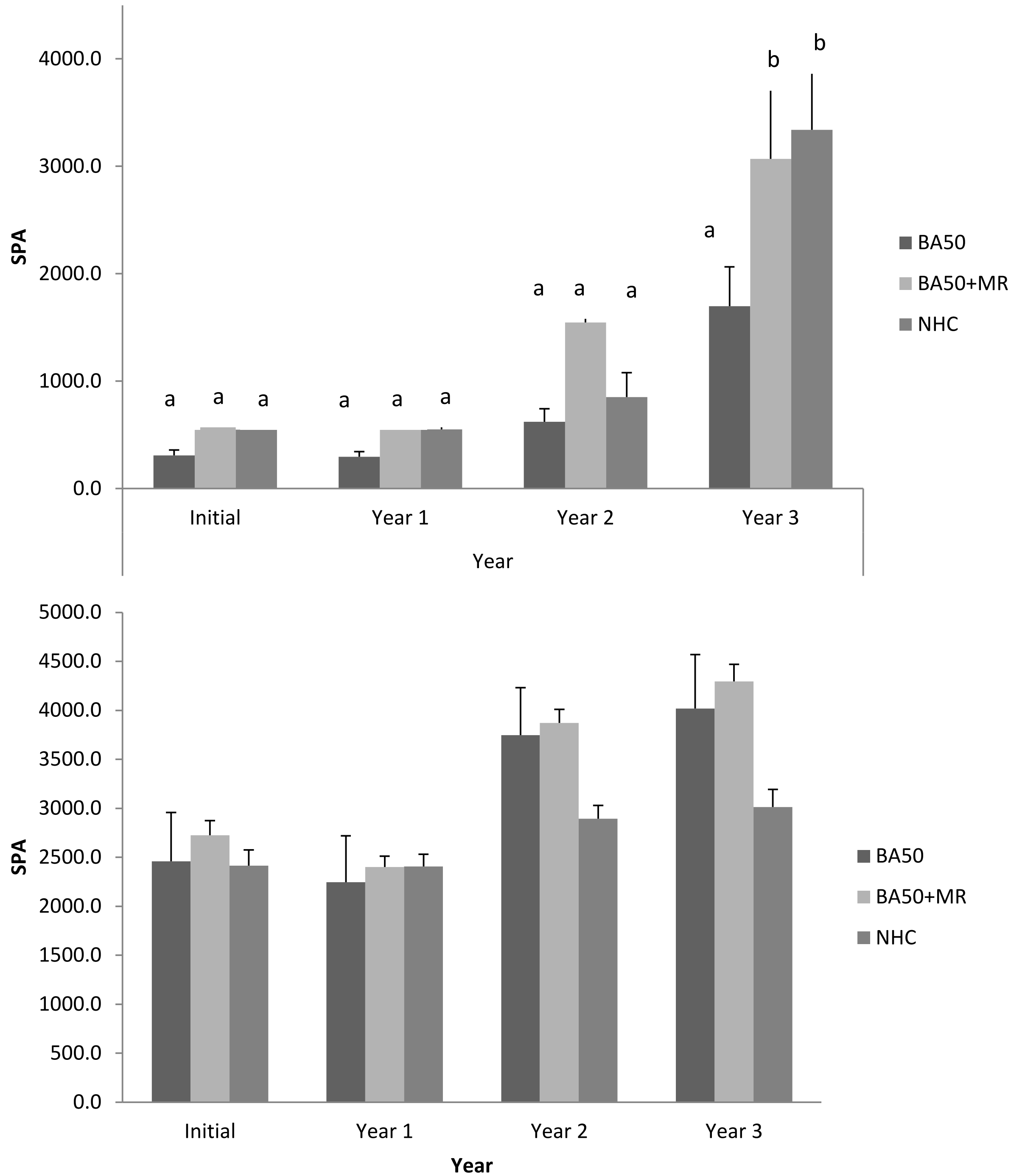

Figure 2--(A) Oak seedlings per acre by treatment by year, and (B) non-oak seedlings per acre by treatment by year (means followed by same letter do not significantly differ at α = 0.05 using Student Newman-Kuels test.

Year 3 mean oak SPA for NHC were 3,109; 227; and 4 for height classes 1, 2, and 3, respectively. This was a change of 2,742; 108; and -2 oak SPA for height class 1, 2, and 3 in the NHC treatment. There were no significant differences found between initial versus year 3 oak SPA by height class for the NHC treatment.

Year 3 post-harvest mean non-oak SPA were 4,018; 4,296; and 3,013 for BA50, BA50+MR, and NHC treatments, respectively (fig. 2b). This was an increase in mean non-oak SPA of 1,560; 1,570; and 599 for BA50, BA50+MR, and NHC treatments, respectively. Primary non-oak species were winged elm, red maple, black cherry *(Prunus serotina* Ehrh.*)*, blackgum *(Nyssa sylvatica* Marsh.*)*, hickory species, and flowering dogwood. Year 3 mean non-oak SPA for BA50 were 671; 1,256; and 2,091 for height classes 1, 2 and 3, respectively. Year 3 mean non-oak SPA for BA50+MR were 892; 1,556; and 1,848 for height classes 1, 2 and 3, respectively. Year 3, mean non-oak SPA for NHC were 928; 1,130; and 955 for height classes 1, 2, and 3, respectively.

DISCUSSION

Sunlight canopy penetration to the ground level was significantly impacted by treatment. BA50+MR provided 14 percent more sunlight than BA50 and 37 percent more sunlight than the NHC treatment. BA50+MR provided light levels in the optimal range for oak seedling establishment and growth. Non-oak midstory removal in BA50+MR helped establish uniform light conditions from lower slope to upper slope. BA50 provided adequate light levels for oak seedling development. However, there was a gradient in sunlight levels from lower slope to upper slope in BA50. This effect was attributed to harvest damage to midstory trees. This resulted in areas where light conditions ranged from optimal to adequate to inadequate for oak seedling development within BA50. Sunlight levels remained inadequate for oak seedling development across all slope positions in the undisturbed stand conditions present in the NHC treatment.

Oak reproduction was also significantly impacted by treatment applications. BA50+MR provided the greatest increase in oak reproduction abundance coupled with height growth. BA50+MR increased oak reproduction in height classes 2 and 3 by 193 and 263 percent. This increase was greater than BA50, which exhibited increases of 112 and 140 percent for height class 2 and 3 oaks. The NHC treatment did have a 90 percent increase in oak reproduction in height class 2 but exhibited a 33 percent decline in height class 3. Increases in height classes 2 and 3 are of great importance because these are the seedlings that have potential to become “free to grow” and contribute to future stand stocking through recruitment into the overstory.

The NHC treatment did provide a glimpse into seedling flux in undisturbed hardwood stands. Oak reproduction remained somewhat constant from initial through year 2 post-treatment abundance numbers. However, year 3 post-treatment abundance values spiked for NHC with a total of 3,109 oak SPA, which was an increase of 2,258 oak SPA over the previous season and the highest of any treatment in year 3. The increase was attributable to a heavy white oak acorn crop observed in fall 2011 (year 2). Conventional wisdom dictates that the majority of these seedlings will not persist into the future if stand conditions remain constant.

An additional question arose as to why the influx of new seedlings was not as substantial in harvested treatments versus the NHC. The author attributes this phenomenon to two factors: (1) environmental conditions and (2) competition from non-oak competitors. Year 3 was a season with extreme temperatures and severe drought in the region. With the capacity of upland sites in the Springfield Plateau region to become very dry, the harvest areas were exposed to higher sunlight levels and temperatures compared to the NHC treatment.

Also, while partial overstory harvests and midstory removal demonstrated improved sunlight conditions for oak seedling development, there remains significant competition from non-oak reproduction, suggesting additional competition control operations such as prescribed burning, could be a beneficial additional treatment to those presented in this study.

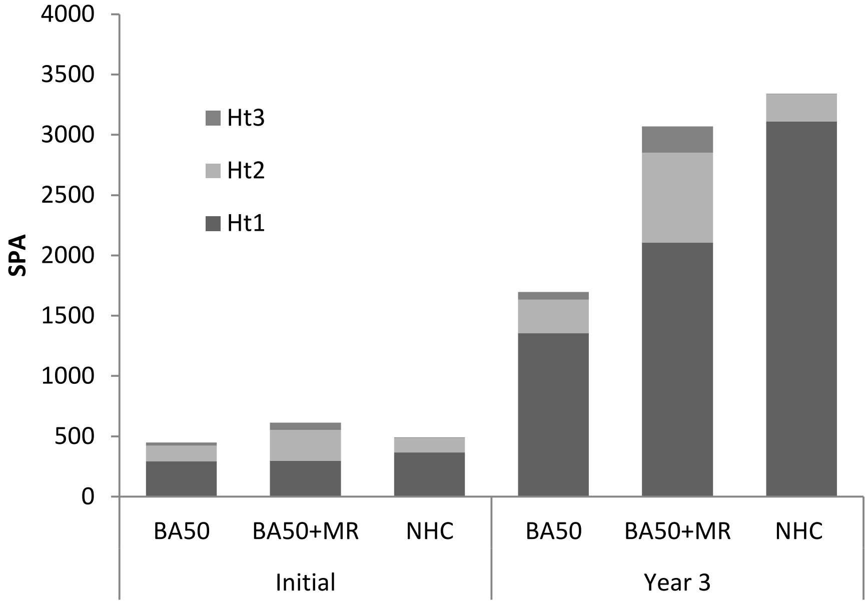

Figure 3--Oak seedlings per acre by height class and treatment for initial versus year 3 post-treatment.

CONCLUSIONS

Undisturbed stand conditions did not allow growth of oak reproduction into taller height classes, keeping them in a poor position to compete with other species. A shelterwood harvest alone (BA50) created variable sunlight conditions at ground level, with only a portion of its area exhibiting adequate sunlight for oak seedling growth and survival. Year 3 results show that a modified shelterwood, combining partial overstory and non-oak midstory removal, generated optimal sunlight conditions for oak seedling development. While the partial overstory removal coupled with midstory removal created conditions beneficial to oak seedling development, non-oak species in these environments remain strong competitors. Additional treatments, such as prescribed burning, could benefit oak reproduction.

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LITERATURE CITED

Battaglia, L.L.; Fore, S.A.; Sharitz, R.R. 2000. Seedling emergence, survival, and size in relation to light and water availability in two bottomland hardwood species. Journal of Ecology. 88: 1041-1050.

Brose, P.; 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.

Canham, C.D.; Denslow, J.S.; Platt, W.J. [and others]. 1990. Light regimes beneath closed canopies and tree fall gaps in temperate and tropical forests. Canadian Journal of Forest Research. 20: 620-631.

Cunningham, K.K.; Peairs, S.E.; Ezell, A.W. [and others]. 2011. Understory light conditions associated with partial overstory removal and mistory/understory control applications in a bottomland hardwood forest. Forests. 2: 984-992.

Ferguson, D.V.; Lowrance, J.S.; McFadden, C.E. 1982. Soil survey of Independence County, Arkansas. Washington, DC: U.S. Department of Agriculture Soil Conservation Service. 148 p.

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. U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 4 p.

Hicks, R.H.; Kennard, D.K.; Rauscher, H.M. [and others]. 2001. Silviculture and management strategies applicable to southern upland hardwoods. In: Proceedings, Southern

Forest Science Conference. [Place of publication unknown]: [Publisher unknown]. 8 p.

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.

Hodges, J.D.; Janzen, G.C. 1986. Studies on the biology of cherrybark oak: recommendations for regeneration. In: Phillips, D.R., ed. Proceedings of the 4th biennial southern silvicultural research conference. Gen. Tech. Rep. SE-42. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 133-138.

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.

Quero, J.L.; Villar, R.; Maranon, T. [and others]. 2008. Relating leaf photosynthetic rate to whole-plant growth: drought and shade effects on seedlings of four *Quercus* species. Functional Plant Biology. 35: 725-737.

Rogers, R., Johnson, P.S., Loftis, D.L., 1993. An overview of oak silviculture in the United States: the past, present, and future. Annals of Forest Science. 50: 535-542.

Smith, D.M.; Larson, B.C.; Kelty, M.J.; Ashton, P.M.S. 1996. The practice of silviculture: applied forest ecology. 9th ed. New York: John Wiley & Sons. 537 p.

AFFORESTING AGRICULTURAL LANDS IN THE MISSISSIPPI ALLUVIAL VALLEY (USA): EFFECTS OF SILVICULTURAL METHODS ON UNDERSTORY PLANT DIVERSITY

Diane De Steven, Callie J. Schweitzer, Steven C. Hughes, and John A. Stanturf[1]

Abstract--To compare methods for bottomland hardwood reforestation on marginal farmlands in the Mississippi Alluvial Valley, four afforestation treatments (natural colonization, sown oak acorns, planted oak seedlings, cottonwood–oak interplant) were established in 1995 on former soybean cropland. Natural, sown, and planted-oak plots were not managed after establishment. Interplant plots received intensive management including two seasons of weed-control disking between planted cottonwoods, after which oaks were interplanted. Previous work found that forest canopy development was accelerated by interplanting; however, the best methods for establishing trees could have different effects on forest community diversity. Multi-year data on understory plant composition were analyzed to determine if less intensive methods promoted greater diversity. Ground-layer vegetation was sampled annually from 1996 to 1998, and again in 2006. Only total biomass was affected by afforestation technique, with the greatest declines in the interplant treatment. Changes in all species composition measures were a function of successional time. Although diversity did not vary substantially with reforestation method, lack of hydrologic restoration favored an understory flora more typical of moist old-fields than natural floodplain forests.

INTRODUCTION

Bottomland hardwood forests once covered 10 million ha in the Lower Mississippi River Alluvial Valley (LMAV). By the 1980s, large flood-control projects and land clearing for agriculture had reduced forest extent by roughly 75 percent (Haynes 2004). This historic forest loss has been addressed by various reforestation efforts in the past few decades (King and Keeland 1999, King and others 2006, Schoenholtz and others 2001). Reforestation methods (termed afforestation when converting from agriculture or other non-forest land use) have varied from completely passive to intensive, depending on land ownership and management objectives. Active, low-cost methods were favored as a way to establish desired species (mainly 'hard-mast' oaks, *Quercus* spp.) and overcome the dispersal limitations of passive colonization. However, the mixed results from active low-intensity techniques raised questions about potential tradeoffs between satisfying habitat/diversity objectives versus enhancing economic returns (such as timber yield) on private lands (Haynes 2004, Stanturf and others 2001, Twedt and Wilson 2002).

A long-term experiment was established in 1995 to compare four afforestation methods ranging from passive to intensive, so that ecological and economic trade-offs could be assessed at operational scales (Gardiner and others 2008, Schweitzer and Stanturf 1999). The methods were natural tree colonization, establishing oak species by direct-seeding or by planting, and interplanting oak seedlings with a fast-growing early-succession tree species. A specific goal was to evaluate if the interplant method could accelerate forest development for both timber and habitat values. Results from this experiment and analogous studies indicated that interplanting favors rapid development of tree height, vertical structure and canopy closure, whereas less intensive methods may allow for greater tree diversity (Stanturf and others 2009, Twedt 2004, Twedt and Wilson 2002).

Most of the afforestation research has focused on forest structure or overstory tree diversity. Understory plant composition is an aspect that has been evaluated only infrequently. Ground-layer biomass was assessed in the long-term experiment as a source of competition for planted tree seedlings (Stanturf and others 2009), but this layer is also a component of community diversity and contributes to habitat values. Intensive afforestation methods, while favorable for tree development, could have negative effects on ground-layer plant diversity. In this paper, we analyze data from the long-term experiment to determine if alternative methods led to differences in understory plant composition.

STUDY SITE AND METHODS

The long-term experimental site is located in Sharkey County, MS, on a tract that was in soybean cultivation until fall 1994. The soils are

[1]Research Ecologist, USDA Forest Service, Southern Research Station, Stoneville, MS 38776; Research Forester, USDA Forest Service, Southern Research Station, Normal, AL 35762; Biological Science Technician, USDA Forest Service, Southern Research Station, Stoneville, MS 38776; and Senior Scientist, USDA Forest Service, Southern Research Station, Athens, GA 30602.

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mapped as shrink-swell clays (Vertisols) in the Sharkey series (Pettry and Switzer 1996). The area is in the Big Sunflower River drainage, part of the Yazoo River basin of the LMAV. Portions of the site may receive dormant-season backflooding in some years (Stanturf and others 2009), but generally the area is isolated from natural flooding of the Yazoo and Mississippi Rivers by an extensive system of flood-control levees and ditch-channels (cf. Faulkner and others 2011, Frederickson 2005).

The basic experimental design is summarized here; see Schweitzer and Stanturf (1999) and Stanturf and others (2009) for detailed descriptions. The experiment was a randomized complete blocks design, with three blocks of four treatment plots representing a gradient of silvicultural intensity: (1) natural tree colonization; (2) sown Nuttall oak acorns (*Q. texana* Buckl.); (3) planted Nuttall oak seedlings; and (4) phased interplanting of cottonwood (*Populus deltoides* W. Bartram ex. Marsh.) and Nuttall oak. Treatment plots were 8.1 ha (20 acres) in size. All plots were prepared by disking prior to establishment. Natural colonization plots had no other manipulation. Direct-seed and oak-seedling plots were planted during March through May 1995 and then received no further management. Acorns were sown at 1.1- by 3.7-m spacing (2,457 acorns/ha), and seedlings were planted at 3.7- by 3.7-m spacing (730 seedling/ha). Also in March 1995, cottonwood cuttings were planted in the interplant plots at 3.7- by 3.7-m spacing. These plots were treated with fertilizer, herbicides and pesticides. Additionally, sub-plot sections received either one or two seasons of weed-control disking between cottonwood rows before oak seedlings were interplanted in March 1997 at 3.7- by 7.4-m spacing (365 seedlings/ha). Cottonwood thinnings and harvest were scheduled for the year 2007 to study yields and eventual success of oak release.

For purposes of this study, the ‘understory’ layer was considered to be all ground-level vegetation (excluding tree seedlings) between planted or volunteer trees. Understory sampling occurred in years 2 through 4 after plot establishment (1996-1998) and again in year 12 (2006). Eight stratified-random 1-m^2 quadrats were sampled per treatment plot (interplant plots were sampled with eight quadrats per one-/two-season disked sub-plots in years 2 through 4 only). In late August-September of each year, all ground-layer vegetation (herbs, shrubs, woody vines) in each quadrat was clipped to ground level, sorted to species, and dried at 40 °C (104 °F) to obtain dry-weight biomass (g/m^2) as a metric of species abundance. In the analyses, we used the quadrat data from areas disked for two seasons (1995, 1996) to represent the ‘intensive’ interplant treatment; this equalized sample area to eight quadrats per treatment plot and provided a balanced statistical design for all years. Diagnostic tests verified that the plant data from the interplant plots did not differ between the one- and two-season disked areas for any sampling year.

Plot-level data were analyzed in a repeated-measures ANOVA model with afforestation treatment as the between-blocks factor and year as the within-blocks repeated measure (total n = 48). Analyses were performed in SYSTAT® (SPSS, Inc.). The F-test results were comparable to Greenhouse-Geisser and Huynh-Feldt adjusted *p*-values, indicating that model assumptions were appropriate (Wilkinson and Coward 1999). Variables were total biomass (g/m^2, averaged over 8 quadrats per plot), species richness per plot, and species composition (numbers and relative percentage biomass) in terms of growth form, wetland indicator class, and nativity. Growth forms were classed as herbaceous broad-leaved (forbs), herbaceous graminoid (grasses/sedges), or woody (shrubs/vines). Species indicator classes were defined from five categories (Reed 1997) as either wetland (OBL, FACW categories), facultative (FAC), or upland (FACU, UPL). Native or non-native (introduced) status was obtained from the U.S. Department of Agriculture PLANTS database (http://plants.usda.gov). Excepting total biomass, no variable had significant treatment effects; therefore, only the year and year-by-treatment tests are reported. Changes in presence and abundance between years 2 and 4 and year 12 were summarized for the 30 prevalent species that comprised > 95 percent of total biomass.

RESULTS

Total biomass (fig. 1) was the only variable that differed among afforestation treatments (F = 30.1; df = 3, 6; P = 0.001); weaker year and year-by-treatment effects (P = 0.04) were an artifact of low biomass in the 1996 natural treatment. Understory biomass was highest in the natural and sown treatments and lowest in the interplant treatment.

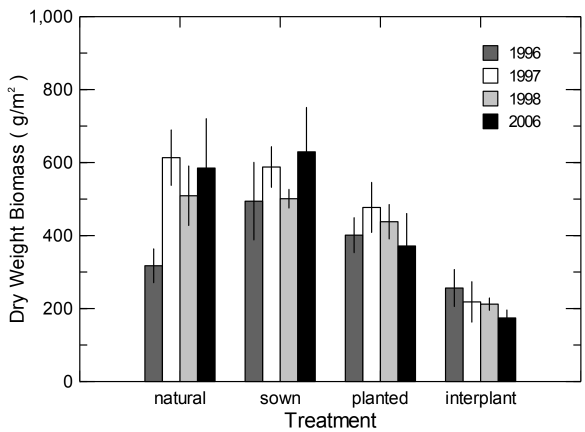

Figure 1 – Changes in ground-layer biomass among afforestation treatments and years. Data are means ± s.e. for n = 3 replicate blocks.

Species richness and all compositional variables differed among years, reflecting successional change over the 12-year period (tables 1 and 2). There were essentially no year-by-treatment interactions. Species richness per plot declined with time, mainly owing to losses of forb species (table 1). Number of herbaceous species (forbs and graminoids) collectively decreased from 13 to 8 species, while the number of woody species increased slightly. Relative biomass of herbaceous plants decreased from 84 to 33 percent, with woody plants increasing to 67 percent of total biomass by year 12. Non-native species were negligible by year 12 (table 1).

With respect to wetland indicator class (table 2), the number of upland species decreased over time, while the number of wetland species fluctuated slightly. Relative biomass of both upland and wetland species declined as the relative biomass of facultative species increased. The net result was that ‘hydrophytic’ species (wetland plus facultative) comprised nearly 75 percent of total biomass by year 12, mainly owing to facultative species.

The temporal trends reflected changes in particular species and species-groups (table 3). Many forb species were no longer detected after 12 years, particularly asters (*Symphyotrichum* spp.) and various upland annuals. Forb biomass became dominated by perennial clonal goldenrods (*Solidago* spp.) and marsh elder (*Iva annua* L.), a robust annual. Graminoid biomass shifted from the highly dominant Johnson grass [*Sorghum halepense* (L.) Pers.], an introduced upland species, to native broomsedge grass (*Andropogon virginicus* L.) and sedges/rushes (*Carex* spp., *Juncus* spp.). The increasing woody biomass became dominated by facultative vines such as trumpet creeper [*Campsis radicans* (L.) Seem. ex Bureau] and poison ivy [*Toxicodendron radicans* (L.) Kuntze], and by blackberry shrubs (*Rubus* spp.). Of 77 identified taxa found over all years combined (data not shown), only 16 of 62 herbaceous species were present by the final year, whereas woody species had increased from only 3 in year 2 to 13 by year 12.

DISCUSSION

Effects of Afforestation Method

The main response to the afforestation treatments was decreased ground-layer biomass with greater silvicultural intensity. Unsurprisingly, this pattern was inverse to the gradient of canopy development. Average tree density after 7 years was lowest in natural and sown plots, intermediate in oak-planted plots, and highest in interplant plots (Hamel 2003). Tree heights after 3 years averaged < 2 m for recolonizing trees and sown/planted oaks versus 8 m for the cottonwoods; this height difference widened to 3 to 4 m versus > 14 m by year 7 (Hamel 2003, Stanturf and others 2009). The much lower ground-layer biomass in the interplant plots (fig. 1) also reflected an effect of rapid cottonwood height-growth, which allowed woody vines to climb vertically and thus displaced some vine biomass to the canopy layer (Personal observation. 2013. S.C. Hughes, Biological Science Technician, USDA Forest Service, Southern Research Station, Stoneville, MS 38776). Twedt and Wilson (2002) also found an inverse pattern attributable to differences in

Table 1--Temporal change in total species richness, and in numbers and relative biomass (percent) of species by non-native status and by growth form. Data for each year are per-plot means (s.e.) over blocks and treatments (n = 12)

Variable	1996	1997	1998	2006	Year effect[a]	Year-by-treatment interaction[a]
			number of species			
Species richness	16 (1)	16 (1)	12 (1)	14 (1)	*	ns
Non-native species	4 (0.4)	2 (0.2)	2 (0.4)	0.3 (0.1)	**	ns
Herbaceous forbs	10 (1)	10 (1)	7 (1)	4 (0.4)	**	ns
Herbaceous graminoids	3 (0.5)	4 (0.4)	3 (0.5)	4 (0.4)	n.s.	ns
Shrubs/woody vines	2 (0.1)	2 (0.1)	2 (0.2)	5 (0.4)	**	ns
			percent of total biomass			
Non-native biomass	47 (6)	37 (8)	21 (6)	0.1 (0.1)	**	ns
Forb biomass	40 (6)	46 (8)	63 (7)	29 (5)	**	*
Graminoid biomass	44 (6)	40 (7)	23 (5)	4 (1)	**	ns
Shrub/woody vine biomass	15 (3)	13 (4)	14 (5)	67 (6)	**	ns

[a]ANOVA F-test significance for year (df = 3, 24) and interaction (df = 9, 24) is noted as *P < 0.05, **P < 0.01, ns = not significant.

Table 2--Temporal change in numbers and relative biomass (percent) of species by wetland indicator class. Data and ANOVA tests as in table 1

Variable	1996	1997	1998	2006	Year effect[a]	Year-by-treatment interaction[a]
			number of species			
Wetland species	4 (0.3)	5 (1)	3 (0.5)	4 (0.3)	*	ns
Facultative species	7 (1)	7 (0.5)	6 (1)	6 (0.5)	ns	ns
Upland species	4 (0.3)	3 (0.4)	2 (0.4)	2 (0.2)	**	ns
			percent of total biomass			
Wetland species	13 (3)	27 (5)	47 (6)	7 (1)	**	ns
Facultative species	35 (7)	35 (5)	29 (5)	67 (4)	**	ns
Upland species	52 (8)	38 (7)	24 (5)	26 (4)	**	ns

[a]ANOVA F-test significance for year (df = 3, 24) and interaction (df = 9, 24) is noted as *P < 0.05, **P < 0.01, ns = not significant.

canopy development on reforested sites across the LMAV, with greater tree heights and lower ground-layer cover on tracts planted in oak seedlings compared to direct-seeded tracts.

Despite the contrast in forest structure between interplant plots and other treatments, afforestation method had no notable effects on understory species composition. One likely reason is that cottonwood canopies are not dense, thus light penetration to ground level (Gardiner and others 2001) would allow species of more open treatments to persist in the interplant plots as well. Possibly the species composition of interplant plots differed subtly in the first sampling year (year 2), when quadrat placement in that treatment was adjusted (as needed) to sample vegetated spots adjacent to bare soil patches that had not yet regrown from after disking (cf. Methods). However, species composition did not differ among treatments in years 3 and 4, suggesting that any subtle effects of the early disking were ephemeral.

Successional Change

Biomass as an abundance metric may overweight the contribution of woody species to total plant coverage, but the data were representative of species trends over time. The changes in plant composition paralleled the typical pattern of succession on abandoned fields and afforested tracts across the LMAV

Table 3--Change in mean dry-weight biomass of abundant herb-layer species from years 2 through 4 (1996–1998, averaged) to year 12 (2006). Species are grouped by growth form and wetland indicator class. Species nomenclature follows the USDA PLANTS database (http://plants.usda.gov)

Species	Indicator class	Life form[b]	1996–1998	2006
			------------ *g/m*^{2} ------------	
Herbaceous broad-leaved				
Ipomoea wrightii[a]	wetland	vine (A)	2.1	0
Lythrum alatum	wetland	forb (P)	5.6	5.8
Sesbania sp. (*herbacea/exaltata*)	wetland	forb (A)	2.0	1.6
Symphyotrichum divaricatum	wetland	forb (A)	6.9	0
Symphyotrichum lanceolatum	wetland	forb (P)	3.0	0
Symphyotrichum spp. (*dumosum, ontarionis, pilosum*)	facultative	forb (P)	51.2	0
Ambrosia trifida	facultative	forb (A)	4.7	0
Iva annua	facultative	forb (A)	9.1	17.4
Desmanthus illinoensis	facultative	forb (P)	15.4	0
Eupatorium serotinum	facultative	forb (P)	2.2	1.4
Ambrosia artemesiifolia	upland	forb (A)	2.0	0
Chamaesyce spp. (*hyssopifolia, nutans*)	upland	forb (A)	2.2	0
Rudbeckia hirta	upland	forb (A)	9.0	0
Sida spinosa	upland	forb (A)	8.1	0
Triodanis biflora	upland	forb (A)	7.0	0
Solidago altissima/gigantea	upland	forb (P)	88.5	72.5
Herbaceous graminoid				
Carex frankii	wetland	sedge (A)	3.3	3.6
Juncus spp. (*dichotomus, diffisissimus*, others)	wetland	rush (P)	1.2	3.8
Andropogon virginicus	facultative	grass (P)	2.4	6.9
Paspalum dilatatum[a]	facultative	grass (P)	6.6	0
Setaria spp. (*geniculata, glauca*[a])	facultative	grass (P/A)	5.1	0
Sorghum halepense[a]	upland	grass (P)	125.5	0.1
Woody shrub/vine				
Brunnichia ovata	wetland	vine	13.5	15.6
Ampelopsis arborea	facultative	vine	0	3.9
Campsis radicans	facultative	vine	41.0	173.2
Toxicodendron radicans	facultative	vine	0.1	106.2
Rubus trivialis	facultative	shrub	0.9	2.5
Rubus spp. (*argutus*)	upland	shrub	0.4	25.2

[a]Non-native.
[b] (A) = annual, (P) = perennial.

(Battaglia and others 2002, Twedt 2004). Early-succession herb species, particularly annuals and non-native agricultural weeds, were excluded by expansion of perennial and woody species. Only 15 of the 30 most prevalent species were detected after 12 years, with dominance shifting to woody vines, blackberry shrubs, and old-field herbs such as goldenrod and broomsedge.

The vegetation became more hydrophytic, mainly owing to greater importance of facultative woody vines. These vines are common species in native floodplain forests (Sharitz and Mitsch 1993); many are animal-dispersed and thus could readily colonize afforested sites. In contrast, wetland species were under-represented. The understory of experimental plots lacked many typical herbs of floodplain forests, such as wetland sedges (*Carex lurida* Wahlenb., *C. louisianica* L.H. Bailey), cutgrasses (*Leersia* spp.), panic-grasses [*Phanopyrum gymnocarpon* (Elliott) Nash, *Panicum rigidulum* Bosc ex Nees), false-nettle [*Boehmeria cylindrical* (L.) Sw.], lizard's tail (*Saururus cernuus* L.), water-willows (*Justicia* spp.), and bugle-weeds (*Lycopus* spp.) (Sharitz and Mitsch 1993, Wharton and others 1982). Few of these species are animal-dispersed, and many may depend on floodwaters for their distribution.

They may also be less likely to persist in the seedbank following intensive farming (Middleton 2003).

Implications

Ability of floodplain plants to colonize reforestation sites in the LMAV will be limited by relative isolation from river flooding and from remnant native forests. Local habitat conditions also constrain understory plant composition. Intermittent backwater flooding at the study site is largely rainfall-driven and ephemeral. Because the experiment was designed partly to assess afforestation options for timber production, there was no hydrology manipulation to enhance local wetness conditions for wetland species. In contrast, some LMAV tracts that are afforested under federal conservation programs often include attempts to enhance local site hydrology. Typically, ditches are blocked to create shallow-water and managed moist-soil areas; occasionally, flood-control levees may be breached to allow local flooding from adjacent streams (Hunter and others 2008, King and Keeland 1999, King and others 2006). Such practices can promote greater abundance of wetland plants, but they do not restore the original river flooding regime (see Lockaby and Stanturf 2002).

Active reforestation efforts can be successful in providing forest habitat, productivity, and carbon-sequestration functions (Faulkner and others 2011, Hamel 2003, Haynes 2004). Given the constraints on restoring natural river hydrology in the LMAV, a continuing challenge is that these efforts will not necessarily replicate the plant diversity of native bottomland forests. Understanding the potential trade-offs of alternative methods can assist in selecting restoration options that meet a variety of ecological and landowner objectives.

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LITERATURE CITED

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: 1-17.

Faulkner, S.; Barrow, W., Jr.; Keeland, B. [and others]. 2011. Effects of conservation practices on wetland ecosystems services in the Mississippi Alluvial Valley. Ecological Applications. 21(3) Supplement: S31-S48.

Frederickson, L.H. 2005. Contemporary bottomland hardwood systems: structure, function, and hydrologic conditions resulting from two centuries of anthropogenic activities. In: Frederickson, L.H.; King, S.L.; Kaminski, R.M., eds. Ecology and management of bottomland hardwood systems: the state of our understanding. Gaylord Memorial Laboratory Special Publ. No. 10. Puxico, MO: University of Missouri-Columbia: 19-35.

Gardiner, E.; Schweitzer, C.J.; Stanturf, J.A. 2001. Photosynthesis of Nuttall oak (*Quercus nuttallii* Palm.) seedlings interplanted beneath an eastern cottonwood (*Populus deltoides* Bartr. ex Marsh.) nurse crop. Forest Ecology and Management. 149: 283-294.

Gardiner, E.; Stanturf, J.; Leininger, T. [and others]. 2008. Establishing a research and demonstration area initiated by managers: the Sharkey Restoration Research and Demonstration Site. Journal of Forestry. 106: 363-369.

Hamel, P.B. 2003. Winter bird community differences among methods of bottomland hardwood forest restoration: results after seven growing seasons. Forestry. 76: 189-197.

Haynes, R.J. 2004. The development of bottomland forest restoration in the Lower Mississippi River Alluvial Valley. Ecological Restoration. 22: 170-182.

Hunter, R.G.; Faulkner, S.P.; Gibson, K.A. 2008. The importance of hydrology in restoration of bottomland hardwood wetland functions. Wetlands. 38: 605-615.

King, S.L.; Keeland, B.D. 1999. A survey and evaluation of reforestation of the Lower Mississippi River Alluvial Valley. Restoration Ecology. 7: 348-359.

King, S.L.; Twedt, D.J.; Wilson, R.R. 2006. The role of the Wetlands Reserve Program in conservation efforts in the Mississippi Alluvial Valley. Wildlife Society Bulletin. 34: 914-920.

Lockaby, G.; Stanturf, J. 2002. Potential effects of restoration on biogeochemical functions of bottomland hardwood ecosystems. In: Holland, M.; Warren, M.; Stanturf, J.A., eds. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? Gen. Tech. Rep. SRS-50. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 116-119.

Middleton, B.A. 2003. Soil seed banks and the potential restoration of forested wetlands after farming. Journal of Applied Ecology. 40: 1025-1034.

Pettry, D.E.; Switzer, R.E. 1996. Sharkey soils in Mississippi. Bull. 1057. Mississippi State, MS: Agriculture and Forest Experiment Station. 37 p.

Reed, P.B., Jr. 1997. Revision of the national list of plant species that occur in wetlands. Washington, DC: U.S Department of Interior Fish and Wildlife Service. 209 p.

Schoenholtz, S.H.; James, J.P.; Kaminski, R.K. [and others]. 2001. Afforestation of bottomland hardwoods in the Lower Mississippi Valley: status and trends. Wetlands. 21: 602-613.

Schweitzer, C.J.; Stanturf, J.A. 1999. A comparison of large-scale reforestation techniques commonly used on abandoned fields in the Lower Mississippi Alluvial Valley. In: Haywood, J.D., ed. Proceedings of the 10th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-30. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 136-141.

Sharitz, R.R.; Mitsch, W.J. 1993. Southern floodplain forests. In: Martin, W.H.; Boyce, S.G.; Echternacht, A.C., eds. Biodiversity of the southeastern United States: lowland terrestrial communities. New York: John Wiley & Sons: 311-372.

Stanturf, J.A.; Gardiner, E.S.; Shepard, J.P. [and others]. 2009. Restoration of bottomland hardwood forests across a treatment intensity gradient. Forest Ecology and Management. 257: 1803-1814.

Stanturf, J.A.; Schoenholtz, S.H.; Schweitzer, C.J. [and others]. 2001. Achieving restoration success: myths in bottomland hardwood forests. Restoration Ecology. 9: 180-200.

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: 287-298.

Wharton, C.H.; Kitchens, W.M.; Sipe, T.W. 1982. The ecology of bottomland hardwood swamps of the Southeast: a community profile. Rep. FWS/OBS-81/37. Washington, DC: U. S. Department of Interior Fish and Wildlife Service. 134 p.

Wilkinson, L.; Coward, M. 1999. Linear models. Chapter 13. In: SYSTAT 9 Statistics I. Chicago: SPSS, Inc.: 365-398.

EFFECTS OF SEASON OF CUT AND RESIDUAL OVERSTORY DENSITY ON STUMP SPROUT GROWTH AND DEVELOPMENT

Tara L. Keyser[1]

Stump sprouts contribute to the regeneration potential of upland hardwood forests in the southern Appalachian Mountains (Cook and others 1988); however, most of the information regarding stump sprout potential and subsequent sprout growth and development is from studies following regeneration cuts. The effects of less-than-stand-replacing disturbances (e.g. thinnings) on stump sprout potential is lacking. In addition, the season during which trees are harvested has been shown, in some instances, to affect the ability of certain upland hardwood tree species to produce stump sprouts (Buell 1940, Harrington 1984). Lack of data related to the growth and development of stump sprouts following a range of canopy-reducing disturbances and timing of harvests limits our ability to predict the regeneration response of a stand subjected to alternative management scenarios. The objective of this study was to examine differences in stump sprout potential in upland hardwood forests in the southern Appalachian Mountains across a range of canopy-reducing disturbances. Preliminary (2-year) results related to: (1) the effects of season of cut on the probability of sprouting of upland hardwood tree species, and (2) the effects of residual overstory density and season of cut on subsequent stump sprout growth and development are presented.

In 2009, 24 plots 0.1-ha in size were established in upland, mixed-hardwood forest types on Bent Creek Experimental Forest in Asheville, NC. On 12 plots, basal area was mechanically reduced from below by 10, 20, 30, or 40 percent between January and February 2009 (dormant season). Basal area was reduced from below on another 12 plots by 10, 20, 30, or 40 percent between July and August 2010 (growing season). Each season of cut and level of basal area reduction combination were replicated three times (n = 3). On each cut stump, with the number of cut stumps varying with the level of basal area reduction, the following data were recorded: (1) sprouting (yes/no); (2) height of the dominant sprout in each clump (m); and (3) maximum length (m) and width (m) of each individual sprout clump.

Logistic regression (PROC GLIMMIX) was utilized to examine the probability of sprouting for each species as a function of: (a) parent tree diameter at breast height (d.b.h.; cm); (b) season of cut (dormant vs. growing); and (c) the interaction between parent tree d.b.h. and season of cut. The hierarchical nature of the data (i.e., cut stumps nested within plots) was accounted for by including plot as a random effect in the probability of sprouting model. A two-factor analysis of variance (ANOVA) was used to examine how season of cut and basal area reduction as well as the interaction between basal area reduction and season of cut affected sprout growth after two growing seasons post-treatment, with dependent variables being: (a) height (m) of dominant sprout; and (b) area (m^2) of sprout clump calculated from maximum length and width data, with area modeled as an ellipse.

In total, 1,369 individuals were felled and the associated cut stumps were analyzed. The sample size of nine species, including red maple (*Acer rubrum* L.), sweet birch (*Betula lenta* L.), hickory (*Carya* spp.), flowering dogwood (*Cornus florida* L.), yellow-poplar, (*Liriodendron tulipifera* L.), blackgum (*Nyssa sylvatica* Marsh.), sourwood (*Oxydendrum arboretum* DC), white oak (*Quercus alba* L.), and chestnut oak (*Quercus prinus* L.), was sufficient to analyze the effects of season of cut and parent tree d.b.h. on the probability of stump sprouting by species. Only for sweet birch was the probability of stump sprouting affected by the season of cut, with trees cut during the growing season less likely to sprout (54 percent sprout rate) than trees cut in the dormant season (93 percent sprout rate). The relationship between the probability of stump sprouting and parent

[1]Research Forester, USDA Forest Service, Southern Research Station, Asheville, NC, 28806.

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Table 1—Mean height (m) of the dominant sprout in each clump (averaged across species) and mean area (m^2) of individual sprout clumps (averaged across species) in each basal area reduction treatment

Basal area reduction	Growing season height (*m*)		Growing season area (m^2)		Dormant season height (*m*)		Dormant season area (m^2)	
Percent	Mean	SE[a]	Mean	SE	Mean	SE	Mean	SE
10	1.2	0.1	1.3	0.2	1.2	0.2	1.7	0.5
20	1.6	0.3	1.7	0.5	1.6	0.2	2.4	0.1
30	1.9	0.1	2.4	0.4	2.0	0.1	2.9	0.1
40	2.1	0.1	3.0	0.1	2.0	0.1	3.4	0.5

[a]SE = standard error

tree d.b.h. was not statistically significant (P > 0.05) for sweet birch (sprout rates presented above), red maple (95 percent sprout rate), dogwood, (88 percent sprout rate), sourwood (98 percent sprout rate), hickory species (77 percent sprout rate), chestnut oak (86 percent sprout rate), and yellow-poplar (91 percent sprout rate). For blackgum and white oak, a negative relationship between parent tree d.b.h. and the probability of stump sprouting (P < 0.05) was observed (fig. 1).

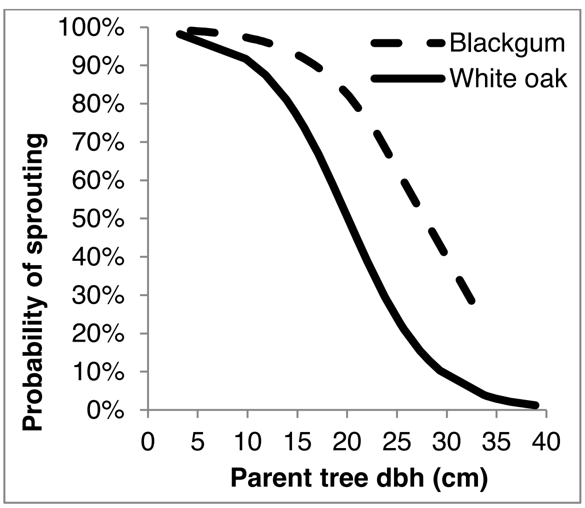

Figure 1--Probability of sprouting as a function of parent tree d.b.h. (cm) for white oak and blackgum.

Averaged across species, the height of the dominant sprout was positively affected by the percent of basal area removed (P < 0.05) (table 1) but was not influenced by the season of cut or the interaction between season of cut and basal area reduction (P > 0.05). Regardless of season of cut, height of the dominant sprout was similar between the 10 and 20 percent reductions in basal area and the 30 and 40 percent reductions in basal area. The area of individual sprout clumps was affected by the percent of basal area removed ($F_{3,16}$ = 10.2, P = 0.0005) as well as the season of cut ($F_{1,16}$ = 4.9, P = 0.0423) but not the interaction between season of cut and basal area removed (P > 0.05). Averaged across the levels of basal area reduction, sprout clumps from trees cut in the dormant season were slightly larger in area than those cut in the growing season averaging 2.6 m^2 (± 0.2) and 2.1 (± 0.2) m^2, respectively. The relationship between the area of individual sprout clumps and level of basal area reduction was similar to that of dominant sprout height (table 1). As a percent of the 0.1-ha plot occupied by sprouts, it is apparent that as overstory density is reduced, growing space is quickly occupied by sprouting species of which most were red maple and sourwood. Expressed as a percentage of total plot area, sprout clumps occupied between 0.6 and 35 percent of the 0.1-ha plots.

Although thinnings are not designed with regeneration in mind, they can affect the regeneration potential of a stand by altering the size distribution of the advance reproduction layer as well as initiate the production and growth of stump sprouts. The rapid growth of stump sprouts, if not controlled though further intermediate treatments, can alter the regeneration potential of a stand. The increase in growing space and light conditions that may facilitate the growth and development of a diverse advance reproduction layer, therefore, may be short-lived, which may limit the ability of the advance regeneration layer to contribute to species composition after a final regeneration harvest.

LITERATURE CITED

Buell, J.H. 1940. Effect of season of cutting on sprouting dogwood. Journal of Forestry. 38: 649-650.

Cook, J.E.; Sharik, T.L.; Smith, D.W. 1998. Oak regeneration in the southern Appalachians: potential, problems, and possible solutions. Southern Journal of Applied Forestry. 22: 11-18.

Harrington, C.A. 1984. Factors influencing initial sprouting of red alder. Canadian Journal of Forest Research. 14: 357-361.

EFFECTS OF DIFFERENT MANAGEMENT REGIMES ON SURVIVAL OF NORTHERN RED OAK UNDERPLANTINGS IN THE RIDGE AND VALLEY PROVINCE

Adam E. Regula, David W. McGill and Cynthia D. Huebner[1]

Abstract--While dominant throughout much of the eastern United States, a recent decline in oak regeneration has merited substantial research. Ultimately, successful regeneration entails the establishment of advance reproduction of sufficient size and density to provide a high probability of ascendancy to dominant or co-dominant status. Potential prescriptions for achieving this include manipulation of light infiltration and control of competing vegetation through shelterwood harvests and prescribed burning. Diameter-limit cutting is a method used on private forests which creates diverse post-harvest conditions which can favor fast-growing, shade-intolerant competition or shade-tolerant species depending on initial stand structure and diameters harvested. This study examines the effect of five management regimes on northern red oak (*Quercus rubra* L.) underplantings through a 2-year assessment of 1+0 bareroot seedlings. Treatments consist of: (1) control sites with no disturbance for at least 40 years; (2) a single prescribed burn; (3) repeat prescribed burns; (4) shelterwood harvests (average 25 percent residual basal area); and (5) diameter limit cuts removing merchantable trees of a minimum diameter. Each treatment is replicated on two sites within the Ridge and Valley physiographic province of West Virginia and Virginia. Transects are established on the east-northeast and south-southwest aspect of each site. Deer fences were constructed on half of all plots to test for the effect of deer herbivory. Survival after the first and second growing seasons is presented. No statistically significant differences in survival were found among management regimes in either year. Both first- and second-year results showed fencing to significantly increase survival. The fence x management regime interaction was also significant in both years. Survival on south-southwest aspects was statistically greater than on east-northeast aspects after two growing seasons.

INTRODUCTION

Oaks (*Quercus spp.*) are an important species group throughout the forests of the eastern United States. In the Ridge and Valley physiographic province, oak-hickory and oak-pine are the most prevalent forest types (Eyre 1980, McNab and Avers 1996). Associated with this prevalence and geographic extent is ecological and economic importance. However, despite widespread dominance in eastern deciduous forest ecosystems, the future status of oak is in question. On many sites, the size and quantity of advance reproduction is inadequate for successful regeneration and the perpetuation of oaks as a major component of future stands (Widmann and others 2012, Woodall and others 2008). This is a concern and has prompted research to better understand the origin of oak dominance, the drivers behind the inadequacy of reproduction, and prescriptions which address this inadequacy. However, widely applicable and consistently successful solutions have proven elusive, and continued research is necessary (Dey and others 2009, 2010; Johnson and others 2002; Loftis 2004).

Prescribed burning and shelterwood harvests are two of the most widely researched and implemented prescriptions to promote the establishment and growth of oak advance reproduction. Intermittent low intensity burning by Native Americans is frequently cited as a major factor contributing to the historical dominance of oak (Abrams 1998; Brose and others 2001; Guyette and others 2006; Hart and Buchanan 2012; Hutchinson and others 2008; Nowacki and Abrams 2008; Pyne 1997, 2001). Not surprisingly, the reintroduction of fire, in the form of prescribed burning, has garnered attention. The shelterwood method is intended to facilitate the growth of large oak advance reproduction by incrementally removing the overstory. This maintains sufficient canopy and shading to curb the establishment and growth of fast-growing shade-intolerant competitors while favoring oaks (Brose and Van Lear 2011; Dey and Parker 1997; Dey and others 2008, 2010; Downs and others 2011; Hannah 1988; Iverson and others 2008; Johnson and others 2002; Loftis 1990; Nyland 2007; Schlesinger and others 1993). However, research on the effectiveness of prescribed fire and the shelterwood method has yielded mixed results. In addition, both methods require patience and flexibility for full and proper implementation. When time and flexibility are not available, or oak advance reproduction is insufficient to meet management objectives even following implementation, then augmenting natural oak reproduction with underplantings may be desired

[1]Master's Candidate and Professor, respectively, West Virginia University, Division of Forestry and Natural Resources, Morgantown, WV 26506; and Research Botanist, USDA Forest Service, Northern Research Station, Morgantown, WV 26505.

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(Johnson and others 1986, Sander 1971, Sander and others 1976, Schuler and Robison 2010).

In addition to these silvicultural practices, diameter-limit cutting is primarily guided by short-term economic considerations as opposed to efforts to manage future species composition and regeneration (Nyland 2005). As a result, the environmental conditions created by diameter-limit cuts can be highly variable. In some cases, diameter-limit cutting may contribute to what Abrams and Nowacki (1992) refer to as post-logging accelerated succession. It perpetuates the transition to more shade-tolerant species composition by failing to create openings of sufficient size to promote the development of shade-intolerant species. When large portions of the overstory are removed, accumulated advance reproduction of shade-tolerant species in the understory is released (Abrams and Nowacki 1992; Dey and others 2010; Johnson and others 2002; Nyland 2005, 2007). Because of the widespread use of diameter-limit cutting, its effect on oak advance reproduction and the potential viability of underplanting in the absence of desirable reproduction merits study.

This study examines the 2-year survival of northern red oak (*Q. rubra* L.) underplantings under five different forest management regimes: (1) control sites which were characterized by no harvesting or evident disturbance within 40 years, (2) a single prescribed burn, (3) repeat prescribed burns, (4) diameter-limit cuts, and (5) the seedcut of a shelterwood harvest. In addition, the effects of aspect and fencing to exclude deer were tested, as well as the interactions of all three factors.

MATERIALS AND METHODS

Study Area and Site Description

Ten sites, two replicates of each management regime, were located within the Ridge and Valley physiographic province of Virginia and West Virginia. The province is bounded to the west by the Appalachian Plateau's Allegheny Front and by the Blue Ridge Mountain province to the east. Topography is dominated by long, parallel, southwest-northeast oriented ridges and broad valleys. Oak-hickory and oak-pine are the predominant forest types (Eyre 1980). Mean annual temperatures and precipitation specific to study sites were estimated using National Climate data from nearby weather stations. In the Franklin, WV and Harrisonburg, VA area, mean annual temperature is 11.3 °C and mean annual precipitation is 91.5 cm. Estimated mean annual temperature and precipitation are 12.5 °C and 103.9 cm for sites in the vicinity of Moorefield, WV and Front Royal, VA, respectively (NOAA 2012).

Soils are predominantly silt-loams of the Calvin, Cateach, Dekalb, Berks, Opequon, Faywood, Schaffenaker-Drall, Lehew-Hazleton-Dekalb, and Shouns series. These are classified as moderately deep and well-drained to excessively well-drained with moderate to rapid permeability (NRCS 2012). Site indices for these soils range from 18.3 to 21.3 m for northern red oak, base age 50.

Basal area on control sites averaged 27 m^2/ha and was dominated by mixed oaks and a lesser component of pines. Single-burn sites received one spring prescribed burn within 10 years prior to planting while repeat burns were burned twice within that time period. Basal area on single- and repeat-burn sites was comparable to that on control sites, averaging 26 m^2/ha and 28 m^2/ha, respectively.

Overstories on single-burn sites were almost exclusively oak-dominated. Overstories on repeat-burn sites were oak-dominated as well, though one site had a notable component of sugar maple (*Acer saccharum* Marsh.). Diameter-limit cut sites varied in regard to minimum diameter harvested. Guidelines at one site called for all hardwoods > 35.6 cm in diameter at breast height (d.b.h.) to be harvested and softwoods left standing in 2009. The resulting stand had a basal area of 19 m^2/ha with overstories dominated by mixed oaks and a component of hemlock [*Tsuga canadensis* (L.) Carrière] and white pine (*Pinus strobus* L.). On the other diameter-limit cut site, only merchantable timber 45.7 d.b.h. and greater was harvested in 2007 and 2008. Basal area on this stand was 25.5 m^2/ha following harvest. Oaks were dominant in the overstory here as well, with a sizable component of red maple (*Acer rubrum* L.). Shelterwood sites were reduced to 25 percent residual basal area in 2008, resulting in oak-dominated stands with an average basal area of 2 m^2/ha.

Experimental Design and Statistical Analysis

Each site consisted of two 100-m transects, one each on east-northeast and south-southwest aspects, making a total of 20 transects. Actual aspect of east-northeast transects ranged between 350° and 176° azimuth. South-southwest aspects ranged between 182° and 280° azimuth. This design provided two replicates of each management regime plus aspect combination. Six 0.001-ha plots were established on a given transect. Of these six plots, three were randomly selected and fenced using 1.2-m-high woven-wire fencing. Fences were constructed around plots in a 3.8- by 3.8-m square. It is recognized that it is within a deer's ability to jump over fences of this height. However, given the relatively small area enclosed, it was assumed that this height posed a sufficient deterrent to deer. Three 1+0 bareroot oak seedlings were planted within each plot in alignment with cardinal directions. Initial designs called for 10 plots per transect to be planted with four seedlings per plot. This was adjusted due to time constraints during planting. Therefore, those transects planted first included a greater number of plots and seedlings, resulting in a total of 146 plots and 478 seedlings.

Seedlings were purchased from Clements State Tree Nursery in West Columbia, WV in March 2011 and stored at 5 °C until planting. Seedlings were of unimproved stock and grown from seed collected throughout West Virginia and southern Ohio. Planting was conducted during April and May 2011. Seedlings were kept damp and shaded during planting. The spring of 2011was cool and wet, with much of the study area receiving above average rainfall in April and May making for good planting conditions (NOAA 2012).

Survival was recorded as a binary response variable. Survival at the end of the first growing season was recorded during October and November 2011. Survival at the end of the second growing season was recorded during October through December of 2012.

Mixed linear models were tested using the MIXED procedure in SAS 9.3®. This allowed for the inclusion of site and plot in the model as random effects. Fixed effects included management regime, aspect, and fencing. As plots were the experimental units, survival was averaged at the plot level. Percent survival was arcsine-square root transformed to assure homogeneity of variances.

RESULTS

Year One

The overall survival rate at the end of one growing season was 72 percent. Among management regimes, the highest survival rate was found on repeat-burn sites (86 percent), followed by shelterwood sites (79 percent), single-burn sites (74 percent), control sites (61 percent), and finally diameter-limit cut sites (58 percent). Survival on south-southwest aspects was nearly identical to that on east-northeast aspects (72 versus 71 percent, respectively). Fenced plots experienced a high average survival rate of 85 percent relative to 62 percent on unfenced plots (table 1). Only fencing ($P < 0.0001$) and the regime x fence interaction ($P = 0.0059$) were statistically significant at $\alpha = 0.05$ (table 2).

Table 1--Mean survival by factor level for years 1 and 2

	Percent survival (standard error of mean)	
Treatment level	Year 1	Year 2
Control	61.5 (5.6)	46.9 (5.6)
DLC	58.3 (7.8)	56.9 (8.1)
Repeat burn	85.9 (4.6)	74.4 (5.3)
Single burn	73.6 (4.9)	70.8 (5.8)
Shelterwood	79.2 (4.2)	79.4 (3.8)
Unfenced	62.5 (4.0)	57.3 (4.2)
Fenced	81.1 (2.7)	74.8 (3.2)
Northeast	71.2 (3.5)	61.6 (3.8)
Southwest	72.2 (3.6)	70.8 (3.7)
Total	71.9 (2.5)	66.2 (2.7)

Table 2--Type III test of fixed effects on survival in years 1 and 2

Effect	Num DF	Den DF[a]	F Value	Pr > F
	---------------------------Year 1------------------------------			
Regime	4	5.39	1.9	0.2409
Fence	1	130	19.04	<0.0001
Aspect	1	130	0.28	0.5968
Fence x Regime	4	130	3.8	0.0059
	---------------------------Year 2-------------------------------			
Regime	4	5.12	1.26	0.3931
Fence	1	129	19.45	<0.0001
Aspect	1	129	5.03	0.0266
Fence x Regime	4	129	3.85	0.0054

[a]Denominator degrees of freedom adjusted using Satterthwaite's adjustment.

Comparing average survival rates on fenced versus unfenced plots within a given management regime revealed a statistical difference to be present only on diameter-limit cut sites ($P = 0.0002$). On these sites, survival on fenced plots averaged 83 percent compared with 33 percent on unfenced plots. When comparing survival rates between management regimes in the absence of fencing, both unfenced repeat-burn (82 percent) and shelterwood plots (77 percent) experienced statistically greater survival than unfenced diameter-limit cut plots (33 percent). When only fenced plots were examined, tests showed no statistical differences between management regimes.

Year Two

The overall survival rate decreased to 66 percent following two growing seasons. Among management regimes, the highest survival rate was found on shelterwood sites (79 percent), followed by repeat-burn sites (74 percent), single-burn sites (71 percent), diameter-limit cut sites (57 percent), and control sites (47 percent). Survival on south-southwest aspects was greater than on east-northeast aspects (71 versus 62 percent). Average survival on fenced plots (75 percent) remained higher than on unfenced plots (57 percent) (table 1). Fencing ($P < 0.0001$), aspect ($P = 0.0266$), and management regime x fence interaction ($P = 0.0054$) were statistically significant (table 2).

As with first growing season results, only under diameter-limit cuts were survival rates greater in fenced plots (83 percent) than unfenced plots (31 percent, $P < 0.0001$). Among fenced plots, there were no statistically significant differences between management regimes after two growing seasons. This was true among unfenced plots as well.

DISCUSSION

Results from this study are mixed. The significance of the fence effect, particularly on diameter-limit cut sites, is evidence of the ability of deer herbivory to negate the effects of cultural prescriptions. Estimates from the 2012 hunting season and communication with resource managers placed deer densities in the region at approximately 7 to 7.5 deer/km^2. While this is not extremely high, studies have shown 7.9 deer/km^2 to be the threshold at which herbivory initiates a shift in species composition and negatively affects regeneration (Tilghman 1989). Such regional estimates may be of limited use, however, as density is difficult to quantify and varies across space and time. Further, browse pressure is not simply a function of density but also the appeal and "apparency" of plants in relation to the surrounding species and landscape (Seagle and Liang 1997). As with many management regimes, diameter-limit cutting creates visible disturbances on the landscape and produces browse attractive to deer. In addition, one diameter-limit cut site consisted of multiple stands located on larger properties which were harvested intermittently. This landscape-level disturbance regime may have produced browse to support larger deer populations relative to more contiguous mature forests. On the other diameter-limit cut site, the landowner was known to feed deer, increasing density in the immediate vicinity. It is therefore possible that higher deer densities specific to these diameter-limit cut sites, and not the regime per se, were responsible for the significant management regime x fence interaction. In addition to the management regime x fence interaction, lower 2-year survival on northeast aspects reflected more mesic conditions associated with greater interfering vegetation on these sites, making them less conducive to oak regeneration.

The lack of statistical differences in survival between management regimes was contrary to expectations. However, this is likely the result of the limited number of replicates, making statistical significance difficult to achieve. Though not significant, higher survival under the shelterwood and burn regimes is consistent with the objective of the prescriptions. A study with a greater number of replications may show a stronger statistical relationship between survival and regimes. While the average survival rate among diameter-limit cut sites was low, it was relatively high on fenced plots. This suggested that, when deer browse was limited, conditions for underplanting within diameter-limit cuts were comparable to those under these other management regimes.

CONCLUSION

As oak regeneration remains problematic, continued research on methods to promote sufficient advance reproduction is needed. Establishing this through a natural regeneration system is possible on more xeric sites such as those within the Ridge and Valley (Larsen and Johnson 1998). However, prescriptions such as prescribed burning and the shelterwood method require patience and do not guarantee the desired size and quantity of reproduction. Further, diameter-limit cutting may leave stands exhausted of seed trees of desired species and quality.

The results of this study are encouraging regarding the use of underplanting in conjunction with these management activities when deer herbivory is not a concern. However, caution should be exercised when drawing conclusions about the long-term success of these seedlings. In the absence of data on the current state of competition, speculation on their future performance is tenuous. Keeping this in mind, this study supports continued research on the use of underplanting in the Ridge and Valley as more tools and flexibility are desired in promoting oak regeneration.

LITERATURE CITED

Abrams, M.D. 1998. The red maple paradox. BioScience. 48: 355-364.

Abrams, M.D; Nowacki, G.J. 1992. Historical variation in fire, oak recruitment, and post-logging accelerated succession in central Pennsylvania. Bulletin of the Torrey Botanical Club. 119: 19-28.

Brose, P.; Schuler, T.; Van Lear, D.; Berst, J. 2001. Bringing fire back: the changing regimes of the Appalachian mixed-oak forests. Journal of Forestry. 99: 30-35.

Brose, P.H; Van Lear, D.H. 2011. Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. Canadian Journal of Forest Research. 28: 331-339.

Dey, D.C.; Douglass, J.; 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.

Dey, D.C.; Parker, W.C. 1997. Overstory density affects field performance of underplanted red oak (*Quercus rubra* L.) in Ontario. Northern Journal of Applied Forestry. 14: 120-125.

Dey, D.C.; Royo, A.A.; Brose, P.H. [and others]. 2010. An ecologically based approach to oak silviculture: a synthesis of 50 years of oak ecosystem research in North America. Revista Columbia Forestal. 13: 201-222.

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.

Downs, J.D.; Williams, R.A.; Downs, J.A. 2011. The effects of shelterwood harvesting on oak regeneration two years after harvest in southern Ohio. 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: 262-269.

Eyre, F.H., ed.1980. Forest cover types of the United States and Canada. Bethesda, MD: Society of American Foresters. 148p.

Guyette, R.P.; Dey, D.C.; Stambaugh, M.C.; Muzika, R. 2006. Fire scars reveal variability and dynamics of eastern fire regimes. In: Dickinson, M.B., ed. Fire in eastern oak forests: delivering science to land managers. Proceedings of a conference. Gen. Tech. Rep. NRS-P-1. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 20-39.

Hannah, P.R. 1988. The shelterwood method in northeastern forest types: a literature review. Northern Journal of Applied Forestry. 5: 70-77.

Hart, J.L.; Buchanan, M.L. 2012. History of fire in eastern oak forests and implications for restoration. 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: 34-51.

Hutchinson, T.F.; Long, R.P.; Ford, R.D.; Sutherland, E.K. 2008. Fire history and the establishment of oaks and maples in second-growth forests. Canadian Journal of Forest Research. 38: 1184-1198.

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, P.S.; Dale, C.D.; Davidson, K.R.; Law, J.R. 1986. Planting northern red oak in the Missouri Ozarks: a prescription. Northern Journal of Applied Forestry. 3: 66-68.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2002. The ecology and silviculture of oaks. Wallingford, Oxon: CABI Publishing. 580 p.

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. 1990. A shelterwood method for regenerating red oak in the Southern Appalachians. Forest Science. 36: 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.

McNab, H.W. and Avers, P.E. 1996. Ecological subregions of the United States. USDA Forest Service. http://www.fs.fed.us/land/pubs/ecoregions/. [Date accessed December 19, 2013].

NOAA [National Oceanic and Atmospheric Administration]. 2012. National Climatic Data Center. http://www.ncdc.noaa.gov/cdo-web/. [Date accessed: February 12, 2013].

NRCS [Natural Resource Conservation Service]. 2012. Soil data mart. http://soildatamart.nrcs.usda.gov/. [Date accessed: June 12, 2012].

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the eastern United States. BioScience. 58: 123-138.

Nyland, R.D. 2005. Diameter-limit cutting and silviculture in northern hardwoods. In: Kenefic, L.S.; Nyland, R.D., eds. Proceedings of the conference on diameter-limit cutting in northeastern forests. Gen. Tech. Rep. NE-342. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 16-23.

Nyland, R.D. 2007. Silviculture: concepts and applications: 2d ed. Long Grove, IL: Waveland Press. 704 p.

Pyne, S.J. 1997. America's fires: management on wildlands and forests. Durham, NC: Forest History Society. 54 p.

Pyne, S.J. 2001. Fire: a brief history. Seattle: University of Washington Press. 204 p.

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.; Johnson, P.S.; Watt, R.F.1976. A guide for evaluating the adequacy of oak advance reproduction. Gen. Tech. Rep. NC-23. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 7 p.

Schlesinger, R.C.; Sander, I.L.; Davidson, K.R. 1993. Oak regeneration potential increased by shelterwood treatments. Northern Journal of Applied Foresty. 10: 149-153.

Schuler, J.L.; Robison, D.J. 2010. Performance of northern red oak enrichment plantings in naturally regenerating southern Appalachian hardwood stands. New Forests. 40: 119-130.

Seagle, S.W.; Liang, L-Y. 1997. Bottomland forest composition and seedling diversity: simulating succession and browsing by overabundant deer. In: McShea, W.J.; Underwood, H.B.; Rappole, J.H., eds. The science of overabundance: deer ecology and population management. Washington, DC: Smithsonian Institution Press: 346-365.

Tilghman, N.G. 1989. Impacts of white-tailed deer on forest regeneration in northwestern Pennsylvania. The Journal of Wildlife Management. 53(3): 524-532.

Widmann, R.H.; Cook, G.W.; Barnett, C.J. [and others]. 2012. West Virginia's forests 2008. Resour. Bull. NRS-61. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 64 p.

Woodall, C.W.; Morin, R.S.; Steinman, J.R.; Perry, C.H. 2008. Status of oak seedlings and saplings in the northern United States: implications for sustainability of oak forests. In: Jacobs, D.F.; Michler, C.H., eds. Proceedings of the16th central hardwood forest conference. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 435-542.

THE CONUNDRUM OF CREATING UNDERSTORY LIGHT CONDITIONS CONDUCIVE TO PROMOTING OAK REPRODUCTION: MIDSTORY HERBICIDE AND PRESCRIBED FIRE TREATMENTS

Callie Jo Schweitzer and Daniel C. Dey[1]

Abstract--Challenges remain to regenerating oak (*Quercus* spp.) in eastern upland hardwood forests. It is well established that to be competitive, advance oak reproduction must be of sufficient size to respond favorably upon release. We also know altering the understory light regime, through reductions in stand stocking, basal area, or canopy cover, promotes the growth and competitiveness of small oak advance reproduction. Reductions in stand density can be accomplished by herbicide treatment of the existing midstory, prescribed fire, or harvesting. We found midstory herbicide treatment resulted in an ephemeral increase in light, from < 1 percent of full sunlight pretreatment to 15 to 20 percent post-treatment. This increase in light levels dissipated after three growing seasons. Oak seedlings responded to increased light levels but so did both sugar maple (*Acer saccharum* L.) and yellow-poplar (*Liridendron tulipifera* L.). One and two prescribed fires, implemented on a 3-year-return interval, resulted in 12 percent of full sunlight, compared to 8 percent pre-burn. Conversely, a one-time thin to 50 BA (basal area in square feet per acre) resulted in 50 percent full sun following the initial thin and burn and 37 percent after the second burn. Because these fires were repeated on a 3-year interval, the increase in light was sustained over time compared to the thin only. However, much of the growing space was occupied by aggressive red maple (*Acer rubru*m L.) sprouts after the prescribed burning.

INTRODUCTION

Maintaining or enhancing the oak (*Quercus* spp.) component in upland hardwood forests of the Cumberland Plateau and associated highlands has been a goal for generations of silviculturists. We have learned that appropriate prescriptions are site-specific and driven by the concurrent response of the oak and its competitors. Changes in cover and light are transient and will alter vegetation response, including seedling recruitment (Chiang and others 2005, Iverson and others 2004). Two prescriptions have been touted to create the desired understory conditions to promote oak, including enhanced light and reduced numbers of competitive stems: the use of an herbicide to deaden midstory non-oak species and prescribed fire. Both prescriptions have challenges in their application.

For example, herbicide use is often restricted on National Forest lands due to the public's lack of knowledge and understanding of this treatment. Further, the use of herbicide can be costly for a private landowner, as this treatment is completely non-commercial. Oak survival and growth benefit from the removal of the midstory (Janzen and Hodges 1987, Lhotka and Loewenstein 2008, Lockhart and others 2000, Loftis 1990, Lorimer and others 1994, Miller and others 2006, Motsinger and others 2010, Stringer 2005) but on productive sites, the pulse of increased light reaching the understory is ephemeral (Schweitzer and Dey 2011). Private landowners often are swayed by prescriptions that are promoted to enhance wildlife. Wang and others (2006) found that the habitat created following the deadening of the midstory in upland hardwood forests in the Mid-Cumberland Plateau was positively correlated with the use of the dead stems by bark-foraging and cavity-nesting birds. On the same sites, Felix and others (2010) found that treatments where midstory trees were deadened with herbicide had no effect on either biophysical parameters or egg mass number of amphibians. Similar treatments had no negative effect on terrestrial salamander abundance or demographics (Knapp and others 2003) or on herpetofaunal abundance, richness, or diversity (Cantrell and others 2013).

Prescribed fire in southeastern upland hardwood forests is primarily practiced on public lands, where costs and liabilities are subsidized. Fire has been promoted as a tool that will restore the oak component to upland hardwood forests (Albrecht and McCarthy 2006, Dey and Hartman 2005, Hutchinson and others 2005, Moser and others 2006 Nyland and others 1983, Van Lear and Waldrop 1989). Results from most studies suggest that prescribed burning alone, without additional disturbances involving overstory tree harvesting, will not significantly promote oak

[1]Research Forester, USDA Forest Service, Southern Research Station, Huntsville, AL 35801; and Research Forester, USDA Forest Service, Northern Research Station, Columbia, MO 65211.

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reproduction over the short-term (Blankenship and Arthur 2006, Hutchinson and others 2005, Signell and others 2005). Some have shown that large reductions in canopy cover followed by prescribed fire facilitated regeneration of desirable oak species (Albrecht and McCarthy 2006, Brose 2010, Brose and others 1999). A more detailed ecological understanding of the life stages of the oak genus may allow prescribed fire to be applied as a tool following a life-history gradient to enhance or sustain this species (Arthur and others 2012). Additional concerns on subsequent degrade to residual trees following fire is also receiving renewed, heightened attention (Dey and Schweitzer, in press; Smith and Sutherland 1999; Stambaugh and Guyette 2008). Fire has been documented to be beneficial to certain wildlife species. Herpetofaunal response to fire disturbance has been found to be mixed, with negligible or ephemerally positive impacts on amphibians and positive or non-measurable impacts for reptiles (Sutton and others 2013). Studies of avian communities have found prescribed fire to have no discernible impact on breeding bird composition or total population levels (Aquilani and others 2000, Artman and others 2001, Saab and others 2004).

The objectives of this study were to use results from an herbicide-midstory treatment and a prescribed fire treatment in two common Mid-Cumberland Plateau forest ecosystems, the Plateau tabletop and the Plateau slope or escarpment, to examine: (1) the temporal response of light in the understory, and (2) the response of the oak reproduction to ascertain how these prescriptions are meeting target management goals.

METHODS

Study Areas

The 180,000-acre Bankhead National Forest (BNF), in north-central Alabama, is in the Cumberland Plateau Section of the Appalachian Plateaus physiographical province (Fenneman 1938), and study stands are more specifically characterized by the Strongly Dissected Plateau subregion of the Southern Cumberland Plateau, within the Southern Appalachian Highlands (Smalley 1979). These are Plateau tabletop sites. Base age 50 site indices for loblolly pine (*Pinus taeda* L.), red oaks [northern red oak (*Q. rubra* L.), black oak (*Q. velutina* Lam.), scarlet oak (*Q. coccinea* Munchh.), and southern red oak (*Q. falcata* Michx.)], and white oaks [white oak (*Q. alba* L.) and chestnut oak (*Q. prinus* L.)] are 75 feet, 65 feet, and 65 feet, respectively (Smalley 1979). Soils are loamy, formed in residuum weathered from sandstones and conglomerates (Smalley 1979). Climate of the region is temperate with mild winters and moderately hot summers with a mean temperature of 55.4 °F and mean precipitation of 59 inches (Smalley 1979). Study stands were located on broad, flat ridges with elevations ranging from 720 to 1,220 feet above sea level. The BNF, established by proclamation in 1914, has a long history of repeated logging and soil erosion caused by poor farming practices during the Depression era. Study stands were representative of this history, with non-managed loblolly pine planted 25 to 45 years ago and substantial hardwood encroachment. There were three pine species, dominated by loblolly pine at 87 percent of the total basal area (BA), with a smaller portion of Virginia pine (*P. virginiana* Mill.) and shortleaf pine (*P. echinata* Mill.). Other species included upland oaks (chestnut oak, white oak, northern red oak, scarlet oak, black oak and southern red oak) representing 7 percent of stand BA, yellow-poplar (*Liriodendron tulipifera* L.) at 6 percent of BA, and red maple (*Acer rubrum* L.) at 2 percent of BA. Understory vegetation included flowering dogwood (*Cornus florida* L.), bigleaf magnolia (*Magnolia macrophylla* Michx.), and sourwood (*Oxydendrum arboretum* DC).

The second study area was located in Jackson County, northeastern Alabama, within the Cumberland Plateau section of the Appalachian Plateaus physiographic province (Fenneman 1938). The site encompasses strongly dissected margins and sides of the plateau and represents Mid-Plateau escarpment sites. Slopes range from 15 to 30 percent. Upland oak site index is 75 to 80, and yellow-poplar site index is 100 [base age 50 years, Smalley Landtype 16, plateau escarpment and upper sandstone slopes and benches-north aspect (Smalley 1982)]. 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). We conducted our study at two separate sites. One site with one block of treatments was located on a south, southwest-facing slope of Miller Mountain with a mean elevation of 1,600 feet. The second site was located on a north-facing slope at Jack Gap, with two blocks: one at 1,200 feet and one at 1,562 feet elevation. Dominant canopy tree species on both sites included *Quercus*, with black oak, northern red oak, white oak, and chestnut oak comprising 46 percent of pretreatment BA. Hickory (*Carya* spp.) accounted for 15 percent of the pretreatment BA, while sugar maple (*A. saccharum* Marsh.) was 13 percent and yellow-poplar 9 percent. Common understory species included flowering dogwood, eastern redbud (*Cercis canadensis* L.), and sourwood.

Study Design

The BNF study employed a randomized complete block design with a 3 by 3 factorial treatment arrangement and four replications of each treatment. For this study, we are only considering six of the total treatments: two residual basal area treatments (heavy thin leaving 50 square feet per acre and an untreated control) with three prescribed burn frequencies: frequent burns every 3 years (stands have received two burns to date), infrequent burns every 9 years (stands have received one burn to date), and an unburned control. Each treatment was replicated 4 times, for a total of 24 treatment stands. Stand size ranged from 22 to 46 acres. Treatments are representative of management practices described in the BNF's Forest Health and Restoration Project for restoring oak forests and woodlands (USDA Forest Service 2003).

Criteria for stand selection were based on species composition, stand size, and stand age. Treatment stands were at least 22 acres in size with basal areas ranging from 122 to 132 square feet per acre (table 1). Commercial thinning was conducted by marking from below smaller trees or trees that appeared diseased or damaged; canopy trees were also removed to meet target residual basal areas. Hardwoods were preferentially retained. Thinning treatments were completed prior to the initiation of the burning treatments (thinning conducted from June through December). Prescribed burning was conducted during the dormant season (January through March) using backing fires and strip head fires to ensure that only surface fire occurred.

The study design for the Jackson County study was a randomized complete block, with three replications of five treatments. Only two treatments are considered for this analysis. Each site (block) comprised one replication of five treatments established along the slope contour. Treatments were randomly assigned to 10-acre areas within each replicated block. Pretreatment basal areas ranged between 101 to 116 square feet per acre (table 2).

For the midstory herbicide treatment, Arsenal® (active ingredient imazapyr, BASF Corporation, Florham Park, N.J.) was used to deaden the midstory. Rates of application were within the range recommended by the manufacturer. Watered solutions were made in the laboratory, and trees received application via waist-level hatchet wounds using a small, handheld sprayer. One incision was made per 2.5 inch of diameter, and each incision received approximately 0.15 fluid ounces 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 oak, ash (*Fraxinus americana* L.) and persimmon (*Diospyros virginiana* L.) stems were excluded from treatment. Three control stands were left untreated.

Field Techniques

On all study sites, we established five 0.2-acre vegetation measurement plots in each treatment stand and measured plots prior to and one growing season after treatment implementation, and again 4 to 8 years post-treatment. All plot centers were permanently marked with rebar, flagging, and GPS coordinates. We permanently tagged all trees > 5.6 inches diameter at breast height (d.b.h.) with aluminum tags. Tree distance and azimuth to plot center were

Table 1--Cumberland Plateau tabletop stands, basal area (BA in square feet per acre) and stems per acre (SPA) of trees ≥ 5.6 inches d.b.h. for six treatments, at times pre (prior to treatment), post1 (first growing season post-treatment) and post2 (first growing season post second burn or 4 years post thin and first burn)[a]

Treatment	Burn	Pre BA	Pre SPA	Post1 BA	Post1 SPA	Post2 BA	Post2 SPA
No harvest	None	131.7	265	137.5a	264a	147.3a	269a
No harvest	One	123.1	381	127.1a	275a	137.0a	277a
No harvest	Two	121.8	318	130.2a	317a	146.2a	333a
Thin to 50 BA	None	132.0	296	50.6b	86b	59.3b	90b
Thin to 50 BA	One	132.0	300	49.5b	86b	57.0b	87b
Thin to 50 BA	Two	127.5	278	50.0b	84b	56.7b	86b
p-value		0.4134	0.7131	0.0001	0.0001	0.0001	0.0001

[a]Columns containing different letters indicate significant differences at $\alpha = 0.05$.

Table 2--Cumberland Plateau escarpment stands, basal area (BA in square feet per acre) and stems per acre (SPA) of trees ≥ 5.6 inches d.b.h. for five treatments at times pre (prior to treatment), post1 (first growing season post-treatment) and post2 (fourth growing season post-treatment). Treatments are control (no harvest or herbicide) and midstory herbicide treatment

Treatment	Pre BA	Pre SPA	Post1 BA	Post1 SPA	Post2 BA	Post2 SPA
Control	101.2	331	100.6	325	110.7	335
Herbicide	116.3	372	114.6	271	122.8	245
p-value	0.4668	0.4081	0.4441	0.02314	0.5002	0.2266

recorded, and we measured and recorded tree species and d.b.h. (diameter tape, to the nearest 0.1 inch). Reproduction was sampled on 0.01-acre circular plots around the same plot center. Seedlings were tallied by species in each reproduction plot by 1-foot height classes, up to 4.5-feet tall. Canopy cover was estimated using a hand-held spherical densitometer, with five measurements obtained at each plot, one 10 feet from plot center in each cardinal direction and one at plot center. Photosynthetically active radiation was measured using two synchronized ceptometers (AccuPar LP-80, Decagon Devices, Pullman, CA). One ceptometer was placed in full sunlight, and the second ceptometer was used to record light in each stand along pre-designated transects.

RESULTS AND DISCUSSION

Jackson County Midstory Herbicide Treatment

The majority of the targeted trees removed in the herbicide treatment were occupying a midstory position. To document the targeted tree response in the herbicide treatment, additional tree data were recorded for all trees 1.5 inches d.b.h. and greater on the Jackson County sites. An average of 381 stems per acre (SPA) were treated with the herbicide, and their average d.b.h. was 2.9 inches. The BA of trees 1.5 inches d.b.h. and larger for the herbicide treatment was reduced from 115.9 square feet per acre to 86.1 square feet per acre, meeting the targeted 75 percent BA retention. Following treatment, the canopy tree BA (represented by trees > 5.6 inches d.b.h.) was not different from the control stands (table 2).

Table 3 gives the percent of full sunlight received in the understory for the month of August, for one, four and eight growing seasons post-treatment. The percent of canopy cover is also given in table 3. Only during the first growing season post-treatment was the amount of light reaching the understory in the herbicide treatment significantly greater than controls, 16.5 percent compared to 8 percent of full sunlight. By the second post-treatment growing season, light was 10 percent of full sunlight, and remained around this level for eight growing seasons post-treatment. There was no difference in the amount of overstory canopy cover, as the herbicide treatment targeted midstory trees only.

Table 3--Percentage of full sunlight received and canopy cover in the control and midstory deadened with herbicide treatments over eight growing seasons, for the month of August, Cumberland Plateau escarpment stands

	---2001 (pretreatment)---		---------2002----------		----------2004---------		----------2009---------	
	full sun	canopy cover	full sun	canopy cover	full sun	canopy cover	full sun	canopy cover
	percent							
Control	n/a[a]	99.7	8.0	96.8	3.0	98.8	5.2	97.5
Herbicide	n/a	98.3	16.5	94.6	6.3	96.2	8.3	93.4

[a]n/a = no pretreatment (2001) data collected.

The response of the reproduction cohort was assessed by looking at all oaks, yellow-poplar, sugar maple, and all other species combined. Oak in the reproduction stratrum were primarily black and white oaks, with lesser representation of northern red, scarlet and chestnut oaks. The other species category included a host of species: ash (*Fraxinus* spp. L.), basswood (*Tilia americana* L.), American beech (*Fagus grandifolia* Ehrh.), blackgum (*Nyssa sylvatica* March.), elm (*Ulmus* spp. L.), flowering dogwood, hickory (*Carya* spp.) and sassafras [*Sassafras albidum* (Nutt.) Nees]. The number of oaks in control stands did not change over time, while the number of small oak seedlings, up to 1 foot in height, decreased appreciatively in the herbicide treatment, from 1,523 to 1,142 SPA (table 4 and fig. 1). This decrease was countered by a concurrent increase in the SPA of oak in all other size classes, as oaks increased 400 SPA in the 1- to 2-foot size class, 62 SPA in the 2- to 3-foot size class, and 10 SPA in the 3- to 4.5-foot size class (table 4 and fig. 1). Following the herbicide treatment, the number of yellow-poplar seedlings also increased, from 52 to 1,599 SPA, but after eight growing seasons this number consistently declined as the amount of light dissipated. However, the number of sugar maple stems in the three largest size categories increased, with changes of 282, 271, and 62 SPA for the 1- to 2-foot, 2- to 3-foot, and 3- to 4.5-foot height categories, respectively. Interestingly, the number of sugar maple in the understory also increased in the control stands, with a positive change of 305, 48, and 10 SPA in the same size categories, respectively.

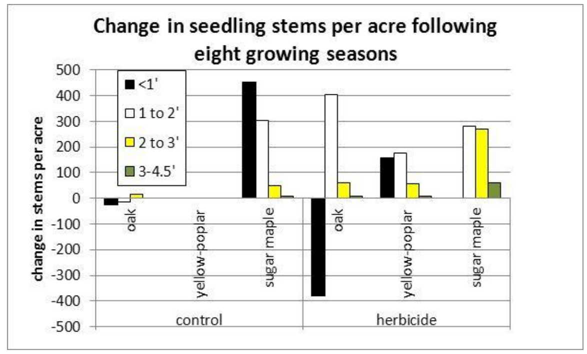

Figure 1--The change in stems per acre of oak, yellow-poplar, and sugar maple seedlings by four height classes, for Cumberland Plateau escarpment stands under untreated (control) and herbicide (removal of midstory trees using herbicide) prescriptions after eight growing seasons.

Bankhead National Forest Thinning and Burning Treatments

The BNF stand basal area and stems per acre are given in table 1. Pretreatment stand BA was dominated by loblolly pine, which accounted for 87 percent of the BA and 85 percent of the SPA. The thin to 50 BA was successfully implemented, with the majority of the removed BA comprised of pine. The prescribed burning had no effect on overstory composition and structure ($P = 0.6697$), and there were no burn by thinning interactions ($P = 0.7040$). The percent of total BA and SPA for upland oaks was 7 and 8 percent, for yellow-poplar 6 and 5 percent, and for both red maple and black cherry, 2 and 3 percent, respectively.

The percent of full sunlight received in each of the six treatments is shown in figures 2 and 3.

Table 4--Cumberland Plateau escarpment stands regeneration data, in stems per acre, by four height size classes, for all oak combined, yellow-poplar, sugar maple, and all other species combined. Year 2001 is pretreatment data

Oak	Control	2001	776	129	14	5
		2004	486	29	0	0
		2009	752	114	29	5
	Herbicide	2001	1,523	29	0	0
		2004	995	76	24	10
		2009	1,142	433	62	10
Yellow-poplar	Control	2001	0	0	0	0
		2004	205	0	0	0
		2009	0	0	0	0
	Herbicide	2001	52	10	5	0
		2004	1,599	243	5	5
		2009	209	186	62	10
Sugar maple	Control	2001	1,057	119	29	19
		2004	857	224	52	24
		2009	1,509	424	76	29
	Herbicide	2001	1,485	143	71	76
		2004	1,157	362	62	38
		2009	1,485	1,157	343	138
Other species	Control	2001	3,446	1,228	505	190
		2004	3,027	1,228	443	257
		2009	4,022	1,918	724	367
	Herbicide	2001	3,975	981	257	205
		2004	3,108	962	386	176
		2009	3,451	3,046	1,333	752

Stands that were not subjected to burning or thinning had the lowest light levels. Thinning to 50 BA resulted in at least 50 percent of full sun, compared to an average of 20 percent pre-thin. The amount of sunlight in stands that have received two burns was three times that of pre-burn levels, while the light was twice that of the pre-burn level for stands receiving only one burn. All three unthinned treatments showed light levels below 14 percent regardless of burn treatment. Pretreatment canopy cover ranged from 91.3 to 96.3 percent. Burning did not affect the canopy cover while thinning reduced canopy cover from 93 to 66 percent. After four growing seasons, the canopy cover increased to 83.3, 82.9 and 82.4 percent for the thin with no-burn, thin with one burn, and thin with two burn treatments, respectively.

The reproduction cohort was dominated by stems < 1-foot tall, with an average of 4,239 SPA in this size class among the six treatments (table 5). Small red maple seedlings (those < 1-foot tall) comprised 46 percent of the total in this size class pretreatment. Oaks were dominated by chestnut oak (29 percent of all oak regeneration tallied), white oak (26 percent), southern red oak (15 percent), and scarlet oak (14 percent). Other species of oaks tallied in the reproduction cohort included black oak, northern red oak, and post oak. Sixty-seven percent of the oak reproduction was < 1-foot tall.

Following the first treatment (thinning, burning or a combination), the oak and red maple both responded. Thinning alone increased the SPA of oak in all the larger size classes, and after four growing seasons there were 80 SPA of oak 3- to

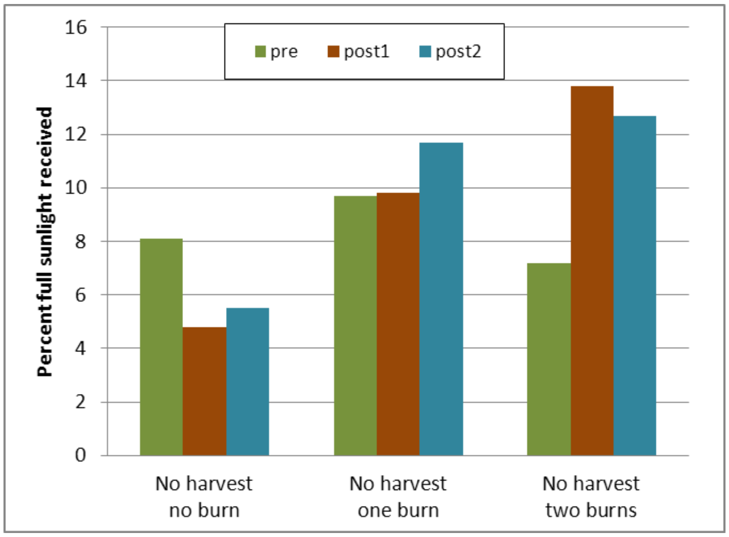

Figure 2--Percent of full sunlight received at 4.5 feet above the forest floor for Cumberland Plateau tabletop stands under three treatments. Stands were not thinned and were subjected to none, one, or two burns. Burns were conducted during the dormant season. Post1 light measurements taken in August of the first growing season and post2 measurements taken in August of the fourth growing season.

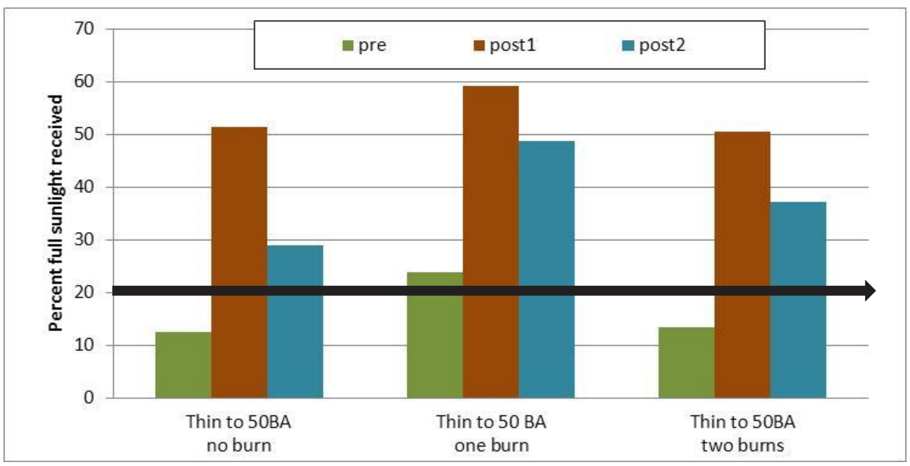

Figure 3--Percent of full sunlight received at 4.5 feet above the forest floor for Cumberland Plateau tabletop stands under three treatments. Stands were thinned to 50 square feet per acre residual basal area and were subjected to none, one, or two burns. Thinning was done in the growing season prior to the burn; burns were conducted during the dormant season. Post1 light measurements were taken in August of the first growing season, and post2 measurements were taken in August of the fourth growing season. The arrow represents the reported needed threshold of light for encouraging oak seedling growth.

4.5-feet tall (tables 5 and 6). Four growing seasons after just a single prescribed burn, oak seedlings increased by 300 SPA in the 2- to 3-foot height class; this increase was only 63 SPA when stands received two burns. Thinning alone also increased oak that was 3- to 4.5-feet tall (by 20 SPA to 80 SPA total), while there was a decline of 10 SPA following one burn and a gain of only 2 SPA in this size class after two burns. However, in concert with the increase in oak reproduction, red maple seedlings also increased in response to the thinning and burning. Red maple seedlings were lost from the < 1-foot height class following each burn. Subsequent recruitment was found in the next height class, as red maple seedlings increased by about 500 SPA in the 2- to 3-foot height class after one burn, and added about 400 SPA after two. However, a larger change was noted in the 3- to 4.5-foot height class, where red maple

Table 5--Cumberland Plateau tabletop stands regeneration data, in stems per acre, by four height size classes, for all oaks combined and red maple, for six treatments. BA is basal area in square feet per acre

	----------Treatment---------		----------------------------Seedling height class----------------------------			
Species	Harvest	Burn	Up to 1 foot	1 to 2 feet	2 to 3 feet	3 to 4.5 feet
Oak	No harvest	None	690	230	55	40
	No harvest	One	625	150	25	10
	No harvest	Two	800	235	40	0
	Thin to 50 BA	None	575	130	105	30
	Thin to 50 BA	One	845	475	155	80
	Thin to 50 BA	Two	500	163	64	46
Red maple	No harvest	None	1,975	165	110	45
	No harvest	One	2,155	240	155	75
	No harvest	Two	1,835	280	115	75
	Thin to 50 BA	None	2,160	90	40	35
	Thin to 50 BA	One	1,830	325	275	170
	Thin to 50 BA	Two	1,655	340	95	95

Table 6--Cumberland Plateau tabletop stands change in stems per acre (SPA) of seedlings by four height size classes for six treatments. Time at post1 is the first growing season post thin and first burn; time at post 2 is the first growing season post second burn (4 years post thin and first burn). BA is basal area in square feet per acre

	--------Treatment--------			------------------------Seedling height class-------------------------			
Species	Harvest	Burn	Time	Up to 1 foot	1 to 2 feet	2 to 3 feet	3 to 4.5 feet
Oak	No harvest	None	Post1	-20	-35	10	25
			Post2	20	-5	5	15
	No harvest	One	Post1	40	0	0	-5
			Post2	45	25	50	15
	No harvest	One	Post1	235	-45	-15	10
		Two	Post2	385	-30	-5	5
	Thin to 50 BA	None	Post1	26	163	30	86
			Post2	-80	68	35	20
	Thin to 50 BA	One	Post1	270	305	180	50
			Post2	285	405	300	-10
	Thin to 50 BA	One	Post1	-140	60	62	35
		Two	Post2	-272	17	63	2
Red maple	No harvest	None	Post1	-525	35	-40	15
			Post2	-280	165	-20	-15
	No harvest	One	Post1	-552	772	520	60
			Post2	65	370	285	45
	No harvest	One	Post1	66	475	266	52
		Two	Post2	401	531	351	32
	Thin to 50 BA	None	Post1	-1,540	555	500	240
			Post2	-1,185	245	175	130
	Thin to 50 BA	One	Post1	-770	890	490	175
			Post2	-795	1,150	495	55
	Thin to 50 BA	One	Post1	-889	791	573	112
		Two	Post2	-1,149	336	393	152

seedlings increased by 152 SPA to 247 SPA after two burns and added 55 SPA to total 225 large seedlings after one burn.

On more productive sites such as those found on the Cumberland Plateau escarpment, the outcome of reducing the midstory using an herbicide to increase light and allow small oak advance reproduction to grow into a more competitive position was questionable. Others have found mixed results with this prescription (Dillaway and others 2007, Gordon and others 1995, Harmer and others 2005, Janzen and Hodges 1987, Lhotka and Lowenstein 2008, Lockhart and others 2000, Loftis 1990, Lorimer and others 1994, Stringer 2005). We were able to non-commercially kill the midstory and allow an ephemeral increase in light compared to control stand conditions (Schweitzer and Dey 2011). However, the amount of light never exceeded the 20 percent threshold commonly reported in the literature as the minimal amount needed to promote oak seedling growth (Dey and others 2012, Gardiner and Hodges 1998, Gottschalk 1994, Lhotka and Lowenstein 2008, Lockhart and others 2000, Loftis 1990, Lorimer and others 1994, Motsinger and others 2010, Parker and Dey 2008, Stringer 2005). Although small oak seedlings did respond and increase SPA in the next larger height class, there was a concurrent initial response by a shade-intolerant competitor, yellow-poplar, which germinated from seed in the seed bank, with almost 1,600 seedlings per acre appearing in the second growing season post-treatment.

Additionally, the small sugar maple, those 1 to 2 inches in diameter that were not treated, also responded and soon occupied the midstory position. A compensatory increase in the larger size classes of the sugar maple reproduction also contributed to the competition for light in the understory. Sugar maple seedlings were being recruited in all size classes, in both the control and the herbicide treated stands. The canopy of these stands was dominated by sugar maple, and sugar maple was ranked first in importance value prior to any treatment (Schweitzer and Dey 2011); the importance of sugar maple dropped slightly post-herbicide treatment but was on the increase after eight growing seasons. We are not finding a notable reduction of sugar maple densities, suggesting vigorous sugar maple establishment and recruitment, unlike that found by others (Belden and Pallardy 2009, Pallardy and others 1988, Rochow 1972).

Success of this treatment will be gauged once the overstory trees are removed in the final phase of this prescription.

Combining reduced stand density prescriptions with prescribed burning to control competing vegetation has shown promise on certain sites (Albrecht and McCarthy 2006; Brose 2010; Dey and Hartman 2005; Iverson and others 2004, 2008). For those sites on the Cumberland Plateau escarpment, site characteristics may be prohibitive to the implementation of prescribed fire. Cumberland Plateau tabletop sites are more conducive to this treatment, mainly due to topography. With no stand density reduction, light levels remained below 14 percent, although following burning, there was an increase from 7 to 14 percent. Following one prescribed fire without any stand density reduction, SPA of oak seedlings in the larger size classes decreased; after two burns the response remained similar, although there was an increase of approximately 400 SPA in the smallest size class. Conversely, the effect on red maple, the primary oak competitor in these systems, was pronounced, with substantial recruitment and increases in all size classes. The fires implemented thus far have been done in January or February and are characterized as low intensity. Others have found that dormant season fires may have limited impact on light levels while encouraging sprouting of competitors (Alexander and others 2008, Green and others 2010, Hutchinson and others 2005).

Reducing stand density to 50 BA resulted in light levels increasing above the 20 percent level; burning after thinning did not impact understory light levels. On these more xeric sites, oak reproduction was abundant, but primarily < 1-foot tall. Four growing seasons post thin and one burn, oak seedlings in the 2- to 3-foot height classes increased by 700 SPA, while the same sized red maple seedlings increased by 1,600 SPA. In the largest size class, 3- to 4.5-foot tall, red maple SPA outnumbered oak seedlings by 155 SPA. Adding another burn did not change these relationships, with red maple seedlings outnumbering oak in all size classes, with 200 more SPA in the largest class. Resprouting of red maple has been noted by others following prescribed burns (Arthur and others 2012, Dey and Hartman 2005, Waldrop and others 1992), and managers are increasingly aware that using fire in hardwood systems must be more targeted

(Alexander and others 2008, Arthur and others 2012, Green and others 2010).

Research on regenerating oak has resulted in several prescriptions aimed solely at that goal. However, applying these prescriptions on a stand level and obtaining results found at plot-scale has been problematic. Managers need tools such as shelterwood harvests, thinning, and prescribed burning that can be implemented with biological and economical efficiency. Obtaining results that are predictable has remained elusive for stands located on the Mid- and Southern Cumberland Plateau. We have learned that on more productive sites, such as those on the Plateau escarpment, the increase in light levels following a midstory herbicide treatment is ephemeral. Tweaking of the prescription to target the smaller sugar maple (1 to 3 inches d.b.h.) may allow for a less transient light response as those trees would be removed and would not occupy the newly created growing space. However, removing those stems may change stand density in a manner that would enhance germination and growth of yellow-poplar. On these productive sites, light levels may not be high enough to sustain the yellow-poplar, but their competitiveness under these conditions must be taken into account. Using prescribed fire without stand density reduction was not conducive to increasing light or the number of competitive oaks. Reducing stand density alone on more xeric Plateau tabletop sites increased oak reproduction numbers; red maple also increased. Adding fire did not control the red maple competition. Changes in this prescription may include burning at the end of the dormant season/beginning of the growing season, with the expectancy of more intense fires. The conundrum remains on both mesic and xeric sites, where the intent is to reduce stand density and increase available light in the understory in order to develop competitive oak reproduction without provoking competitive species.

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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 a single and repeated prescribed fires. Forest Ecology and Management. 256: 1021-1030.

Aquilani, S.M.; LeBlanc, D.C.; Morrell, T.E. 2000. Effects of prescribed surface fires on ground- and shrub-nesting neotropical migratory birds in a mature Indiana oak forest, USA. Natural Areas Journal. 20: 317-324.

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(5): 257-266.

Artman, V.L.; Sutherland, E.K.; Downhower, D.F. 2001. Prescribed burning to restore mixed-oak communities in southern Ohio: effects on breeding-bird populations. Conservation Biology. 15: 1423-1434.

Belden, A.C.; Pallardy, S.G. 2009. Successional trends and apparent *Acer saccharum* regeneration failure in an oak-hickory forest in central Missouri, USA. Plant Ecology. 204: 305-322.

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.; Van Lear, D.; Cooper, R. 1999. Using shelterwood harvests and prescribed fire to regenerate oak stands on productive upland sites. Forest Ecology and Management. 113: 125-141.

Cantrell, A.; Wang, Y.; Schweitzer, C.; Greenberg, C. 2013. Short-term response of herpetofaunal to oak regeneration treatments on the Mid-Cumberland Plateau of southern Tennessee. Forest Ecology and Management. 295: 239-247.

Chiang, J.-M.; Arthur, M.A.; Blankenship, B.A. 2005. The effect of prescribed fire on gap fraction in an oak forest understory on the Cumberland Plateau. Journal of the Torrey Botanical Society. 132: 432-441.

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(5-6): 955-978.

Dey, D.C.; Hartman, G. 2005. Returning fire to Ozark highland forest ecosystems: effects on advance reproduction. Forest Ecology and Management. 217: 37-53.

Dey, D.C.; Schweitzer, C.J. [In press]. Timing fire to minimize oak damage in managing oak ecosystems. In: Holley, A.G.; Haywood, J.D.; Connor, K.F., eds. Proceedings of the 17th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-xx. Asheville, NC: US Department of Agriculture Forest Service, Southern Research Station.

Dillaway, D.N.; Stringer, J.W.; Rieske, L.K. 2007. Light availability influences root carbohydrates, and potentially vigor, in white oak advance regeneration. Forest Ecology and Management. 250: 227-233.

Felix, Z.; Wang, Y.; Schweitzer, C. 2010. Effects of experimental canopy manipulation on amphibian egg deposition. Journal of Wildlife Management. 74(3): 496-503.

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

Gardiner, E.S.; Hodges, J.D. 1998. Growth and biomass distribution of cherrybark oak (*Quercus pagoda* Raf.) seedlings as influenced by light availability. Forest Ecology and Management. 108: 127-131.

Gordon, A.M.; Simpson, J.A.; Williams, P.A. 1995. Six-year response of red oak seedlings planted under a shelterwood in central Ontario. Canadian Journal of Forest Research. 25(4): 603-613.

Gottschalk, K.W. 1994. Shade, leaf growth and crown development of *Quercus rubra*, *Quercus velutina*, *Prunus serotina* and *Acer rubrum* seedlings. Tree Physiology. 14: 735-749.

Green, S.R.; Arthur, M.A.; Blankenship, B.A. 2010. Oak and red maple seedling survival and growth following periodic prescribed fire on xeric ridgetops on the Cumberland Plateau. Forest Ecology and Management. 259: 2256-2266.

Harmer, R.; Boswell, R.; Robertson, M. 2005. Survival and growth of tree seedlings in relation to change in the ground flora during natural regeneration of an oak shelterwood. Forestry. 78(1): 21-32.

Hutchinson, T.F.; Sutherland, E.K.; Yaussy, D.A. 2005. Effects of repeated prescribed fires on the structure, composition, and regeneration of mixed-oak forests in Ohio. Forest Ecology and Management. 218: 210-228.

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.

Iverson, L.R.; Prasad, A.M.; Hutchinson, T.D. [and others]. 2004. Fire and thinning in an Ohio oak forest: grid-based analyses of fire behavior, environmental conditions, and tree regeneration across a topographic moisture gradient. 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: 190-197.

Janzen, G.C.; Hodges, J.D. 1987. Development of advanced oak regeneration as influenced by removal of midstory and understory vegetation. Gen. Tech. Rep. SO-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Research Station: 273-278.

Knapp, S.M.; Haas, C.A.; Harpole, D.N.; Kirkpatrick, R.L. 2003. Initial effects of clearcutting and alternative silvicultural practices on terrestrial salamander abundance. Conservation Biology. 17: 752-762.

Lhotka, J.M.; Lowenstein, E.F. 2008. 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.

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.

Loftis, D.L. 1990. A shelterwood method for regenerating red oak in the southern Appalachians. Forest Science. 36(4): 917-929.

Lorimer, C.G.; Chapman, J.W.; Lanbert, 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.

Miller, G.W.; Kochenderfer, J.N.; Fekedulegn, D.B. 2006. Influence of individual reserve trees on nearby reproduction in two-aged Appalachian hardwood stands. Forest Ecology and Management. 224: 241-251.

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. Gen. Tech. Rep. NRS-P-1. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 49-61.

Motsinger, J.R.; Kabrick, J.M.; Dey, D.C. [and others]. 2010. Effect of midstory and understory removal on the establishment and development of natural and artificial pin-oak advance reproduction in bottomland forests. New Forests. 38: 195-213.

Nyland, R.D.; Abrahamson, L.P.; Adams, K.B. 1983. Use of prescribed fire for regenerating red and white oak in New York. In: America's hardwood forests - opportunities unlimited: proceedings of the 1982 Society of American Foresters national convention. Washington, DC: Society of American Foresters: 163-167.

Pallardy, S.G.; Nigh, T.A.; Garrett, H.E. 1988. Changes in forest composition in central Missouri: 1968-1982. American Midland Naturalist. 120: 380-390.

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.

Rochow, J.J. 1972. A vegetational description of a mid-Missouri forest using gradient analysis techniques. American Midland Naturalist. 87: 377-396.

Saab, V.A.; Dudley, J.; Thompson, W.L. 2004. Factors influencing occupancy of nest cavities in recently burned forests. Condor. 106: 20-36.

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.

Signell, S.A.; Abrams, M.D.; Hovis, J.C.; Henry, S.W. 2005. Impact of multiple fires on stand structure and tree regeneration in central Appalachian oak forests. Forest Ecology and Management. 218: 146-158.

Smalley, G.W. 1979. Classification and evaluation of forest sites on the Southern Cumberland Plateau. Gen. Tech. Rep. SO-23. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 59 p.

Smalley, G.W. 1982. Classification and evaluation of forest sites on the Mid-Cumberland Plateau. Gen. Tech. Rep. SO-38. New Orleans: U.S. Department of Agriculture Forest Service, Southern Research Experiment Station. 58 p.

Smith, K.T.; Sutherland, E.K. 1999. Fire-scar formation and compartmentalization in oak. Canadian Journal of Forest Research. 29: 166-171.

Stambaugh, M.; Guyette, R. 2008. Prescribed fire effects on the wood quality of three common oaks in the Ozark region, *Q. coccinea, Q. velutina, Q. alba*. Jefferson City, MO: Missouri Department of Conservation. 23 p.

Stringer, J. 2005. Oak shelterwood: how to apply the system to stimulate oak regeneration. Forest Landowner. 64: 27-29.

Sutton, W.B.; Wang, Y.; Schweitzer, C.J.; Steen, D.A. 2013. Amphibian and reptile response to thinning and prescribed burning in mixed pine-hardwood forests of northwestern Alabama, USA. Forest Ecology and Management. 295: 213-227.

USDA Forest Service. 2003. Final environmental impact statement, forest health and restoration project, National Forests in Alabama, Bankhead National Forest. Manage. Bull. R8-MB-110B. Atlanta: U.S. Department of Agriculture Forest Service, Southern Region. 353 p.

Van Lear, D.H.; Waldrop, T.A. 1989. History, uses, and effects of fire in the Appalachians. Gen. Tech. Rep. SE-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 24 p.

Waldrop, T.A.; White, D.L.; Jones, S.M. 1992. Fire regimes for pine-grassland communities in the southeastern United States. Forest Ecology and Management. 47: 195-210.

Wang, Y.; Schweitzer, C.J.; Lesak, A.A. 2006. Response of avian bark foragers and cavity nesters to the regeneration treatments in the oak-hickory forests of north Alabama. 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: 363-366.

MECHANICAL SITE PREPARATION AND OUST XP® EFFECTS ON STEM BIOMASS IN THREE-YEAR-OLD NUTTALL OAK SEEDLINGS PLANTED ON A FORMER AGRICULTURAL FIELD

Andrew B. Self, Andrew W. Ezell, Dennis Rowe, Emily B. Schultz, John D. Hodges[1]

Abstract-- Mechanical site preparation is frequently proposed to alleviate problematic soil conditions when afforesting retired agricultural fields. Without management of soil problems, any seedlings planted in these areas may exhibit poor growth and survival. Seeding height and groundline diameter are often used to evaluate effects of site preparation methods, but stem biomass may provide a more appropriate assessment of treatment effect in some circumstances. Four mechanical site preparation and two post-plant Oust XP® treatments were utilized in an attempt to evaluate resulting stem biomass differences. Mechanical site preparation treatments included a control, subsoiling, bedding, and combination plowing. A 1-year Oust XP® treatment was applied over one half of treatment areas. A 2-year treatment of Oust XP® was applied on the remaining half. A total of 1,440 bare-root Nuttall oak (*Quercus texana* Buckley) seedlings were planted in February 2008 on Malmaison Wildlife Management Area near Grenada, MS. All sites were of comparable soils and received above average precipitation for the majority of the 3-year duration of the study. Treatment effects on stem biomass were analyzed. Seedling stems in bedded and combination plowed areas exhibited greater woody biomass (210.25 g and 198.62 g, respectively) compared to seedlings in control or subsoiled areas (139.88 g and 118.9 g, respectively). Seedlings in areas treated with 2 years of Oust XP® exhibited greater stem biomass (202.71 g) compared to seedlings in areas treated with the 1-year Oust XP® treatment (131.11 g).

INTRODUCTION

Over the course of the past few decades, the practice of planting hardwood plantations in the Lower Mississippi Alluvial Valley (LMAV) has accounted for the afforestation of several hundred thousand acres of retired agricultural areas (King and Keeland 1999, Stanturf and others 2004). With an estimated 30 million acres of retired agricultural fields expected to undergo afforestation by the year 2040 (Wear and Greis 2002), the development of successful methods for establishing plantations on former agricultural areas cannot be understated.

Often, growth and survival of planted seedlings on former agriculture fields has not been satisfactory, resulting in a low percentage of oaks in established stands (McGee and Loftis 1986, Schoenholtz and others 2001, Stanturf and others 2004, Stanturf and others 2001). The preponderance of these failed afforestation attempts on former agriculture fields indicates a need for greater understanding of proper plantation establishment techniques. Several factors can decrease seedling growth and survival including: soil conditions, planting techniques, seedling quality, and competing vegetation. These problems can be alleviated through proper planting of high-quality seedlings, as well as applying proper silvicultural practices to enhance survival and growth. Notably, planting success on these areas can be improved through the use of herbaceous weed control (HWC) and mechanical site preparation.

Competing vegetation is often the primary cause of oak plantation failures. While both herbaceous and woody competition pose threats to the survival and growth of planted seedlings, herbaceous competition typically poses the greatest threat during the first years of establishment (Peltzer and Kochy 2001). Improved growth and survival of hardwood seedlings treated with broad-spectrum pre-emergent herbicides has been well documented (Ezell and Catchot 1997, Ezell and Hodges 2002, Ezell and others 2007, Groninger and others 2004, Woeste and others 2005).

One of the primary reasons to use mechanical site preparation is for alleviation of problematic soil conditions. In addition to decreasing soil strength and consequently lessening the difficulty of root penetration in the soil, nutrient availability is increased due to improved accessibility and consolidation (Fisher and

[1]Assistant Extension Professor, Mississippi State University, Department of Forestry, Grenada, MS 38901; Professor, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; Professor, Mississippi State University, Experimental Statistics Unit, Mississippi State, MS 39762; Professor, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; and Professor Emeritus, Ashland, MS 39576.

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Binkley 2000, Kabrick and others 2005, Miller 1993, Morris and Lowery 1988, Russell and others 1997). Variation in water and nutrient availability can lead to variable seedling growth (Lutze and Gifford 1998, Poorter and Nagal 2000, Reich 2002, Ryser and Lambers 1995, Volin and Reich 1996). Differences in the availability of soil nutrients and changes in microsite in areas with different resource availability can result in stem growth differences (Kozlowski and Pallardy 1997, Van Hees and Clerkx 2003). Thus, increases in seedling growth could be a response to HWC and mechanical soil treatment. Shoot growth is of particular interest in determining how young trees respond to HWC and mechanical site preparation.

Researchers often use seedling parameters such as height and diameter in attempts to quantify seedling response to chemical and cultural treatments. Stem biomass is an often-overlooked metric that can provide useful insight into treatment effects on seedling success. Biomass estimation can be extremely useful in ascertaining seedling performance under different treatment regimes. Treatment influences that might remain undetected when utilizing more easily measured parameters may be recognized through biomass determination.

OBJECTIVES

The objectives of this study were: (1) to evaluate the effects of HWC on stem biomass of 3-year-old Nuttall oak (*Quercus texana* Buckley) seedlings; and (2) to evaluate the effects of subsoiling, bedding, and combination plowing on stem biomass of 3-year-old Nuttall oak seedlings.

MATERIALS AND METHODS

Site Description

The study area is located on Malmaison Wildlife Management Area (WMA) approximately 14 miles northeast of Greenwood, MS (90.0531° W, 33.6876° N) in Grenada County. The site was formerly used in row-crop production and was retired from agricultural production in the late 1990s. It was maintained as an opening for wildlife through mowing and disking from agricultural retirement until the initiation of this study. Soils are Falaya and Collins silt loams, and average yearly precipitation is 53.8 inches (NOAA 2011). Soil tests indicated that onsite pH ranges from 6.3 to 7.0. Using the Baker and Broadfoot method for site evaluation, site index (base age 50) for Nuttall oak is 93.

At initiation of this project, dominant onsite herbaceous species were ryegrass (*Lolium* spp.), bermudagrass [*Cynodon dactylon* (L.) Pers], Brazilian vervain (*Verbena brasiliensis* Vell.), and Carolina horsenettle *(Solanum carolinense* L). Forty other herbaceous species occurred in small quantities. Cumulative herbaceous coverage by these species was 100 percent.

Experimental Design

This experiment utilized a randomized complete block design. Three blocks with eight plots per block were established. Each plot received a randomly assigned combination of mechanical site preparation and HWC treatment. The experimental unit was a plot with its unique combination of site preparation and HWC treatments. The response variable was third-year oven-dried weight of stems.

Mechanical Site Preparation and HWC Treatments

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 1-year application and a 2-year application of Oust XP®. Both treatments were applied in 5-foot-wide bands using a rate of 2 ounces of product per acre and were applied over the top of seedlings prior to budbreak. The 1-year Oust XP® application was applied during March 2008. The 2-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 (GPA).

Seedling Establishment

Nuttall oak was chosen for use in the biomass study because it is known for fast growth. Expectations were that there would be an abundant amount of woody biomass for use in analyses. Seedlings were lifted mid-January 2008, and purchased under specifications requiring 1-0 seedlings to be of overall vigorous appearance and have relatively intact root systems. Seedling parameters dictated that seedling stems be 18- to 24-inches tall and possess root systems 8- to 10-inches long with a minimum of eight first-order lateral roots (FOLRs). A total of 480 seedlings were planted at root collar depth using a 10-foot spacing during February 2008 by Mississippi State University personnel.

Stem Biomass Measurements
During November-December 2010, individual trees were sampled for stem biomass production. A total of seven Nuttall oak trees were selected for sampling from each treatment combination (site preparation/HWC) in each block. Stems from a total of 168 trees were collected. Stems were clipped at ground level and tagged by block and tree number. Leaves were removed, and each stem was cut into lengths that would fit into paper sacks labeled to identify each seedling. Stems were then weighed and dried at 212 °F for 48 hours in a desiccating oven before reweighing. The stems were then dried for an additional 24 hours to check for further weight loss. Final weights were recorded to 0.1g after sack weight was subtracted.

Data Analysis
All statistical analyses were performed using Statistical Analysis System version 9.2 (SAS Institute, Cary, NC). Proc Univariate was used for univariate analysis of biomass weight response. Analyses indicated some skewness which was corrected by taking the log of biomass weights. Outlier analysis using Studentized residual and Cook's D tests for outliers indicated that five sample trees were outliers. These trees were eliminated from further analyses. General Linear Modeling (GLM) and Analysis of Variance (ANOVA) were used to test for main effects and interactions, and to estimate least square means (LSMEANS). The LSMEANS LINES option was used to identify differences. Differences were considered significant at $\alpha = 0.05$. While transformed biomass data were used for analyses, actual means are presented for ease of interpretation.

RESULTS AND DISCUSSION

Stem Biomass by Site Preparation Treatment
ANOVA detected a significant main effect difference ($p = 0.0029$, $F = 4.87$) among site preparation treatments for stem biomass. Seedling stems in bedded and combination plowed areas exhibited greater woody biomass (210.25 g and 198.62 g, respectively) compared to seedlings in control or subsoiled areas (139.88 g and 118.9 g, respectively) (table 1).

Table 1--Stem biomass of Nuttall oak seedlings at Malmaison WMA, MS, December 2010 by site preparation treatment

Mechanical	Weight
	grams
Bedded	210.25a[a]
Combination plowed	198.62a
Flat planted (no mechanical)	139.88b
Subsoiled	118.91b

[a]Values followed by different letters are significantly different at $\alpha = 0.05$.

Typically, some separation in stem biomass would be expected between seedlings in control and subsoiled treatments. However, it is possible that excellent precipitation levels during the first two growing seasons of this study (19.4 inch surplus over average for the 2 years) negated the inherent enhanced growth potential of the subsoiling treatment. Greater stem biomass in seedlings planted in bedded and combination plowed areas compared to control or subsoiled areas indicates that over the span of this study, the more intensive treatments consistently produced seedlings with greater biomass.

Stem Biomass by HWC Treatment
ANOVA detected a significant main effect difference ($p = 0.0005$, $F = 12.56$) among HWC treatments for stem biomass. Seedlings in areas treated with 2-years of Oust XP® exhibited significantly greater stem biomass (202.71 g) compared to seedlings in areas treated with the 1-year Oust XP® treatment (131.11 g) (table 2). Greater stem biomass in 2-year Oust XP® areas compared to 1-year Oust XP® areas is the result of a greater "free-to-grow" period of time. During this period, seedlings did not suffer the adverse

effects of herbaceous competition and resources that would have otherwise been lost to competitive stress were allocated to biomass growth.

Table 2--Stem biomass of Nuttall oak seedlings at Malmaison WMA, MS, December 2010 by pre-emergent Oust XP treatment

Pre-emergent treatment	Weight
	grams
One-year	131.11b[a]
Two-year	202.71a

[a]Values followed by different letters are significantly different at $\alpha = 0.05$.

SUMMARY AND CONCLUSIONS

The intent of this study was to determine if seedling biomass was affected by mechanical site preparation or HWC treatments. Mechanical site preparation had substantial impacts on stem biomass growth of Nuttall oak seedlings. Seedlings in bedded and combination plowed treatment areas exhibited significantly more stem biomass compared to those in control or subsoiled areas, and 2-year Oust XP® treatment areas exhibited greater biomass compared to 1-year Oust XP® treatment areas.

No statistical difference was detected in stem biomass accumulation between bedded and combination plowed areas. With no statistical advantage, higher combination plowing costs would preclude its use under conditions similar to those encountered in this study. Thus, based on this study, bedding would be the mechanical site preparation treatment of choice. However, subsoiling should not be discounted as a possible mechanical treatment. In areas where substantial resistant layers are present in the soil, subsoiling offers benefits in the amelioration of compaction problems. Additionally, subsoiling is less expensive than bedding or combination plowing and might be the treatment of choice in some situations where more intensive treatments are not possible (e.g., expense, erosion concerns, contractor availability, etc.). While seedling biomass was greater in 2-year Oust XP® areas compared to biomass in 1-year Oust XP® areas, the greater expense of the 2-year Oust XP® treatment might be cost prohibitive in many management efforts.

LITERATURE CITED

Ezell, A.W.; Catchot, A.L., Jr. 1997. Competition control for hardwood plantation establishment. 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: 42-43.

Ezell, A.W.; Hodges, J.D. 2002. Herbaceous weed control improves survival of planted Shumard oak seedlings. 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: 273-275.

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.

Fisher, R.F.; Binkley, D. 2000. Ecology and management of forest soils. 3rd ed. New York: John Wiley & Sons. 489 p.

Groninger, J.W.; Baer, S.G.; Babassana, D.A.; Allen, D.H. 2004. Planted green ash (*Fraxinus pennsylvanica* Marsh.) and herbaceous vegetation responses to initial competition control during the first 3 years of afforestation. Forest Ecology and Management. 189: 161-170.

Kabrick, J.M.; Dey, D.C.; Van Sambeek, J.W. [and others]. 2005. Soil properties and growth of swamp white oak and pin oak on bedded soils in the lower Missouri River floodplain. Forest Ecology and Management. 204: 315-327.

King, S.L.; Keeland, B.D. 1999. Evaluation of reforestation in the Lower Mississippi River Alluvial Valley. Restoration Ecology. 7: 348-359.

Kozlowski, T.T.; Pallardy, S.G. 1997. Physiology of woody plants. (2d ed.). San Diego: Academic Press. 411 p.

Lutze, J.L.; Gifford, R.M. 1998. Acquisition and allocation of carbon and nitrogen by *Danthonia richardsonii* in response to restricted nitrogen supply and CO_2 enrichment. Plant Cell Environment. 21: 1133-1141.

McGee, C.E; Loftis, D.L. 1986. Planted oaks perform poorly in North Carolina and Tennessee. Northern Journal of Applied Forestry. 3(3): 114-115.

Miller, J.H. 1993. Oak plantation establishment using mechanical, burning, and herbicide treatments. In: Loftis, D.; McGee, C.E., eds. Oak regeneration, serious problems, practical recommendations. Symposium proceedings. Gen. Tech. Rep. SE-84. U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 264-289.

Morris, L.A.; Lowery, R.F. 1988. Influence of site preparation on soil conditions affecting seedling establishment and early growth. Southern Journal of Applied Forestry. 12: 170-178.

NOAA [National Oceanic and Atmospheric Administration]. 2011. 40-year (1971-2010) precipitation totals. U.S. Department of Commerce National Climatic Data Center Web. http://www.noaa.gov. [Date accessed: February 18, 2011].

Peltzer, D.A.; Kochy, M. 2001. Competitive effects of grasses and woody plants in mixed-grass prairie. Journal of Ecology. 89(4): 519-527.

Poorter, L.; Nagel, O. 2000. The role of biomass allocation in the growth response of plants to different levels of light, CO_2, nutrients and water: a quantitative review. Australian Journal of Plant Physiology. 27: 595-607.

Reich, P.B. 2002. Root-shoot relations: optimality in acclimation and adaptation or the “emperor’s new clothes”? In: Waisel, Y.; Eshel, A.; Kafkafi, U., eds. Plant roots: the hidden half. 3rd ed. New York: Marcel Dekker: 205-220.

Russell, D.R., Jr.; Hodges, J.D.; Ezell, A.W. 1997. An evaluation of hardwood reforestation methods on previously farmed lands in central Alabama. 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: 272-276.

Ryser, P.; Lambers, H. 1995. Root and leaf attributes accounting for the performance of fast- and slow-growing grasses at different nutrient supply. Plant Soils. 170: 251-265.

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. Wetlands. 21(4): 602-613.

Stanturf, J.A.; Conner, W.H.; Gardiner, E.S. [and others]. 2004. Recognizing and overcoming difficult site conditions for afforestation of bottomland hardwoods. Ecological Restoration. 22(3): 183-193.

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.

Van Hees, A.F.M.; Clerkx, A.P.P.M. 2003. Shading and root-shoot relations in saplings of silver birch, pedunculate oak and beech. Forest Ecology and Management. 176: 439-448.

Volin, J.C.; Reich, P.B. 1996. Interaction of carbon dioxide and ozone on C_3 and C_4 grasses and trees under contrasting nutrient supply. Physiology of Plants. 97: 674-684.

Wear, D.N.; Greis, J.P. 2002. Southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 635 p.

Woeste, K.E.; Seifert, J.R.; Selig, M.F. 2005. Evaluation of four herbicides and tillage for weed control on third year growth of tree seedlings. Weed Science. 53: 331-336.

SPECIES COMPOSITION OF DEVELOPING CENTRAL APPALACHIAN HARDWOOD STANDS FOLLOWING CLEARCUTTING

Lance A. Vickers and Thomas R. Fox[1]

Abstract--This study examined the species composition of 47 paired stands on submesic sites on the Appalachian Plateau of West Virginia. Paired stands consisted of a mature stand adjacent to a young clearcut that was < 20 years old. The species composition in the mature stands was compared to that of the upper canopy (dominant and codominant) in the clearcuts. The objective of this comparison was to determine if there was evidence of potentially lasting shifts in species composition resulting from clearcutting. This objective was addressed through three research questions related to common regeneration concerns in the region: (1) is there evidence of a shift towards more mesophytic species? (2) Is there evidence of an increase in red maple (*Acer rubrum* L.); and (3) is there evidence of a decrease in oak (*Quercus* spp.)? There were significant differences in species composition between the clearcuts and mature stands. These differences were largely due to increases in fast-growing, shade-intolerant pioneer species {e.g., black locust (*Robinia pseudoacacia* L.), pin cherry (*Prunus pensylvanica* L.), sassafras [*Sassafras albidum* (Nutt.) Nees], etc.}, black cherry (*Prunus serotina* Ehrh.), and yellow-poplar (*Liriodendron tulipifera* L.). Significant differences were not found for mesophytic species, red maple, or oaks. The results of this comparison suggest that future species composition of the young clearcuts may differ only slightly from previous rotations.

INTRODUCTION

Appalachian forests are among the most diverse in the United States. Intricate geologic, climatic, and anthropogenic influences have created mixed stands of species with vastly different characteristics (Braun 1950, Fenneman 1938, Yarnell 1998). More than 50 tree species may be present in these stands, and over 20 of these may be commercial (Miller and Kochenderfer 1998, Smith 1995). In many cases, the current incarnation of these forests originated early in the 20th century, following repeated, landscape-level exploitive disturbance (Yarnell 1998). As a result, vast acreages throughout the Appalachians have reached or are approaching typical rotation ages (Oswalt and Turner 2009). This makes regeneration an important and relevant topic for foresters in the Appalachians.

The diversity and complexity of Appalachian forests create potential for many silvicultural systems to be adopted to regenerate these maturing forests depending on management objectives. However, it also presents challenges for foresters interested in regenerating oak-dominated (*Quercus spp.*) forests back to a similar state (Loftis and McGee 1993). Due to changing disturbance regimes, forest managers and researchers have been concerned about the potential for drastic shifts in species composition from existing conditions following regeneration harvests. It is believed that the absence of late-rotation disturbance, particularly fire, leads to the mesophication of productive sites (Abrams 1992, Nowacki and Abrams 2008). This trend could favor a mixed-mesophytic species composition over the currently oak-dominated forests in parts of the Appalachians.

Increasingly, shifts in species composition of oak-dominated stands towards more mesophytic species and red maple (*Acer rubrum* L.) are being reported and predicted across the Appalachian landscape, particularly on productive sites (Abrams and Downs 1990, Fei and Steiner 2007, Schuler 2004, Schuler and Gillespie 2000). Such shifts could have far reaching consequences for timber markets and ecosystem services and should be considered prior to prescribing regeneration techniques on mature forests throughout the region (Duffy and Meier 1992, Homyack and Haas 2009, McShea and others 2007, Miller and Kochenderfer 1998).

Many acres of public land were regenerated using the clearcut method in the late 20th century, but public scrutiny eventually lead to its decline on public lands. Nonetheless, clearcutting remains a widely practiced regeneration technique on managed private forests. The existing literature on the effects of clearcutting on species composition in hardwood forests is considerable. Clearcutting in central hardwoods can often regenerate fully stocked stands of commercial species (Roach and Gingrich 1968, Sander and Clark 1971). The resulting composition is typically dominated initially by early successional species such as yellow-poplar (*Liriodendron tulipifera* L.), black locust (*Robinia pseudoacacia* L.), sweet birch

[1]Graduate Research Assistant and Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061.

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(*Betula lenta* L.), and black cherry (*Prunus serotina* Ehrh.) (Hilt 1985a, McGee and Hooper 1975, Trimble 1973,). Red maple is also typically competitive following clearcutting in the Appalachians (Tift and Fajvan 1999). The potential for inadequate regeneration of oak following clearcutting, particularly on highly productive sites, has been the focus of extensive research throughout the Appalachians in the central hardwood region (Loftis and McGee 1993).

Many of the aforementioned studies on clearcutting were conducted on stands that were likely exposed to a disturbance regime that no longer occurs (Abrams 1992). The objective of this study was to determine if there was evidence of potentially lasting shifts in species composition resulting from clearcutting Central Appalachian stands under the modern landscape disturbance regime. Three research questions were adopted to address that objective: (1) is there evidence of a shift towards more mesophytic species? (2) Is there evidence of an increase in red maple (*Acer rubrum* L.); and (3) is there evidence of a decrease in oak (*Quercus* spp.)?

METHODS

A paired stand approach was used to address the research questions of this study. Paired stands consisted of a mature stand (≥ 70 years old) in the understory re-initiation stage located adjacent to a young clearcut of similar site characteristics that had reached crown closure but was still in the early stages of stem exclusion (5 to 20 years old). This approach assumed that the two stands were once contiguous with similar composition and productivity. It was further assumed that the mature stand, if harvested, would regenerate similarly to its paired clearcut. A total of 47 paired stands were located on the Appalachian Plateau in West Virginia (Fenneman 1938). Paired stands were located across six counties including Fayette, Greenbrier, Nicholas, Randolph, Tucker, and Webster (fig. 1). The paired stands were owned and managed by a variety of ownership groups including public and private entities. According to landowner records, the paired stands had been relatively free from known disturbance for several years before and since the regeneration harvest that created the clearcuts. The young clearcuts had an average age of 13 years since harvest. Slope, aspect, and landscape position were documented to indirectly estimate site index and help ensure similar site conditions existed between the paired mature and clearcut stands (Meiners and others 1984). Differences in upland oak site index (base-age 50 years) within paired stands ranged from 0 to 10 feet, with an average of 3 feet. The average site index across all stands was 77 feet. All paired stands in this report were categorized as submesic in terms of expected moisture availability using indicator species as proposed by McNab and others (2003).

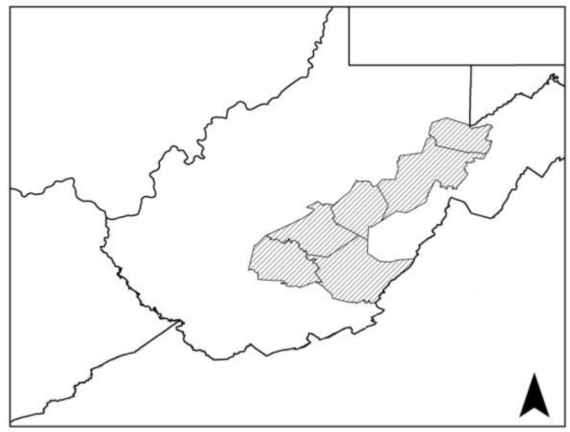

Figure 1--Location map of the study area on the Appalachian Plateau in West Virginia. Shaded areas represent counties in which sample stands were located.

Paired stands were sampled in 2008 from May to September. The mature stands were sampled using fixed area plots (0.01 acre) at a density of 1 plot per acre (maximum of 20 plots per stand). Within a stand, the initial plot location was randomly determined, and all remaining plots locations were determined using a systematic grid. The species and diameter at breast height (d.b.h.) of all stems ≥ 2 inches d.b.h. within the plot was recorded. Variable radius sampling points were also established at each plot location to measure the basal area of all stems ≥ 2 inches d.b.h. using a 10 BAF prism. The clearcuts were sampled using fixed area plots (0.001 acre) located in the same manner as the mature stands. Some of the older clearcuts were sampled using larger plots (0.01 acre). The species, stem origin (seed or sprout), and crown class (dominant, codominant, intermediate, suppressed) of all stems within the clearcut plot were recorded. Multiple stems originating from a single stump were tallied as individuals.

Table 1--Species groups used for this study on the Appalachian Plateau in West Virginia

Group	Species included
Black cherry	black cherry (*Prunus serotina* Ehrh.)
Mesophytic	ash (*Fraxinus* spp.), basswood (*Tilia* spp.), cucumbertree (*Magnolia acuminata* L.), Fraser magnolia (*Magnolia fraseri* Walt.), yellow buckeye (*Aesculus flava* Ait.), yellow birch (*Betula alleghaniensis* Britton)
Midstory	Am. chestnut [*Castanea dentata* (Marsh) Borkh.], Am. beech (*Fagus grandifolia* Ehrh.), Am. holly (*Ilex opaca* Ait.), blackgum (*Nyssa sylvatica* Marsh.) dogwood (*Cornus* spp.), e. hophornbeam [*Ostrya virginiana* (Mill.) K. Koch.] serviceberry (*Amelanchier* spp.), sourwood [*Oxydendrum arboreum* (L.) DC.], striped maple (*Acer pensylvanicum* L.)
Oaks	black oak (*Quercus velutina* Lam.), chestnut oak (*Quercus prinus* L.), northern red oak (*Quercus rubra* L.), scarlet oak (*Quercus coccinea* Muenchh.), white oak (*Quercus alba* L.)
Pioneer	Am. sycamore (*Platanus occidentals* L.), bigtooth aspen (*Populus grandidentata* Michx.), black locust (*Robinia pseudoacacia* L.), pin cherry (*Prunus pensylvanica* L.f.), sassafras [*Sassafras albidum* (Nutt.) Ness.], sweet birch (*Betula lenta* L.)
Red maple	red maple (*Acer rubrum* L.)
Sugar maple	sugar maple (*Acer saccharum* Marsh.)
Yellow-poplar	yellow-poplar (*Liriodendron tulipifera* L.)
Others	e. white pine (*Pinus strobus* L.), hemlock (*Tsuga* spp.), hickory (*Carya* spp.)

Given the magnitude of species that were recorded, nine species groups were created to facilitate data analysis. Species and genera that occupy reasonably similar stand structural and functional roles were grouped together, with the exception of a few individual species (table 1). The species groupings used in this study were: (1) black cherry; (2) mesophytic; (3) midstory; (4) oaks; (5) pioneer; (6) red maple; (7) sugar maple (*A. saccharum* Marsh.); (8) yellow-poplar; and (9) other.

The species composition of all stems ≥ 2 inches d.b.h. in the mature stands was compared to the upper canopy (dominant and codominant stems) species composition within the clearcuts. The upper canopy of young stands can provide insight into future species composition (Ward and Stephens 1994). Crown class was not recorded in the mature stands, thus a comparison of species composition by crown class between the mature and clearcut stands was not possible. Species composition was expressed as percent stems per acre so that an equitable variable could be compared between the mature and clearcut stands which had very different stem densities and stand structures.

The data did not meet the assumption of normally distributed errors required for most parametric statistics, so nonparametric statistics were used instead. The permutation test for matched pairs (PTMP) in Blossom version W2008.04.02 statistical software was used to test for overall differences in species composition between the mature and young clearcuts (Cade and Richards 2005, Cade and Richards 2008, Mielke and Berry 2001). The Wilcoxon Signed Rank test was used to test for differences between the mature stands and young clearcuts for individual species groups using the UNIVARIATE procedure in SAS® version 9.2 statistical software (Ott and Longnecker 2001, SAS 2007). One-tailed tests were used for the comparisons specifically outlined by the three research questions mentioned earlier. Two-tailed tests were used for all other comparisons.

RESULTS

The basal area of all stems ≥ 2 inches d.b.h. in the mature stands averaged 114 square feet per acre (table 2). The mature stands inventoried in this study, and most mature stands across the Appalachians, are generally described as oak-dominated, yet foresters have long realized that most stands contain relatively few but large oak stems. Indeed, oaks made up the largest proportion of basal area in the mature stands (30 percent) but only 14 percent of all stems. The oak-dominated description is reasonable for basal area, but in terms of stem density, another picture emerges. Maples (red and sugar combined) often make up the largest proportion of stems in mature Appalachian hardwood stands, as was found in this study. Maples also had the second highest basal area in the mature stands.

Table 2--Mean composition of stems ≥ 2 inches d.b.h. in 47 mature stands sampled on the Appalachian Plateau in West Virginia

Species group	Basal area (SE ±)	Density (SE ±)
	feet^2/acre	*stems/acre*
Black cherry	08 (3)	15 (4)
Mesophytic	17 (2)	59 (7)
Midstory	03 (1)	27 (5)
Oaks	34 (4)	49 (8)
Pioneer	04 (1)	20 (4)
Red maple	14 (2)	57 (8)
Sugar maple	15 (3)	64 (9)
Yellow-poplar	11 (2)	21 (5)
Other	08 (2)	29 (7)
Total	114	341

Adequate regeneration occurred in the clearcuts with an average 4,637 stems per acre (table 3). Shade-intolerant black cherry, pioneer, and yellow-poplar collectively made up about half of all stems in the clearcuts. Maples were the most common genera in the clearcuts, but most of the stems were in the lower canopy (intermediate and suppressed) of the clearcuts, particularly those of sugar maple. Approximately one-third of all stems in the clearcut were in the upper canopy. Pioneer species were the most numerous in the upper canopy (approximately 25 percent) followed by the red maple, yellow-poplar, black cherry and mesophytic species groups which each made up about 15 percent of all upper canopy stems. Oaks made up about 9 percent of the upper canopy stems. The majority of the stems in the midstory group were already occupying the lower canopy of the clearcuts.

The permutation tests for matched pairs indicated that there were significant differences in species composition between the upper

Table 3--Mean species composition of 47 clearcut stands sampled on the Appalachian Plateau in West Virginia. Lower canopy values are for stems classified in suppressed and intermediate crown positions. Upper canopy values are for stems classified in codominant and dominant crown positions

	-----------Density (SE ±)-------		
Species group	Lower canopy	Upper canopy	Total
	---------*stems per acre*--------		
Black cherry	380 (220)	250 (100)	630
Mesophytic	334 (48)	239 (96)	572
Midstory	398 (64)	98 (23)	496
Oaks	194 (29)	156 (33)	350
Pioneer	415 (57)	442 (97)	857
Red maple	391 (79)	264 (50)	655
Sugar maple	441 (76)	38 (9)	479
Yellow-poplar	325 (58)	223 (38)	549
Other	28 (6)	22 (6)	50
Total	2,906	1,731	4,637

canopy of the clearcuts and the mature stands (p-value < 0.0001). It appears that most of the difference between the mature and clearcut stands was the result of increasing shade-intolerant species. This increase in shade-intolerants was primarily at the expense of sugar maple which was significantly less prevalent in the upper canopy of the clearcuts compared to the mature stands (table 4). The results specifically related to the research questions in this study were surprising. There were no significant differences found for the mesophytic, red maple, or oak species groups.

About 75 percent of the upper canopy red maples and oaks were of sprout origin (table 5). Sprout origin reproduction was also a considerable component of upper canopy reproduction for sugar maple and the mesophytic species group. At least 60 percent of the upper canopy stems of black cherry, pioneer, and yellow-poplar in the regenerating clearcut stands were seemingly of seed origin. These species produce ample seed regularly and some, particularly yellow-poplar and pin cherry, can remain viable for several years in the forest floor (Burns and Honkala 1990).

Table 4--Species composition comparison across 47 paired stands sampled on the Appalachian Plateau in West Virginia. Mean values were calculated from individual stand proportions for each species group. P-values < 0.05 indicate significant differences between mature and clearcut stands as calculated by the Wilcoxon Signed Rank test. Species composition values for the mature stands consider all stems ≥ 2 inches d.b.h. Species composition values for the clearcut stands consider only dominant and codominant stems. Statistical tests were not conducted on the "other" species group

Species Group	Mature	Clearcut	P-Value
	-----(*% stems/acre*)----		
Black cherry	4	14	0.0197
Mesophytic	17	14	0.9988
Midstory	8	56	0.1390
Oaks	14	9	0.0655
Pioneer	6	26	<0.0001
Red maple	17	15	0.5890
Sugar maple	19	2	<0.0001
Yellow-poplar	6	13	0.0004
Other	9	1	-
Total	100	100	-

Table 5--Mean proportion of stems from sprout origin in the upper canopy of 47 clearcut stands on the Appalachian Plateau in West Virginia

Species group	Sprout origin stems
	% stems/acre
Black cherry	40
Mesophytic	68
Midstory	45
Oaks	74
Pioneer	17
Red maple	81
Sugar maple	66
Yellow-poplar	25
Other	-

DISCUSSION

The species composition of clearcuts in this study was similar to other studies reported in the literature (Beck and Hooper 1986, Hilt 1985a, Trimble 1973). It appears that under the modern disturbance regime of little late-rotation disturbance, successional trends can still be reset by providing opportunities for shade-tolerant and mesophytic species to be outcompeted by more aggressive shade-intolerants immediately following clearcutting. An exception may be red maple, which is typically competitive following disturbance across a gradient of site productivity, particularly from stump-sprouts (Hilt 1985a, Loftis 1989, Tift and Fajvan 1999).

No significant changes in species composition were found for red maple in this study. This was not expected, as red maple is widely recognized as a species increasing in abundance and importance throughout its natural range (Fei and Steiner 2007, but see Oswalt and Turner 2009). In addition to reduced fire frequency (Abrams 1992), increasing red maple abundance in the Appalachians has also been attributed to an increased use of diameter-limit and other partial harvesting systems (Deluca and others 2009, Fajvan and others 1998, Kenefic and Nyland 2006). The results of this study suggest that clearcutting in these stands has not promoted an expansion of red maple, at least not initially. Given the shade tolerance and density of red maple, it is possible that it will gradually make up a larger proportion of competitive stems as these young clearcuts continue to develop, but it is not a foregone conclusion (Oliver 1978).

The results for oak regeneration in this study were somewhat more optimistic than reports from the southern Appalachians (Beck and Hooper 1986), but were in line with Hilt (1985a), who found that oaks made up approximately 30 percent of the upper canopy on medium quality sites and about 10 percent on good quality sites in clearcuts across Indiana, Kentucky, and Ohio.

The competitive ability of the oak stems through the remainder of stem exclusion is not certain (Beck and Hooper 1986, Johnson and others 2009, Loftis 1989). Ward and Stephens (1994) found that between 45 and 68 percent of upper canopy northern red oaks at age 25 remained in the upper canopy at age 55 in southern New England. Oaks can persist and eventually become more competitive as clearcuts mature, but additional management activities such as precommercial release treatments are often suggested to improve the survival of existing upper canopy oak stems during the stem-exclusion stage (Elliot and others 1997, Morrissey and others 2008, Oliver 1978, Roach and Gingrich 1968). While such treatments have shown promise (Schuler 2008, Ward 2009), their success is not guaranteed (Heitzman and Nyland 1991, Smith and Lamson 1983, Trimble 1974).

Clearcutting should be used with caution on the most productive sites (Brose and Van Lear 1998, Loftis 1990, Sander 1979). However, there is a growing body of research that indicates greater success of oak regeneration following clearcutting compared to other harvest methods on many sites (Atwood and others 2011, Groninger and Long 2008, Morrissey and others 2008). Kabrick and others (2008) found that in Missouri, red oaks only regenerated successfully following clearcutting compared to single-tree selection, group selection, and single-tree/group selection combination systems. Partial-harvesting practices, including deferment and shelterwood harvests, reduce stump-sprouting of upland oak compared to clearcutting (Atwood and others 2009, Dey and others 2008). Large oak advanced reproduction in the mature stands in this study was well below that recommended by Sander and others (1976) for successful oak regeneration. Without adequate large advance reproduction, oaks are often less successful at regenerating except on sites of poorer quality (Bey 1964, Ross and others 1986, Sander 1972). The high proportion of oaks from sprout origin in the upper canopy of the clearcuts in this study reiterates the importance of stump-sprouting in oak, particularly in the Appalachians (Cook and others 1998, Johnson and others 2009).

Muller (1983) found that 35 years after clearcutting there were no significant differences in species composition on two adjacent watersheds on the Cumberland Plateau in southeastern Kentucky. It is possible that the clearcuts in this study will eventually be similar in composition and structure to that of the mature stands as they continue to develop (Oliver and Larson 1996). The pioneer group, which was composed of short-lived species, is expected to be only a minor component later in the rotation of these clearcuts. However, unlike more xeric sites where sporadic drought has been shown to drastically reduce temporal increases in yellow-poplar following clearcutting, the larger proportion of yellow-poplar on these submesic sites could be a lasting effect (Hilt 1985b, Morrissey and others 2008).

CONCLUSIONS

Clearcutting on submesic sites on the Appalachian Plateau in West Virginia successfully regenerated stands with numerous commercial species. The results of this study suggest that the future species composition of stands regenerating following clearcut harvests may differ only slightly from previous rotations. The current dominance of pioneer species is expected to be short-lived, but a larger presence of yellow-poplar may remain throughout the rotation. Gradual increases in the proportion of more shade-tolerant maples and mesophytic species may occur during later stages of stand development. The future for oaks on these submesic sites is not certain, but it appears likely that oaks will maintain a role as these clearcuts continue to develop. The results of this study do not provide evidence of forest-type shifts towards mesophytic species, increasing red maple abundance, or decreasing oak abundance in young clearcuts on submesic sites on the Appalachian Plateau of West Virginia.

LITERATURE CITED

Abrams, M.D. 1992. Fire and the development of oak forests. Bioscience. 42(5): 346-353.

Abrams, M.D.; Downs. J.A. 1990. Successional replacement of old-growth white oak by mixed mesophytic hardwoods in southwestern Pennsylvania. Canadian Journal of Forest Research. 20: 1864-1870.

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.

Atwood, C.J.; Fox, T.R.; Loftis, D.L. 2011. Effects of various silvicultural systems on regeneration in mixed hardwood stands of the southern Appalachians. Journal of Sustainable Forestry. 30: 419-440.

Beck, D.E.; Hooper, R.M. 1986. Development of a southern Appalachian hardwood stand after clearcutting. Southern Journal of Applied Forestry. 10: 168-72.

Bey, C.F. 1964. Advance oak reproduction grows fast after clear cutting. Journal of Forestry. 62: 339-340.

Braun, E.L. 1950. Deciduous forests of eastern North America. Philadelphia: The Blakiston Co. 596 p.

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.

Burns, R.M; Honkala, B.H., tech. cords. 1990. Silvics of North America: Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture. 877 p. Vol. 2.

Cade, B.S.; Richards, J.D. 2005. User manual for Blossom statistical software. U.S. Geological Survey open-file Rep. 2005-1353. Fort Collins, CO: U.S. Geological Survey, Fort Collins Science Center. 124 p.

Cade, B.S.; Richards, J.D. 2008. Blossom version W2008.04.02. Fort Collins, CO: U.S. Geological Survey, Fort Collins Science Center.

Cook, J.E.; Sharik, T.L.; Smith, D.W. 1998. Oak regeneration in the southern Appalachians: potential, problems, and possible solutions. Southern Journal of Applied Forestry. 22(1): 11-18.

Deluca, T.; Fajvan, M.A.; Miller, G. 2009. Diameter-limit harvesting: effects of residual trees on regeneration dynamics in Appalachian hardwoods. Northern Journal of Applied Forestry. 26(2): 52-60.

Dey, D.C.; Jensen, R.G.; Wallendorf, M.J. 2008. Single-tree harvesting reduces 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. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 26-37.

Duffy, D.C.; Meier, A.J. 1992. Do Appalachian herbaceous understories ever recover from clearcutting? Conservation Biology. 6(2): 196-201.

Elliot, K.J.; Boring, L.R.; Swank, W.T.; Haines, B.R. 1997. Successional changes in plant species diversity and composition after clearcutting a southern Appalachian watershed. Forest Ecology and Management. 92: 67-85.

Fajvan, M.A.; Grushecky, S.T.; Hasler, C.C. 1998. The effects of harvesting practices on West Virginia's wood supply. Journal of Forestry. 96(5): 33-39.

Fei, S.; Steiner, K.C. 2007. Evidence for increasing red maple abundance in the eastern United States. Forest Science. 53(4): 473-477.

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

Groninger, J.W; Long, M.A. 2008. Oak ecosystem management considerations for central hardwoods stands arising from silvicultural clearcutting. Northern Journal of Applied Forestry. 25(4): 173-179.

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.

Hilt, D.E. 1985a. Species composition of young central hardwood stands that develop after clearcutting. In: Dawson, J.O.; Majerus, K.A., eds. Proceedings of the 5th central hardwood forest conference. Society of American Foresters Publ. 85-05. Urbana-Champaign, IL: University of Illinois, Department of Forestry: 11-14.

Hilt, D.E. 1985b. Where has all my yellow-poplar gone? Northern Journal of Applied Forestry. 2: 67-69.

Homyack, J.A.; Haas, C.A. 2009. Long-term effects of experimental forest harvesting on abundance and reproductive demography of terrestrial salamanders. Biological Conservation. 142: 110-121.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2009. The ecology and silviculture of oaks. 2d ed. Cambridge, MA: CABI Publishing. 580 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.

Kenefic, L.S.; Nyland, R.D., eds. 2006. Proceedings of a conference on diameter-limit cutting in northeastern forests. Gen. Tech. Rep. NE-342. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station. 57 p.

Loftis, D.L. 1989. Species composition of regeneration after clearcutting southern Appalachian hardwoods. In: Miller, J.H., ed. Proceedings of the 5th biennial southern silvicultural research conference. Gen. Tech. Rep. SO-74. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 253-257.

Loftis, D.L. 1990. A shelterwood method for regenerating red oak in the southern Appalachians. Forest Science. 36(4): 917-929.

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.

McGee, C.E.; Hooper. R.M. 1975. Regeneration trends 10 years after clearcutting of an Appalachian hardwood stand. Res. Note SE-227. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 4 p.

McNab, W.H.; Loftis, D.L.; Shefield, R.M. 2003. Testing tree indicator species for classifying site productivity in Southern Appalachian hardwood stands. In: Proceedings of the Society of American Foresters 2002 national convention, Winston-Salem, NC, October 5-9, 2002. Bethesda, MD: Society of American Foresters, SAF publication SAF03-01: 350-356.

McShea, W.J.; Healy, W.M.; Deavers, P. [and others]. 2007. Forestry matters: decline of oaks will impact wildlife in hardwood forests. Journal of Wildlife Management. 71(5): 1717-1728.

Meiners, T.M.; Smith, D.W.; Sharik, T.L.; Beck, D.E. 1984. Soil and plant water stress in an Appalachian oak forest in relation to topography and stand age. Plant and Soil. 80(2): 171-179.

Mielke, P.W.; Berry, K.J. 2001. Permutation methods: a distance function approach. New York: Springer-Verlag. 352 p.

Miller, G.W.; Kochenderfer, J.N. 1998. Maintaining species diversity in the central Appalachians. Journal of Forestry. 96(7): 28-33.

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.

Muller, R.N. 1983. Forest regrowth on the Cumberland Plateau and implications for productive management. Southern Journal of Applied Forestry. 7(4): 208-212.

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.

Oliver, C.D. 1978. The development of northern red oak in mixed stands in central New England. Bull. 91. New Haven, CT: Yale University, School of Forestry and Environmental Studies. 63 p.

Oliver, C.D.; Larson, B.C. 1996. Forest Stand Dynamics. New York: John Wiley & Sons. 520 p.

Oswalt, C.M.; Turner, J.A. 2009. Status of hardwood forest resources in the Appalachian region including estimates of growth and removals. Resour. Bull. SRS-142. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 24 p.

Ott, R.L.; Longnecker, M. 2001. Statistical Methods and Data Analysis. 5th ed. Pacific Grove CA: Thomas Learning Inc.: 287-289, 410-416.

Roach, B.A.; Gingrich, S.F. 1968. Even-aged silviculture for upland central hardwoods. Agric. Handb. 355. Washington, DC: U.S. Department of Agriculture. 44 p.

Ross, M.S.; Sharik, T.L.; Smith, D.W. 1986. Oak regeneration after clearfelling in southwest Virginia. Forest Science. 32(1): 157-169.

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. 1979. Regenerating oaks with the shelterwood system. In: Holt, H.A.; Fischer, B.C., eds. Proceedings: the John S. Wright forestry conference. West Lafayette, IN: Purdue Research Foundation: 54-60.

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. 26 p.

Sander, I.L.; Johnson, P.S.; Watt, R.F. 1976. A guide for evaluating the adequacy of oak advance reproduction. Gen. Tech. Rep. NC-23. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 7 p.

SAS 2007. SAS online doc. 9.2. Cary, NC: SAS Institute Inc. http://support.sas.com/documentation/92/. [Date accessed: April 1, 2007].

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. 2008. Crop tree release improves competitiveness of northern red oak growing in association with black cherry. Northern Journal of Applied Forestry. 23(2): 77-82.

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.

Smith, D.W. 1995. The southern Appalachian hardwood region. In: Barrett, J.W., ed. Regional Silviculture of the United States. New York: John Wiley & Sons: 173-225.

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. 7 p.

Tift, B.D.; Fajvan, M.A. 1999. Red maple dynamics in Appalachian hardwood stands in West Virginia. Canadian Journal of Forest Research. 29: 157-165.

Trimble, G.R. 1973. The regeneration of central Appalachian hardwoods with emphasis on the effects of site quality and harvesting practice. Res. Pap. NE-282. Radnor, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 14 p.

Trimble, G.R. 1974. Response to crop-tree release by 7-year-old stems of red maple sprouts and northern red oak advance reproduction. Res. Pap. NE-303. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 6 p.

Ward, J.S. 2009. Intensity of precommercial crop tree release increases diameter growth and survival of upland oaks. Canadian Journal of Forest Research. 39: 118-130.

Ward, J.S.; Stephens. G.R. 1994. Crown class transition rates of maturing northern red oak (*Quercus rubra* L.). Forest Science. 40(2): 221-237.

Yarnell, S.L. 1998. The southern Appalachians: a history of the landscape. Gen. Tech. Rep. SRS-18. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 52p.

Stream Crossings –
Best Management Practices

Moderator:

Hans Williams
Stephen F. Austin State University
Arthur Temple College of Forestry and Agriculture

MONITORING SEDIMENT PRODUCTION FROM FOREST ROAD APPROACHES TO STREAM CROSSINGS IN THE VIRGINIA PIEDMONT

Kristopher R. Brown, W. Michael Aust, and Kevin J. McGuire[1]

Abstract--Reopening of abandoned legacy roads is common in forest operations and represents a reduced cost in comparison to new road construction. However, legacy roads may have lower road standards and require additional best management practice (BMP) implementation upon reopening to protect water quality. Silt fences and elevation measurements of trapped sediment were used to quantify annual sediment delivery rates for reopened bare and existing gravel forest road approaches to stream crossings in the Virginia Piedmont. Additionally, rainfall simulation experiments were performed on reopened legacy road stream crossing approaches to quantify the cost-effectiveness of a range of gravel surface coverage for control of total suspended solids (TSS) concentration from road surface runoff during storm events. In the sediment trap study, mean annual sediment delivery for the reopened bare approaches (98 Mg ha^{-1} $year^{-1}$) was 7.5 times greater than that of the gravel approaches (13 Mg ha^{-1} $year^{-1}$). Problem road approaches were associated with inadequate water control (greater than 75 m between water control structures) and 90 to 100 percent bare soil conditions throughout the year. Median TSS concentration of road surface runoff (g L^{-1}) for the Bare treatment rainfall simulations (2.34 g L^{-1}; 90 to 100 percent bare soil conditions) was 1.8 times greater than Gravel 1 (1.32 g L^{-1}; 25 to 50 percent gravel surface coverage) and 3.3 times greater than Gravel 2 (0.72 g L^{-1}; 50 to100 percent gravel surface coverage). Gravel surfacing of the road approaches cost \$10.27/m of road length for a gravel depth of 7.6 cm and local cost of \$27.78/Mg (\$25 per ton).

INTRODUCTION

Nonpoint sources of stream sedimentation associated with forest operations include forest roads, skid trails, and log decks (Anderson and Lockaby 2011, Croke and Hairsine 2006, Grace 2005). Forest road stream crossings and their associated approaches (i.e., the section of road above and directly connected to the stream crossing) represent erosion sources with the greatest potential for sediment delivery because of nearly direct hydrologic connectivity with streams (Aust and others 2011, Lane and Sheridan 2002, Wear and others 2013). Additionally, stream crossing construction and maintenance involves excavation and fill work with heavy machinery often directly in the stream channel. Excessive stream sedimentation degrades water quality and aquatic habitat (Goode and others 2012, Robinson and others 2010), and protection of water quality is the primary function of forestry best management practices (BMPs) (Aust and Blinn 2004). However, the cost of extensive BMP implementation can be substantial (Shaffer and others 1998). Thus, it is critical to quantify the cost-efficacy of BMPs for forest operations, especially those that represent direct pathways and primary sources of sediment to stream channels (i.e., forest road stream crossings and their associated approaches).

Forest roads provide access for timber harvesting, hazardous fuel reduction efforts, and woody biomass utilization for energy. Reopening of abandoned legacy roads is cheaper than new road construction (Foltz and others 2009), but legacy road construction commonly occurred prior to the BMP era, which has improved standards for water quality protection through the use of proper planning for road location, as well as the management of road grade, stormwater runoff, and erosion and sediment delivery. Upon reopening, legacy roads may require additional BMPs to protect water quality, especially if roads are steep, bare, and have inadequate water control.

Nationwide, forest road sediment delivery to stream channels poses issues for water quality and aquatic habitat degradation (Goode and others 2012, Robinson and others 2010), which is underscored by the 2012 U.S. Supreme Court consideration of the Ninth Circuit Court ruling that was initiated by NEDC versus Brown. The Ninth Circuit ruling stated that roadside ditches are point sources, requiring a National Pollution Discharge Elimination System (NPDES) permit, if they collect and deposit stormwater into the surface waters of the U.S. (Boston 2012). The U.S. Supreme Court decision retained the nonpoint source pollution (NPSP) status of forest roads and silvicultural exemptions by reversing the Ninth Circuit ruling in March 2013, but further litigation is likely until legislation clarifies the NPSP status of forest roads.

[1]Graduate Research Assistant, Professor, and Assistant Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061.

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Nevertheless, it is clear that improved cost-effectiveness and implementation of forest road BMPs are critical for water quality protection.

Two field studies were recently completed in the Virginia Piedmont that addressed the effectiveness of forest road stream crossing approach BMPs. The objectives of the first study were to: (1) use sediment traps to measure annual rates of sediment delivery from forest road approaches to stream crossings due to road reopening in the Virginia Piedmont, (2) compare sediment delivery rates of reopened bare road approaches with existing graveled road approaches, and (3) identify the major road approach characteristics above the stream crossing that govern rates of sediment delivery. The objective of the second study was to evaluate the sediment reduction and cost efficacy of partial and complete graveling of road approaches.

MATERIALS AND METHODS

Study Sites

Fifteen southwestern Virginia Piedmont forest road approaches to stream crossings were selected for study of sediment delivery at the Reynolds Homestead Forest Resources Research Center (RHFRRC), located in Critz, VA (Patrick County). Topography is characterized by rolling hills, with side slopes generally ranging from 8 to 25 percent and a mean elevation of approximately 335 m above mean sea level (NRCS 2013). Mean annual rainfall is 1250 mm, with a mean snow contribution of 270 mm to the total precipitation. Mean air temperature ranges from a low of -1.8 °C in January to a high of 29.7 °C in July (Sawyers and others 2012). The predominant soil series is Fairview sandy clay loam (fine, kaolinitic, mesic typic Kanhapludults). Soil parent material is residuum from mica schist and mica gneiss. Soils are characterized as being moderately eroded and well drained (NRCS 2013), but road construction and traffic can result in soils with reduced infiltration capacity on the running surface. Kadak (2012, unpublished data) used double-ring infiltrometers to measure infiltration rates for six reopened stream crossing approaches at RHFRRC. Infiltration rates ranged from 0.06 to 0.72 cm $hour^{-1}$. In addition, the severe erosion hazard rating for forest roads and trails at RHFRRC (NRCS 2013) underscores the importance of pre-harvest planning to control road grade, minimize stream crossings, and implement BMPs to minimize erosion and sediment delivery. As is typical of the Piedmont region, old agricultural gullies are common because most of the contemporary forested watersheds were formerly in agriculture during the 1800s (Trimble 1974).

Site Survey

In July 2011, a total station (Sokkia total station model SET-520, Tokyo, Japan) was used to measure the length of the road approaches to stream crossings. Length was defined as the distance between the nearest water control structure (i.e. water bar, turnout, or rolling or broad-based dip) and the stream. Road approach slope and mean width of the road (running surface plus ditch, if applicable) were also quantified during the total station survey (table 1).

Treatments

Two different methods were used to quantify sediment delivery and evaluate gravel surfacing BMPs for forest road approaches to stream crossings: (1) a forest operational approach that used silt fences and monthly elevation measurements of trapped sediment to quantify sediment delivery associated with existing graveled road approaches and reopened bare legacy road approaches to stream crossings, and (2) an experimental approach that used rainfall simulation experiments to quantify the cost-effectiveness of a range of gravel surface coverage for control of total suspended solids (TSS) concentration from road surface runoff during storm events.

Sediment trap study--Five road segments were bladed with a bulldozer in late July 2011, creating initial conditions of 100 percent bare soil, to simulate sediment delivery from reopening abandoned legacy roads. Two of the road segments represented road approaches to a 1970s-era abandoned skidder crossing resembling an earthen dam. The remaining three road segments represented sections that

Table 1--Site physical characteristics of the sediment trap and rainfall simulation road approach study plots. Multiple segments within a study plot indicate a break in road grade

Study sites segment ID	Crossing type	Length	Surface width	Slope	Vertical slope	Soil texture
		------------*m*------------		%		
Sediment trap						
Gravel 1,2	Culvert	29.0	3.2	10.0	Linear	Sandy clay loam
Gravel 1,1	-	11.0	3.0	12.0	Linear	Sandy clay loam
Gravel 2	Culvert	24.4	3.4	4.0	Linear	Sandy clay loam
Gravel 3,1	Culvert	12.5	3.4	2.0	Linear	Sandy clay loam
Gravel 3,2	-	39.3	2.7	6.7	Concave	Sandy clay loam
Gravel 4,1	Culvert	9.8	4.5	14.3	Linear	Silt loam
Gravel 4,2	-	31.7	5.2	19.0	Concave	Silt loam
Bare 1	Earth dam	21.0	2.4	21.0	Convex	Clay loam
Bare 2	Earth dam	10.0	2.1	19.0	Concave	Silty clay loam
Bare 3,2	-	19.5	2.7	14.3	Concave	Silty clay loam
Bare 3,1	-	22.6	2.1	13.5	Linear	Silty clay loam
Bare 4	-	129.5	3.0	3.9	Linear	Loamy sand
Bare 5	-	75.1	2.4	4.0	Linear	Silty clay loam
Rainfall simulation						
Crossing 1 right,2	Ford	12.7	4.0	13.5	Concave	Sandy clay loam
Crossing 1 right,1	-	13.1	3.4	5.4	Linear	Sandy clay loam
Crossing 1 left,1	-	9.6	3.0	1.3	Linear	Sandy clay loam
Crossing 1 left,2	-	31.7	2.4	5.7	Concave	Sandy clay loam
Crossing 2 right	Ford	19.2	3.2	13.7	Convex	Sandy clay loam
Crossing 2 left,1	-	19.5	2.4	14.9	Concave	Sandy clay loam
Crossing 2 left,2	-	15.8	2.9	16.0	Linear	Sandy clay loam
Crossing 3 right,2	Ford	29.0	3.8	15.0	Concave	Sandy clay loam
Crossing 3 right,1	-	10.6	2.6	19.1	Linear	Sandy clay loam
Crossing 3 left,1	-	10.0	2.2	6.5	Linear	Clay loam
Crossing 3 left,2	-	13.5	3.3	6.3	Linear	Clay loam

were not immediately connected to stream channels. Where cut and fill road profiles existed, in-sloping and outboard edge berm installation were used to redirect all runoff and sediment to the base of the plot, where silt fences were installed across the entire width of the running surface (and ditch, if applicable) to trap all sediment from the road prism, similar to the method suggested by Robichaud and Brown (2002) (fig. 1). Silt fences were also installed at the base of three culvert crossing approaches on a graveled road that was constructed in November 2010. Finally, silt fences were installed at the base of a legacy gravel road approach to a culvert crossing that was reshaped with a bulldozer blade and re-graveled. Reshaping included in-sloping to the cutbank on the backslope portion of the hillslope and crowning at the toeslope. The treatments resulted in five bare and four graveled road segments.

Rainfall simulation study--Six road approaches to three unimproved ford stream crossings on an abandoned legacy road at RHFRRC were bladed with a bulldozer in late July 2011 in a similar manner to that described above. Open-top box culverts (Trimble and Sartz 1957) were installed at the base of the stream crossing approaches to redirect stormwater runoff toward a 2.5- by 45.7-cm cutthroat flume (Tracom Fiberglass Products, Alpharetta, GA) where runoff volumes were measured during rainfall simulation experiments. Prior to the Bare treatment rainfall simulations, road approaches were tracked with a bulldozer to create conditions of 90 to 100 percent bare soil. Following the Bare treatment rainfall simulations, a dump truck was used to spread a mixture of size 3, 5, and 7 (5.1 to 1.9 cm in diameter) granite gravel at the base of the stream crossing approaches to the following standards: gravel depth = 7.6 cm; width = width of the running surface (mean = 3.0 m); and length = 9.8 m. This treatment was called Gravel 1, and it resulted in 25 to 50 percent gravel surface coverage on the approaches. Following the Gravel 1 rainfall simulations, an additional 9.8 m section of gravel was added to double the length of the first application. This treatment was called Gravel 2, and it resulted in 50 to 100 percent gravel surface coverage on the approaches. Gravel cost $27.78/Mg ($25 per ton) and applying the gravel to a depth of 7.6 cm for a running surface width of 3.0 m resulted in a cost per linear meter of $10.27.

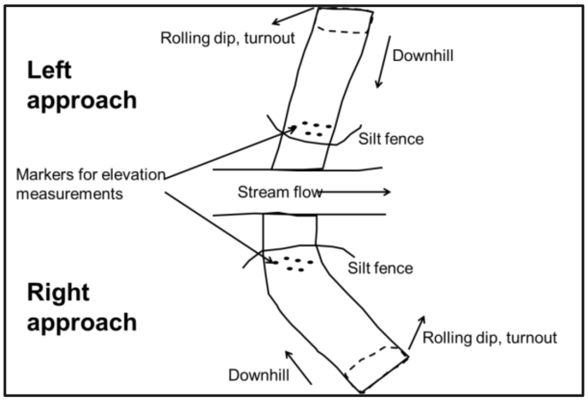

Figure 1--Plan view of the sediment trap study plots and instrumentation.

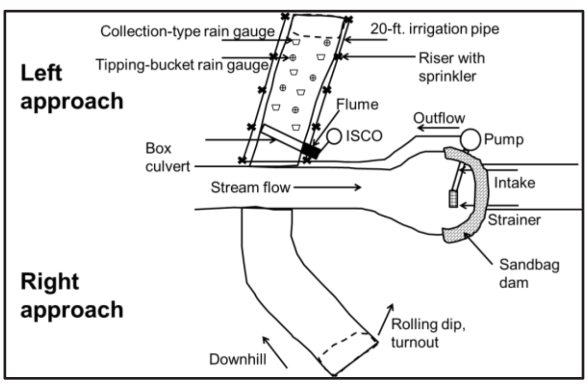

Figure 2--Plan view of the rainfall simulation study plots and instrumentation.

A high-pressure pump with an 18-horsepower gas-driven engine and a maximum flow rate of 37.9 L/second was used to pump source water for the rainfall simulation experiments from temporary impoundments that were created approximately 50 m downstream of each stream crossing (fig. 2). The irrigation setup included a 10.2-cm intake hose with a strainer, a 7.6-cm outflow component, and fire hose to connect to 7.6-cm aluminum irrigation pipelines that ran parallel to the road approach and on both sides of the road. A 3.0-m pipe and 90^o angle couplings joined the two parallel segments of pipe at the base of the road plot. The pipelines ran on each side of the road surface and consisted of 6.1-m irrigation pipes that were coupled together for the length of the approach. A water control structure and open-top box culvert served as the upper and lower boundaries of the road approach plots, respectively. Similar to the silt fence study plots, in-sloping and outboard edge berm installation were used to redirect all runoff and sediment to the box culverts at the base of the plots. Quick coupling risers with sprinkler heads were located at 6.4-m intervals along the study plots, and they stood 3.4-m high. The same irrigation pipe and risers were used previously in rainfall simulation experiments to test agricultural BMP effectiveness to minimize soil erosion during rain events (Dillaha and others 1988). The irrigation setup is designed to apply rainfall to the study plots at a rate of 5.1 cm hour^{-1}. A series of three rainfall simulation experiments, ranging in length from 10 to 50 minutes, were performed for each treatment (Bare, Gravel 1, and Gravel 2) for each of the six road approach study plots for a total of 3 x 3 x 6 = 54 rainfall simulation experiments.

Field Measurements

Sediment trap study--A network of erosion pins and differential leveling with a total station was used to approximate monthly sediment deposition at the silt fences between August 2011 and August 2012. Sediment volumes (m^3) for each measurement interval were calculated by multiplying road surface depositional area (m^2) by elevation gain (m). Sediment volumes were converted to a sediment load (Mg) by multiplying by bulk density (Mg m^{-3}) of the trapped sediment. Bulk densities were obtained for the sediment sampled via the soil exaction method (SSSA 1986). Hourly rainfall data obtained from a Soil Climate Analysis Network weather station (NRCS 2010) located approximately 0.8 km from the study sites were summed to calculate total rainfall per measurement interval. Mean rainfall intensity (cm hour^{-1}) was also calculated for each measurement interval.

Rainfall simulation study--Rainfall amount and intensity were measured with six wedge collection-type rain gauges and five automatic tipping-bucket rain gauges (ECRN-50 Low-Resolution Rain Gauge, Decagon Devices, Inc., Pullman, WA) that were set to log data at 1-minute intervals. Surface runoff volume was measured with a 2.5- by 45.7-cm cutthroat flume fitted to the outflow end of the open-top box culverts. The flume was equipped with a pressure transducer (HOBO U20 Water Level Data Logger, Onset Computer Corporation, Bourne, MA) to measure water level at 1-minute intervals. Water level data was converted to discharge (L $second^{-1}$) through the use of a stage-discharge equation for the specific dimensions of the cutthroat flume. An ISCO automatic stormwater sampler (ISCO 3700 Series, Teledyne ISCO, Lincoln, NE) was programmed to collect 500 mL samples of stormwater runoff from the water flowing through the flume at 2- to 5-minute intervals, depending on the duration of the rainfall simulation event. Stormwater runoff samples were analyzed in the lab for total suspended solids (TSS, in g L^{-1}) by way of vacuum filtration of a known amount of sample volume and obtaining dry weights of the sediment trapped by the filters.

Statistical Analysis

Differences in median sediment delivery per measurement interval by road surface type (bare, graveled) were analyzed with a Wilcoxon signed-rank test (d.f. = 1; N = 117; $\alpha = 0.10$). Differences in median TSS concentration of surface runoff (g L^{-1}) by treatment type (Bare, Gravel 1, Gravel 2) were analyzed with a Kruskal-Wallis rank sum test (d.f. = 2 ; N = 681; $\alpha = 0.05$), and Steel-Dwass all pairs nonparametric comparisons were used to determine statistically significant groups.

RESULTS AND DISCUSSION

Field Measurements

Sediment trap study--Total rainfall during the study period (August 5, 2011 to August 5, 2012) was 1495 mm. Mean annual sediment delivery for the reopened bare approaches (98 Mg ha^{-1} $year^{-1}$) was 7.5 times greater than that of the gravel approaches (13 Mg ha^{-1} $year^{-1}$). Problem road approaches were associated with inadequate water control (> 75 m intervals for water control structures) and 90 to 100 percent bare soil conditions throughout the year. Reopened bare road approaches having both forest canopy cover during the growing season and litterfall ground coverage during the fall and winter months had lower sediment delivery rates than bare approaches located in a clearcut area with 4-year-old loblolly pine regeneration (fig. 3). Median sediment delivery per measurement period was 2.0 Mg ha^{-1} for the bare plots and 0.3 Mg ha^{-1} for the gravel plots (Chi-square statistic = 22.2, d.f. = 1, $p < 0.0001$). These findings exemplify the importance of proper implementation of water control and gravel surfacing BMPs to minimize direct inputs of surface runoff and sediment from forest road approaches to stream crossings. For haul roads, the Virginia Department of Forestry BMP manual recommends that gravel surfacing be used to cover the entire road approach that is delivering sediment to the stream and to redistribute stormwater runoff from the road surface and out over a well-vegetated or rough surface at least 7.6 m before the stream crossing (VDOF 2011).

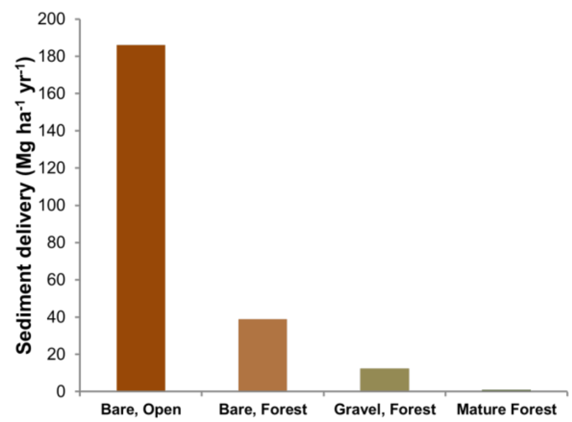

Figure 3--Mean sediment delivery rate (Mg ha^{-1} $year^{-1}$) by road surface type (bare, gravel) and forest canopy cover (open, forest) as compared with typical erosion rates in a mature forest setting. Canopy cover during the growing season and litterfall during the fall and winter seasons helped to reduce bare soil percentages, and thus sediment delivery rates at the “Bare, Forest” sites.

Rainfall simulation study--Simulated rainfall events had recurrence intervals of < 1 to 5 years for Critz, VA. Surface runoff was commonly generated within the first 5 minutes of the onset of rainfall. Gravel application reduced TSS concentration of surface runoff from the road approach study plots (fig. 4). Median TSS concentration of road surface runoff (g L^{-1}) for the Bare treatment rainfall simulations (2.34 g L^{-1}; 90 to 100 percent bare soil conditions) was 1.8 times greater than Gravel 1 (1.32 g L^{-1}; 25 to 50 percent gravel surface coverage) and 3.3 times greater than Gravel 2 (0.72 g L^{-1}; 50 to 100 percent gravel surface coverage). All treatments were significantly different (Bare > Gravel 1 > of

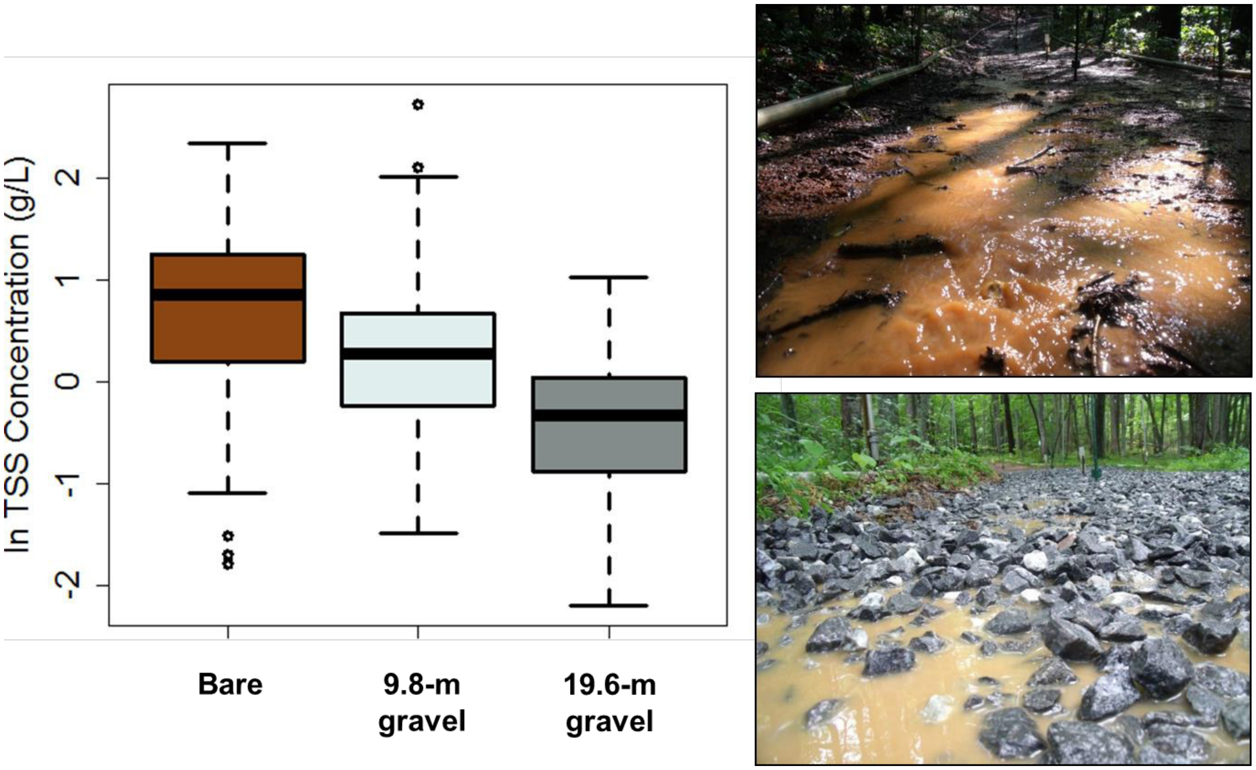

Figure 4--Box and whisker plots (showing the 5th, 25th, 50th, 75th, and 95th percentiles) of natural log-transformed TSS concentration of surface runoff (g L^{-1}) for all of the rainfall simulation experiments by treatment type (Bare, Gravel 1, and Gravel 2). N = 228, 222, and 231 for the Bare, Gravel 1, and Gravel 2 treatments, respectively.

Gravel 2). Local granite gravel cost for a mixture of size 3, 5, and 7 (5.1 to 1.9 cm diameter) was $27.56/Mg ($25 per ton). For a gravel application depth of 7.6 cm (3 inches), the cost surfacing the stream crossing approaches was$10.27/m ($3.13 per foot) of approach length.

CONCLUSIONS

Legacy roads and associated stream crossings have the potential to deliver significant quantities of sediment to streams if the roads have inadequate water control, are too steep, or lack surface cover. Therefore, upon reopening, legacy roads may require corrective BMPs to protect water quality. Corrective BMPs, such as gravel, can minimize the sediment contributions of stream crossing approaches. Judicious BMP implementation can reduce sediment inputs to streams and strike a balance between sediment reduction efficacy and BMP implementation cost. Many of the potential threats to water quality associated with forest roads, as well as the cost associated with corrective BMP implementation, can be minimized through pre-harvest planning to properly locate forest roads, minimize road length and stream crossings, and control grade, water, and erosion. Therefore, careful road design is an important BMP in itself.

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LITERATURE CITED

Anderson, C.J.; Lockaby, B.G. 2011. Research gaps related to forest management and stream sediment in the United States. Environmental Management. 47(1): 303-313.

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.

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. South Journal of Applied Forestry. 35(3): 123-130.

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.

Croke, J.C.; Hairsine, P.B. 2006. Sediment delivery in managed forests: a review. Environmental Reviews. 14: 59-87.

Dillaha, T.A.; Ross, B.B.; Mostaghimi, S. [and others]. 1988. Rainfall simulation: a tool for best management practice education. Journal of Soil and Water Conservation. 43(4): 288-290.

Foltz, R.B.; Copeland, N.S.; Elliot, W.J. 2009. Reopening abandoned forest roads in northern Idaho, USA: quantification of runoff, sediment concentration, infiltration, and interrill erosion parameters. Journal of Environmental Management. 90(8): 2542-2550.

Goode, J.R.; Luce, C.H.; Buffington, J.M. 2012. Enhanced sediment delivery in a changing climate in semi-arid mountain basins: implications for water resource management and aquatic habitat in the northern Rocky Mountains. Geomorphology. 139: 1-15.

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

Lane, P.N.J.; Sheridan, G.J. 2002. Impact of an unsealed forest road stream crossing: water quality and sediment sources. Hydrological Processes. 16(13): 2599–2612.

NRCS [Natural Resources Conservation Service]. 2010. Soil climate analysis network site 2089. Portland, OR: U.S. Department of Agriculture Natural Resources Conservation Service, National Water and Climate Center. http://www.wcc.nrcs.usda.gov/nwcc/site?sitenum=2089&state=va. [Date accessed: February 21, 2013].

NRCS [Natural Resources Conservation Service]. 2013. Web soil survey. Washington, DC: U.S. Department of Agriculture Natural Resources Conservation Service. http://websoilsurvey.nrcs.usda.gov/. [Date accessed: February 21, 2013].

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

Robinson, C.; Duinker, P.N.; Beazley, K.F. 2010. A conceptual framework for understanding, assessing, and mitigating ecological effects of forest roads. Environmental Reviews. 18: 61-86.

Sawyers, B.C.; Bolding, M.C.; Aust, W.M.; Lakel, W.A., III. 2012. Effectiveness and implementation costs of overland skidding closure techniques in the Virginia Piedmont. Journal of Soil and Water Conservation. 67(4): 300-310.

Shaffer, R.M.; Haney, H.L., Jr.; Worrell, E.G., Aust, W.M. 1998. Forestry BMP implementation costs for Virginia. Forest Products Journal. 48(9): 27-29.

SSSA [Soil Science Society of America]. 1986. Methods of soil analysis . Part I. Physical and mineralogical methods. 2[d] ed. Agronomy Monogr. 9(1). Klute, E.A., ed. Madison, WI: American Society of Agronomy, Inc. and Soil Science Society of America, Inc. 1188 p.

Trimble, S.W. 1974. Man-induced soil erosion on the southern piedmont: 1700-1970. Ankeny, IA: Soil and Water Conservation Society. 80 p.

Trimble, G.R.; Sartz, R.S. 1957. How far from a stream should a logging road be located? Journal of Forestry. 55: 339-341.

VDOF [Virginia Department of Forestry]. 2011. Virginia's forestry best management practices for water quality. Tech. Man. 2011. Charlottesville, VA: Virginia Department of Forestry. 165 p. http://www.dof.virginia.gov/wq/index-BMP-Guide.htm. [Date accessed: February 4, 2013].

Wear, L.R.; Aust, W.M.; Bolding, M.C. [and others]. 2013. Effectiveness of best management practices for sediment reduction at operational forest stream crossings. Forest Ecology and Management. 289(1): 551-561.

SEDIMENT DEPOSITION FROM FOREST ROADS AT STREAM CROSSINGS AS INFLUENCED BY ROAD CHARACTERISTICS

A.J. Lang, W.M. Aust, M.C. Bolding, and K.J. McGuire[1]

Abstract--Recent controversies associated with ditched forest roads and stream crossings in the Pacific Northwest have focused national attention on sediment production and best management practices (BMPs) at stream crossings. Few studies have quantified soil erosion rates at stream crossings as influenced by road characteristics and compared them to modeled rates. Soil erosion rates were measured and modeled from forest roads that represented a range of road classes (permanent high standard to temporary low standard). Forty road approaches were identified in the Piedmont and Mountain regions of Virginia and categorized into four general road classes. Road attributes were characterized at each crossing (BMPs used, road width, grade, gravel, cover, cut and fill slope ratios, ditch characteristics, etc.). At each stream crossing, conveyor belts were installed as water-control devices across the road to divert sediment from the stream crossing approach into silt fence sediment traps. Sediment pins were installed adjacent to the silt fence to allow periodic measurement of sediment depths. Additionally, erosion potentials for approaches were modeled with the Universal Soil Loss Equation (USLE) as modified for forestry and compared to actual sediment deposition near the stream. Data presented represents < 1year of sediment measurements from the stream crossings.

INTRODUCTION

Erosion from unsealed road surfaces is a primary contributor of sedimentation within forests in the United States (Hewlett 1982, Stuart and Edwards 2006, Yoho 1980). Sedimentation, regardless of its origin, can negatively impact aquatic stream life and society (Gibson and others 2005, Hewlett 1982). Forestry best management practices (BMPs) are methods and practices designed to minimize water quality problems associated with forest management practices (Sohngen and others 1999). BMPs have been designed to focus on forest operations with the greatest risks of environmental degradation, such as stream crossings, skid trails, landing sites, and roads (Aust and Blinn 2004). Although BMPs address a variety of nonpoint source pollutants, including nutrients, temperature, organics, and chemicals, the primary purpose of BMPs is to reduce erosion and subsequent sediment yields. Many states throughout the United States commonly use forestry BMPs at stream crossings to reduce negative impacts to waters (Shepard and others 2004).

In forest management, significant attention has been directed to BMPs applied at stream crossing because of the proximity to stream networks (Aust and Blinn 2004). Road construction disturbances, such as clearing vegetation, constructing ditches, and compaction of the road surface on stream crossings and their approaches, alter the hydrology, increasing the probability overland flow and subsequent soil erosion and sediment delivery to the stream (Ziegler and others 2007). Planning road location, increased water control structures, increased surface coverage, and decreased approach grades, among others, have been shown to reduce erosion rates and sediment delivery at stream crossings (Luce and Black 1999, Swift 1985).

Several equations and soil loss models have been developed to evaluate the effectiveness of BMPs and other methods to curb soil losses from stream crossings. Erosion models have a great potential to aid land managers and other stakeholders in planning for and selecting preventative measures (Fu and others 2010). The complexities of soil erosion over spatial and temporal scales and a lack of quantified values over time create difficulty within the modeling processes (Jetten and others 1999, Sukhanovskii 2010). The Universal Soil Loss Equation (USLE) method of estimating soil erosion is the most widely used in the United States because of its ease and ability to generate estimations in the field (Christopher and Visser 2007). While the estimates are imperfect, several studies have shown the USLE estimations to be capable of accurately ranking erosion rates from different treatments (Sawyers and others 2012, Wade and others 2012).

Additional legal focus has been placed on forest road stream crossings. In the Pacific Northwest, a series of lawsuits have challenged the status of forest operations to construct stream

[1]Graduate Research Assistant, Professor, Associate Professor, and Assistant Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, Virginia, 24061.

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crossings under the silviculture exemption without additional Environmental Protection Agency oversight (Boston 2012, Boston and Thompson 2009). While the U.S. Supreme Court ruling has reversed the decision of the Ninth Circuit Court, further litigation is likely to ensue. The issue emphasizes the need for better understanding of sediment delivery from stream crossings and further research assessing the effectiveness of BMPs (Anderson and Lockaby 2011). The objective of this study was to quantify sediment deposition across different road standards and compare them to modeled erosion estimates.

MATERIALS AND METHODS

Study Sites

Twenty-four stream crossing approaches were located on six tracts of timber in the Piedmont physiographic regions of Virginia on MeadWestvaco (MWV) managed lands. MWV was managing stands of loblolly pine (*Pinus taeda* L.) for 18- to 25-year rotations, and hunt clubs leased the land for recreational purposes. An additional 16 stream crossing approaches were located on three tracts in the Ridge and Valley physiographic region of Virginia on USDA Forest Service and Virginia Tech school forest managed lands. Forest Service roads were gated and received low traffic throughout the study. The Virginia Tech school forest had moderate to high levels of traffic, as it was utilized for teaching exercises and by municipal personnel. All streams are classified as intermittent or perennial.

Installation

A rubber conveyor belt was installed at approximately a 45° angle across the roads at the lowest topographic point nearest the stream crossing using hand tools. At each stream crossing approach, a narrow trench was excavated at a 30° to 45° angle across the road. A conveyor belt with the dimensions of approximately 30-cm wide by 1.25-cm thick was cut to length according to road width and buried, leaving approximately 15 cm of the belt exposed above the surface in order to divert water and sediment from the road into the adjacent silt fence catchment area (Robichaud and Brown 2002). Several pins were placed in the catchment area for periodic measurements of sediment depths (Lakel and others 2010). The rubber conveyor belt allowed for passage of vehicular traffic. To prevent the belts from being pulled out of the trench, two 0.9- by 46-cm rebar stakes were driven in on the edges of the belt (away from the travel surface) at an angle. In addition to the stakes, three pairs of scrap boards were affixed to each side of the belt using screws to fasten them to the bottom portion of the conveyor belt. Some locations also required the conveyor belt to be altered further to disperse the tension generated by traffic. In these instances, vertical slits were made in the belt to alleviate additional force and prevent the belt from being removed by traffic passes. Installations occurred between June and November 2012.

Data Collected

The following approach characteristics were collected: GPS location, aspect, distance to nearest water control structure, length to natural grade break, width of the running surface, template, road surfacing type and coverage, grade, soil texture, slope shape, canopy coverage, crossing type, number of water control structures, ditch length, width, and depth, cut and fill slope dimensions and the percent vegetation coverage, and time since last road maintenance. Road classes were assessed and assigned by a panel of professional foresters and were used as a basis for comparison (table 1). Sites were revisited seasonally to re-examine canopy and surface coverage. Precipitation data were collected from the nearest known weather stations to study sites. Periodic measures of sediment depth were taken using an electronic total station to the nearest 0.01 foot.

Sediment Delivery Calculation

Sediment yield is the ratio of sediment delivery and total gross erosion (Glymph 1954, Lu and others 2006, Williams 1977). This study compared trapped sediment deposition (sediment yield) verses USLE model estimates of total potential erosion to estimate sediment delivery ratios. Factors that affect sediment delivery ratios include sediment source, proximity to water, watershed and soil characteristics, and topography (Lu and others 2006).

Table 1--Road class sampling by physiographic region

Road class	Mountains	Piedmont	Total
Class 1	0	4	4
Class 2	8	3	11
Class 3	8	12	20
Class 4	0	5	5
Total	16	24	40

RESULTS AND DISCUSSION

Road Characteristics and Sediment Deposition

Higher road classes were found to produce less sediment than lower class roads (table 2). However, the individual road characteristics that collectively created the road classes were not good indicators of trapped or modeled sediment deposition. Specifically, mean percent slope, road area, percent bare soil and distance to the nearest water control structure varied in predicted and measured sediment deposition by assigned road classification. The variance observed can be understood and justified by realizing the spatial and temporal complexity of road approaches. Each approach is a complex combination of road characteristics. Better characteristics for one attribute may offset poorer characteristics of another and vice versa. Documented characteristics of approaches in different road classes are important for calibration of sediment delivery models.

Table 2--Mean trapped and predicted sediment yield and sediment delivery ratio (SDR) by assigned road class

Road Class	Trapped sediment yield	Predicted sediment yield	SDR
	------*tons/acre/year*------		*percent*
Class 1	0.05	1.93	3
Class 2	1.23	0.61	100
Class 3	1.48	1.59	93
Class 4	6.92	6.18	100

Sediment Delivery

The mean modeled (USLE) gross erosion rates by road class were similar to the trapped deposition (table 2). None of the road classes exceeded a mean of 7 tons per acre per year of trapped sediment deposition. Mean trapped sediment increased with decreasing road standards. Mean (USLE) predicted soil erosion increased in classes 2 through 4, while the class 1 mean predicted (USLE) was greater than class 2 and 3. Several of the class 1 roads in this study had steep approaches that may have increased predicted means (USLE) since the USLE model emphasizes slope. If sediment delivered to silt fence catchment areas equated to sediment delivered to streams, then the sediment delivery ratio for class 1 roads was only 3 percent, while all other classes were at or near 100 percent (table 2).

CONCLUSION

Assemblages of road characteristics collectively control the variation in sediment yield. Lower standard roads have a greater erosion potential on a per unit area basis, but may erode less than higher standard roads due to smaller areas. Research has been conducted regarding road characteristics and their influence on erosion, but less research has examined the sediment delivery ratio attributable to BMPs and road standards. Observed road approaches in this study captured a range of road characteristics that enabled us to make simple calculations of sediment delivery ratios. These sediment delivery ratios are an index of BMP efficiency. Additionally, the range of data should provide information to better calibrate erosion models. Overall, it appears that the enhanced BMPs used by the higher class roads decreased the sediment delivery to streams.

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LITERATURE CITED

Anderson, C.J.; Lockaby, B.G. 2011. Research gaps related to forest management and stream sediment in the United States. Environmental Management. 47(2): 303-313.

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.

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.

Christopher, E.A.; Visser, R. 2007. Methodology for evaluating post harvest erosion risk for the protection of water quality. New Zealand Journal of Forestry. 52(2): 20-25.

Fu, B.H.; Newham, L.T.H.; Ramos-Scharron, C.E. 2010. A review of surface erosion and sediment delivery models for unsealed roads. Environmental Modelling Software. 25(1): 1-14.

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.

Glymph, L.M. 1954. Studies of sediment yields from watersheds. International Union of Geodesy and Geophysics, International Association Hydrology 10th general assembly. Part 1: 178-191.

Hewlett, J.D. 1982. Principles of forest hydrology. Athens, GA: University of Georgia Press. 183 p.

Jetten, V.; Roo, A.; Favis-Mortlock, D. 1999. Evaluation of field-scale and catchment-scale soil erosion models. Catena. 37: 521-541.

Lakel, W.A.; Aust, W.M.; Bolding, M.C. [and others]. 2010. Sediment trapping by streamside management zones of various widths after forest harvest and site preparation. Forest Science. 56(6): 541-551.

Lu, H.; Moran, C.J.; Prosser, I.P. 2006. Modelling sediment delivery ratio over the Murray Darling basin. Envrionmental Modelling Software. 21(9): 1297-1308.

Luce, C.H.; Black, T.A. 1999. Sediment production from forest roads in western Oregon. Water Resource Research. 35(8): 2561-2570.

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

Sawyers, B.C.; Bolding, M.C.; Aust, W.M.; Lakel, W.A. 2012. Effectiveness and implementation costs of overland skid trail closure techniques in the Virginia Piedmont. Journal of Soil Water Conservation. 67(4): 300-310.

Shepard, J.P.; Aust, W.M.; Dolloff, C.A. [and others]. 2004. Forestry best management practices research in the eastern United States: the state of the science 2002. Water, Air, and Soil Pollution. Focus 4(1): 1-3.

Sohngen, B.; Mendelsohn, R.; Sedjo, R. 1999. Forest management, conservation, and global timber markets. American Journal of Agriculture Economics. 81(1): 1-13.

Stuart, G.W.; Edwards, P.J. 2006. Concepts about forests and water. Northern Journal of Applied Forestry. 23(1): 11-19.

Sukhanovskii, Y.P. 2010. Rainfall erosion model. Eurasian Soil Science. 43(9): 1036-1046.

Swift, L.W., Jr. 1985. Forest road design to minimize erosion in the South Appalachians. In: Blackmon, B.G., ed. Proceedings of forestry and water quality: a mid-south symposium. Monticello, AR: University of Arkansas, Cooperative Extension Service: 141-151.

Wade, C.R.; Bolding, M.C.; Aust, W.M.; Lakel, W.A. 2012. Comparing sediment trap data with the USLE-forest, RUSLE2, and WEPP-road erosion models for evaluation of bladed skid trail BMPs. Transactions of the American Society of Agricultural Engineers. 55(2): 403-414.

Williams, J.R. 1977. Sediment delivery ratios determined with sediment and runoff models. International Association of Hydrological Sciences Publication. 122: 168-179.

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

Ziegler, A.D.; Negishi, J.N.; Sidle, R.C. [and others]. 2007. Persistence of road runoff generation in a logged catchment in Peninsular Malaysia. Earth Surface Processes and Landforms. 32(13): 1947-1970.

EFFECTIVENESS OF FORESTRY BMPS FOR STREAM CROSSING SEDIMENT REDUCTION USING RAINFALL SIMULATION

Brian C. Morris, M. Chad Bolding, and W. Michael Aust[1]

Abstract--Recent decisions by the United States Supreme Court and United States Environmental Protection Agency (EPA) have re-emphasized the importance of forestry best management practices (BMPs) at stream crossings. Stream crossings are potential major sources of sediment due to their direct connectivity between the potential erosion source and the stream, which eliminates potential sediment reduction provided by filter/buffer strips and streamside management zones. The effectiveness of stream crossing BMPs for sediment control were tested for a permanent bridge crossing, culvert crossing, and improved ford crossing on three first-order streams in the Virginia Piedmont using rainfall simulation. The three crossings were located on a low standard legacy road having unimproved ford crossings before experimentation. All legacy fords received three levels of rainfall intensity via simulation prior to crossing installation. Following crossing installation, rainfall simulations were performed at each of the crossings under the following three treatments: (1) minimal levels of BMP erosion control (Low); followed by (2) installation of BMPs recommended by the Virginia BMP Manual (Medium); and (3) erosion control measures beyond the Virginia BMP Manual (High). Stream sediment (TSS) was monitored upstream and downstream during rainfall simulations to determine total sediment contribution from each individual crossing. The comparison of minimal BMPs, recommended BMPs, and extensive protection provides insight into the erosion associated with the crossing types and the effectiveness of current BMPs for nonpoint source pollution (NPSP) reduction. The Culvert crossing produced a sediment concentration (2.9 g/L) that was double the concentration produced by the Ford crossing (1.4 g/L) and over 10 times the concentration of the Bridge crossing (0.2 g/L).

INTRODUCTION

Forestry best management practices (BMPs) have proven to be effective (Aust and Blinn 2004, Briggs 1998, Shepard 2006, Wynn and others 2000). However, stream crossings have been identified as the primary source of stream sedimentation in forested landscapes (Taylor and others 1999). This sedimentation associated with stream crossings is due to the stream crossing approach and structure providing a source area for erosion (i.e. road surface, cut and fill slopes) that is able to flow directly into the stream channel (Lane and Sheridan 2002). The lack of water-control structures between the crossing structure and the stream results in a high sediment delivery ratio. Current methods of reducing erosion from non-point source pollutants (NPSP) (i.e. stream crossings) and subsequent sedimentation are based upon BMPs which are administered by individual states in accordance with the Federal Water Pollution Control Act of 1972 (Ice and others 2010). BMP requirements and their administration differ by state (i.e. regulatory or voluntary BMPs); however, recent U.S. Supreme Court cases (i.e. Decker versus Northwest Environmental Defense Center) have emphasized the national importance of forestry activities associated with stream crossings. The U.S. Supreme Court overturned a Ninth Circuit Court ruling that would have required National Pollutant Discharge Elimination System (NPDES) permits for any concentrated waste water discharge on forest roads, including stream crossings and ditched roads. The court ruling allows for states to maintain their current BMP systems for stream crossings; however, the Environmental Protection Agency (EPA) has clarified the treatment of forest roads as NPSP. The probability of future lawsuits that will call the rules into question emphasizes the need to consider stream crossing BMPs. These potential changes have increased interest in finding erosion control measures that will be both environmentally and economically efficient (Boston 2012). To better understand the sediment production from specific types of stream-crossing structures, studies must be designed to isolate sediment that is produced from the crossing structure from sediment produced upstream or on the road approach (Taylor and others 1999).

OBJECTIVES

This study was designed to isolate the sediment production from stream-crossing structures. This approach allowed us to determine the sediment contribution from three different crossing structures (Ford, Culvert, and Bridge). Three levels of BMPs were also applied to each crossing; Low (no BMPs), Medium (BMPs equivalent to Virginia Department of Forestry Standards) and High (BMPs beyond Virginia Department of Forestry Standards).

[1]Graduate Research Assistant, Associate Professor, and Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061.

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METHODS

This study was conducted on Virginia Tech's Reynolds Homestead Forest Resources Research Center in Critz, VA. The site is along the western edge of the Piedmont physiographic region. Stream crossings and BMPs were installed on three first-order streams that cross a single legacy (> 100 years old) road. All three original stream crossings on the legacy road were unimproved legacy ford crossings with native stream beds, with steep and non-perpendicular approaches. The road was built prior to current BMPs and road design standards. Prior to conducting the study, ISCO 3700 automatic water samplers (Teledyne ISCO, Lincoln, NE) were installed 66 feet upstream and 66 feet downstream of each crossing, and HOBO® water level loggers (Onset Computer Corporation, Bourne, MA) were also installed upstream of each crossing. The three legacy ford crossings were replaced by a Bridge (100-acre watershed), a Culvert (42-acre watershed), and an improved Ford (80-acre watershed). The Bridge installation included abandoning the old ford crossing and approximately 25 feet of the approach to allow for a 24- by 12- by 0.75-foot white oak (*Quercus alba* L.) stringer bridge, consisting of three 4-foot-wide panels. The 25 feet of the crossing approach were abandoned to allow for the bridge to be aligned perpendicular to the stream. The western approach required the construction of an abutment to support the 9-foot span. This was completed using 3-foot-wide by 3-foot-tall gabion baskets that were filled with #3-0 rock (2- to 8-inch stone). Both approaches were covered with geotextile and 3 inches of #357 (1/2- to 2-inch stone) drain rock prior to installing the bridge panels to prevent the panel from sitting directly on the soil. The initial rainfall simulation with Low BMPs consisted of the bridge installed to drivable conditions with bare soil on the approaches and fill material. The Medium level of BMPs included the addition of gravel to the running surface of the road, and the High level of BMPs included the addition of rip-rap to the fill slopes and covering of all bare soil. The gravel was applied from the crossing structure beyond the next brake in slope, resulting in no bare soil on the road surface subjected to rainfall simulation.

The legacy ford replaced by the Culvert crossing consisted of steep approaches that did not cross the stream at right angles. Legacy approaches were abandoned, and a new road alignment was located. A 40-foot-long by 36-inch-diameter culvert replaced the legacy ford. The channel was excavated with a New Holland TN 750, 75 hp farm tractor with a three-point backhoe attachment. The culvert pipe was installed at the natural stream gradient and at an elevation which allowed for bed load material to be transported into and settle in the culvert bottom, providing a natural stream bed within the culvert. Fill material was sourced from the road realignment, pushed over the culvert, and compacted with a John Deere 450E bulldozer. Approximately 3 feet of fill material was added on top of the culvert to allow for proper vertical road alignment. The fill slopes were compacted with the bulldozer. The Low treatment consisted of no BMPs which resulted in bare soil on the road surface and fill slopes with no water-control structures between the road surface and the stream. The Medium level of BMPs included the addition of geotextile and gravel on the running surface of the road, and the High level of BMPs consisted of the addition of rip-rap to the fill slopes directly above the channel and the application of grass seed and straw mulch on all bare soil.

The improved Ford crossing was constructed within the original legacy ford. The Low level of BMPs included limited rock on the road surface and re-grading the approaches to allow for easier truck traffic, simulating a disturbance that would be created if a log truck crossed the stream. The Medium level of BMPs consisted of improving the road alignment slightly and adding gravel down to the water line of the stream within the running surface of the road. The High level BMP treatment included the installation of Geo-Web in the stream bed and the application of gravel within the running surface in the Geo-Web. The installation of the Geo-Web required excavating the stream channel 6 inches below the natural gradient and backfilling with Virginia Department of Transportation (VDOT) #5 (average ¾-inch stone) gravel.

All three crossings received three levels of rainfall simulation. Simulations were conducted utilizing an 18-hp centrifugal pump with a 4-inch-diameter suction hose submerged in a pond that was downstream of the crossings. The pump pressurized 3-inch-diameter (50- to 100-foot length) fire hose which fed a 2-inch PVC manifold that was used to distribute water to eight sprinkler risers which were 10-feet tall with 1-inch PVC pipe connected to Wobler© sprinkler heads. The sprinkler heads were chosen due to the ability to change the nozzle diameter and the simulated rainfall intensity. This resulted in three distinct rainfall intensities during the simulations Low (0.5 to 1.0 inch per hour), Medium (1.5 to 2.0 inches per hour) and High (2.0 to 2.5 inches per hour). Each of the three rainfall simulations was conducted for 30 minutes. The sprinklers were arranged to allow for rainfall on the stream-crossing structure with minimal rainfall application to the approaches beyond the crossing structure and surrounding area. During the simulations, water samples were collected by the upstream ISCO at 10-minute intervals, downstream at 5-minute intervals during the simulation, and for 30 minutes after rainfall ended. Water samples were processed for total suspended solids (TSS) concentration in g/L by vacuum-filtering 250 ml of stream water through 47-mm ProWeigh filters. The filters with the sediment were dried for 24 hours at 105 °C and weighed. The HOBO water level loggers were used to monitor stream stage during the events. Stream discharge was determined through the use of state-discharge relationships that were created by comparing stage measurements with discharge measurements made with the salt dilution method (Moore 2004, 2005). The sediment concentration and stream discharge were used to determine mass of sediment produced by the crossing during the simulated storm events.

RESULTS AND DISCUSSION

Rainfall simulation experiments were effective at creating artificial storm events for all three crossings. The High-intensity rainfall simulation resulted in rainfall intensities of 2.0 to 2.5 inches per hour while the Medium-intensity simulation resulted in rainfall intensities of 1.5 to 2.0 inches per hour, and the Low-intensity simulation resulted in a rainfall intensity of 0.5 to 1.0 inches per hour over a 30-minute rainfall duration. When all rainfall intensities are combined for each BMP level and crossing type, the Low level of BMPs on the Culvert crossing produced the greatest TSS concentration while the Medium BMP level on the Ford results in the second greatest TSS concentration, and all three BMP treatments on the Bridge produce maximum sediment concentrations below the Culvert and Ford (fig. 1). The maximum sediment contribution for the Culvert crossing was 2.9 g/L while the maximum sediment contribution for the Ford was 1.4 g/L, and the maximum sediment contribution for the Bridge was 0.2 g/L. When comparing the crossings by BMP level, the High level of BMPs for all three crossings resulted in decreased stream sediment, although the Bridge showed little response in sediment levels for the three levels of BMPs. The sediment concentrations for the three levels of BMPs for the Ford showed a different pattern than the Culvert, with the maximum concentration occurring when the Medium level of BMPs was subjected to rainfall simulation. However, the High level of BMPs on the Ford still resulted in lower sediment concentrations than the Medium and Low. The average TSS concentration during the rainfall simulations shows the Low BMP treatment on the Culvert producing the greatest TSS concentration while the ranking of treatments for the Ford follows that of the Culvert with the Low level of BMPs producing the greatest average TSS concentration followed by the Medium BMP level and the High BMP level producing the lowest average TSS concentration (fig. 2). The Bridge crossing produced average TSS concentrations below the average concentrations of Culvert and Ford crossings at all levels of BMPs (fig. 2).

The increased TSS concentrations produced by the Culvert crossing suggest that in order to further reduce sedimentation from stream crossings the Culvert crossing should be an area of focus. The Culvert crossing showed greater maximum TSS concentrations during the High and Medium rainfall intensities than the Low rainfall intensity with the Medium rainfall intensity and the Low BMP level producing the greatest Maximum sediment concentration (fig. 3). The maximum TSS concentration for the construction phase was 2.5 g/L while the maximum concentration for the High rainfall

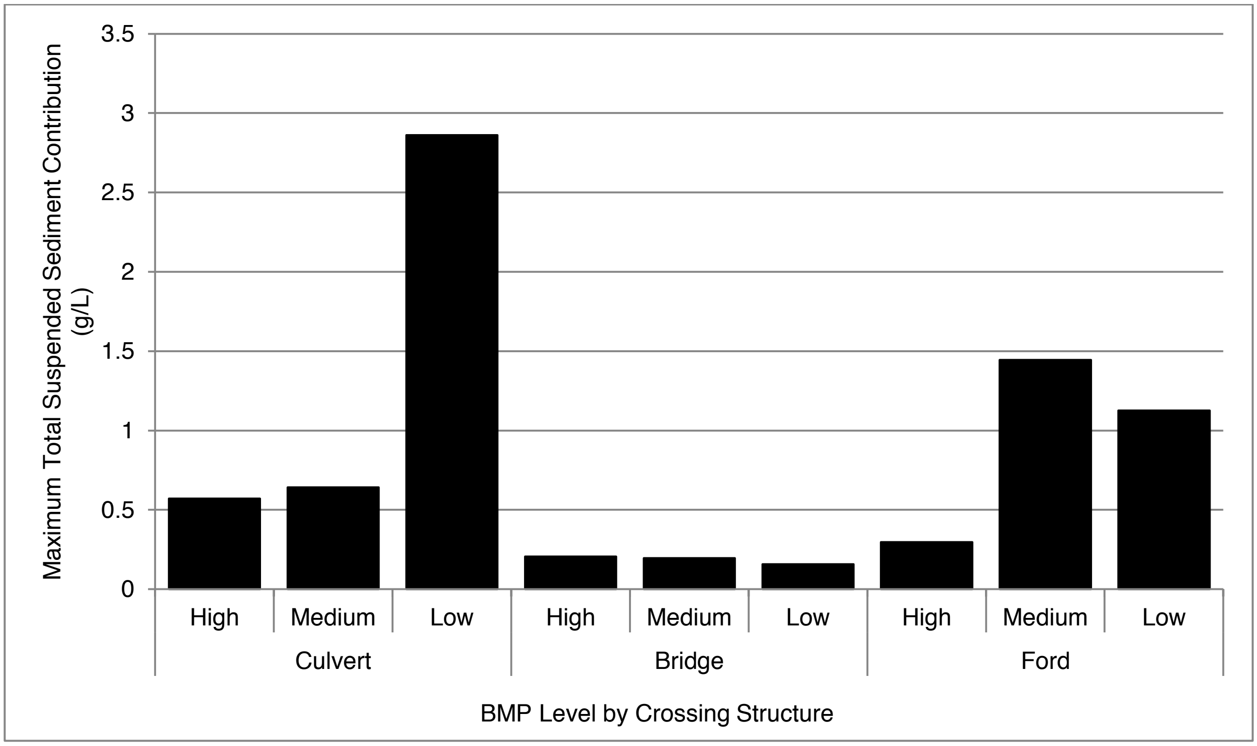

Figure 1--Maximum total suspended sediment concentration (g/L) by BMP level and crossing structure for all levels of rainfall simulation.

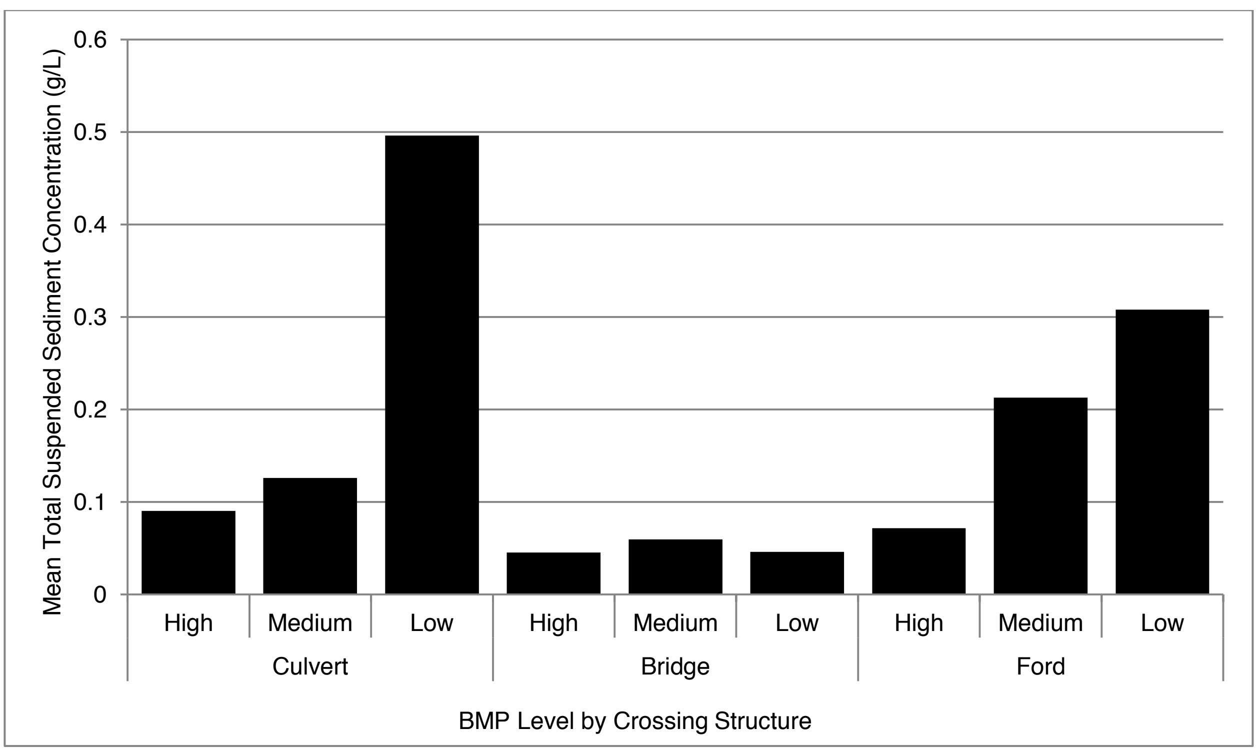

Figure 2--Mean downstream total suspended sediment concentration (g/L) by BMP level and crossing structure for all levels of rainfall simulation.

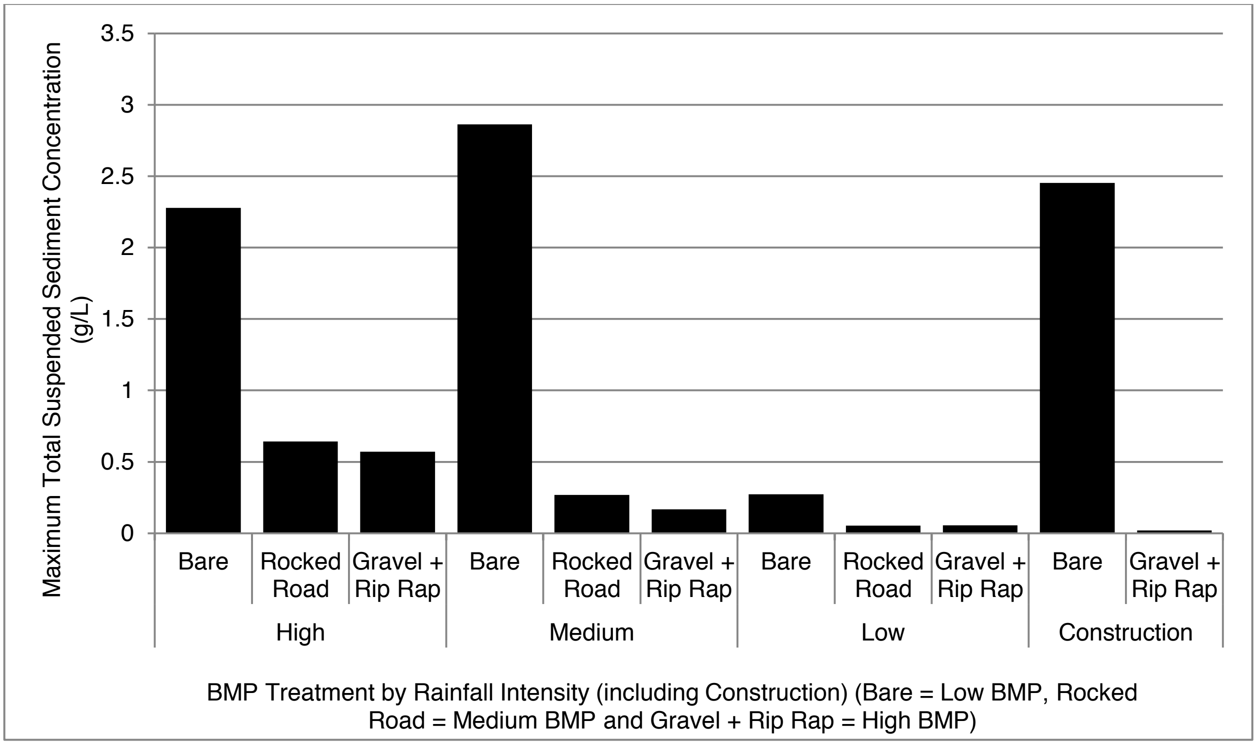

Figure 3--Maximum total suspended sediment concentration (g/L) at the culvert crossing by BMP treatment and rainfall intensity, including the construction phase (no rainfall during construction phase).

intensity was 2.3 g/L and the Medium rainfall intensity was 2.9 g/L. For all three levels of rainfall simulation, the greatest TSS concentration was observed during the Bare (Low) BMP treatment with the Rocked Road (Medium) BMP and Gravel + Rip Rap (High) BMP treatments resulting in reduced TSS concentrations, though the difference between Low BMPs and Medium BMPs was greater than the difference between Medium BMPs and High BMPs. (fig. 3). The mean TSS concentrations followed a similar pattern; however, at the High rainfall intensity the difference between the mean TSS concentration for the Rocked Road (Medium) BMP level and Gravel + Rip Rap (High) BMP was < 0.03 g/L, with the High BMP treatment resulting in a slightly greater mean (fig. 4). The maximum and mean TSS concentration was greatest during the Medium rainfall simulation. This was likely due to overland flow that filled a depression near the inlet of the culvert. Near the end of the Medium rainfall simulation on the Low BMP treatment, the water in the depression had overtopped the stream bank and entered the stream. This was not the result of a BMP failure; rather, it was the result of construction practices and the legacy road alignment, as the depression was formed near the outlet of a water turnout on the legacy road which had been abandoned when the culvert was built.

The maximum sediment concentration for the rainfall simulations occurred during the Low BMP treatment as did the maximum sediment delivery. The construction phase resulted in the introduction of 3.8 tons of sediment into the stream over an 8-hour construction period, compared to 2.5 tons during the 30-minute Low BMP High rainfall intensity simulation, and 4.1 tons of sediment being produced during the Medium rainfall intensity simulation on the Low BMP treatment (fig. 5). The 2.5 tons produced during the Low BMP and High rainfall simulation, as well as the 3.8 tons produced during construction phase, are comparable to the 2.8 tons produced during rainfall events and 3.5 tons produced during construction activities on a culvert in the Central Highland of Australia (Lane and Sheridan, 2002). Although the TSS produced during rainfall simulation are similar to the 2.8 tons produced during Lane and

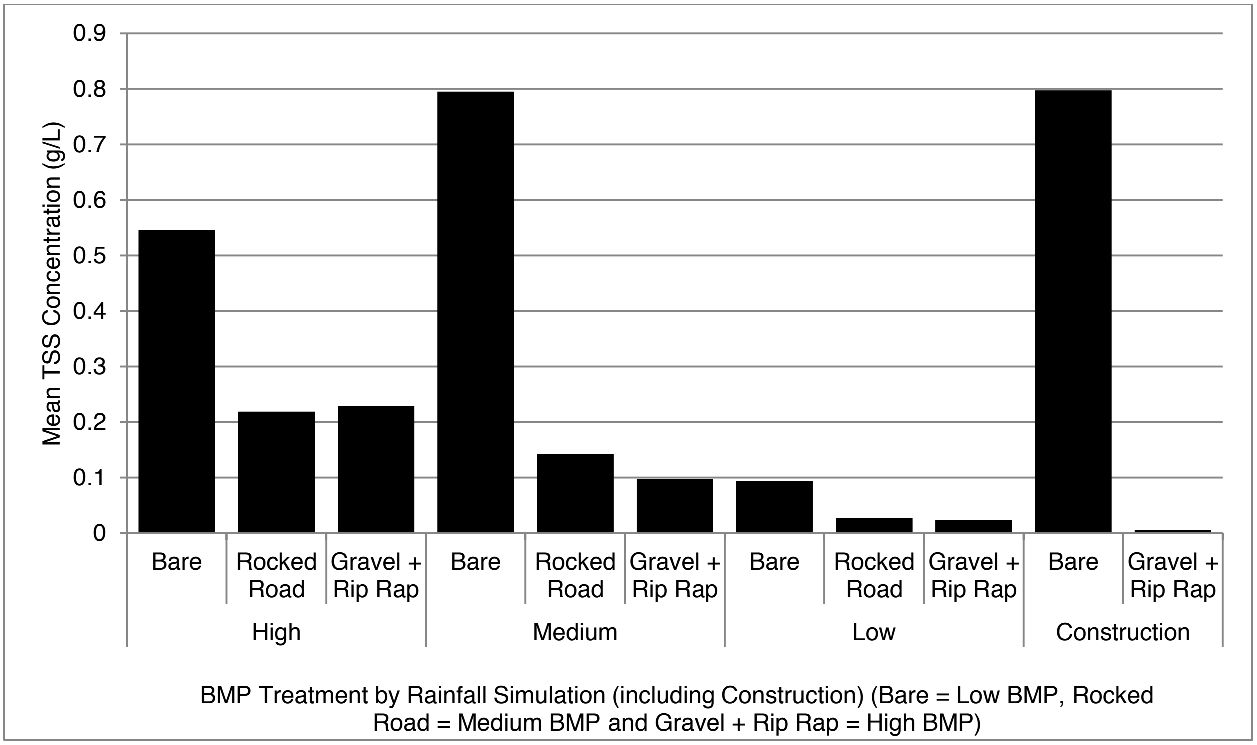

Figure 4--Mean total suspended sediment concentration (g/L) at the culvert crossing by BMP treatment and rainfall intensity, including the construction phase (no rainfall during construction phase).

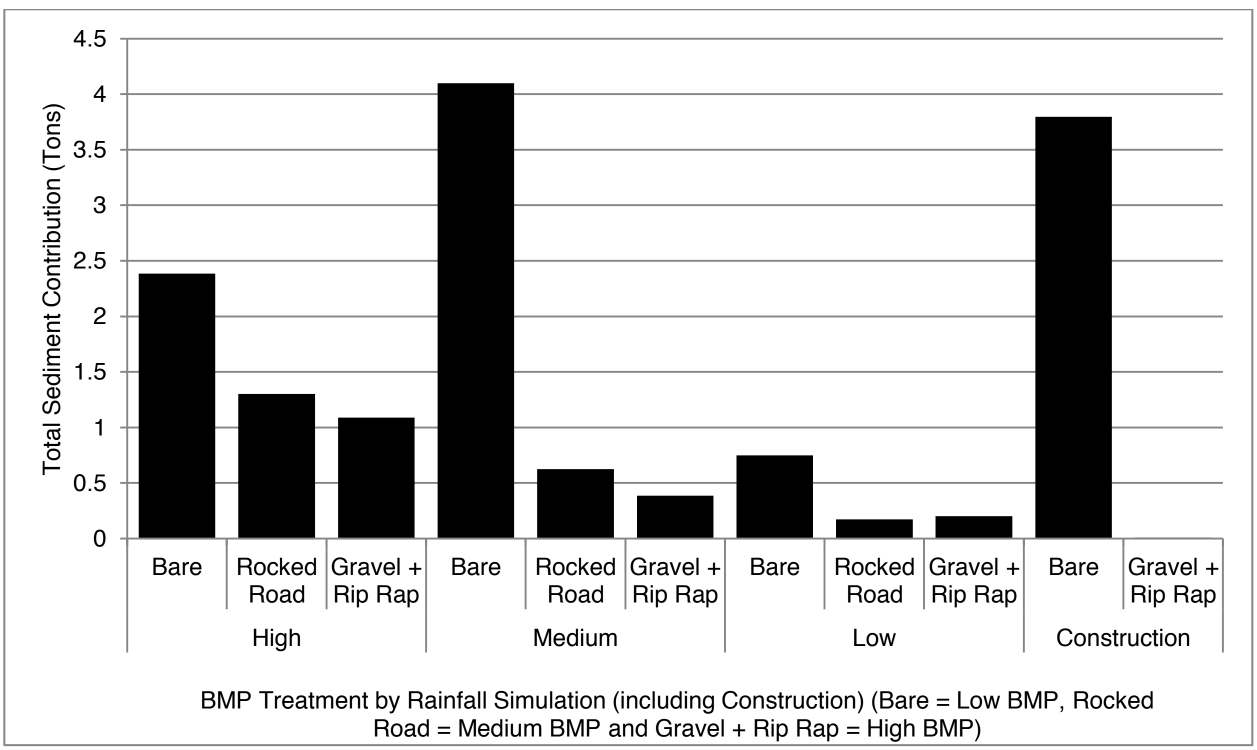

Figure 5--Sum of total sediment contribution (tons) for the culvert crossing during simulations and construction activities by BMP treatment and rainfall intensity (including construction).

Sheridan's (2002) experiment, their monitoring covered a timeframe of approximately 6 months. The single-event sediment production during the rainfall simulation could be due to differences in site-specific factors such as soil type, soil cover, and the time since soil disturbance. Total sediment production (tons) decreased with an increase in BMPs implemented for the Medium and High rainfall simulations while the difference between the Medium and High BMP treatments at the Low rainfall simulation was minimal. Additionally, the initial construction resulted in a large mass of sediment being introduced to the channel due to the required excavation of the channel. The addition of rip-rap to the fill slopes resulted in minimal sedimentation due to the rock being placed around the culvert and stream channel by hand. The use of larger equipment could result in increased sedimentation during this phase of construction.

The rainfall simulation experiment on the Culvert produced more sediment than the Ford or Bridge, suggesting that further investigation should focus on BMPs for culverts. Initial construction activities contributed almost 4 tons of sediment during an 8-hour construction period. Subsequent rainfall simulations resulted in a maximum sediment contribution of 4.1 tons during a 30-minute rainfall simulation. The construction resulted in a large sediment contribution due to the need to excavate the channel to place the culvert at the proper grade in the stream. The 4.1 tons of sediment were produced during the Low BMP simulations with no soil surface cover on the road surface and fill slopes. The addition of rock on the running surface and fill slopes further decreased the sediment contribution from the Medium and High BMP simulations. The addition of rock on the fill slopes and running surface will also facilitate a longer usable life for the crossing structure. The additional erosion that would be present without the rock, seed, and mulch would require additional maintenance as the running surface and fill slopes begin to erode. The use of rock will not only reduce the potential sediment contribution but could also reduce future maintenance needs of the crossing. The nature of constricting a stream to a culvert pipe will always require maintenance and attention to prevent the pipe from clogging and subsequent failure of the crossing which could result in a much greater sediment contribution. The current BMPs for stream crossings are effective at reducing sedimentation when compared to stream crossing structures with no erosion control measures (BMPs) in place. Additional BMPs may further reduce erosion; however, the cost of the additional BMPs must be compared to the stream health benefits obtained by further reduction erosion, and sedimentation.

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.

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.

Briggs, R.D. 1998. Compliance with forestry best management practices in Maine. Northern Journal of Applied Forestry. 15(2): 57-68.

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

Lane, P.N.J.; Sheridan, G.J. 2002. Impact of an unsealed forest road stream crossing: water quality and sediment sources. Hydrological Processes. 16: 2599-2612.

Moore, R. 2004. Introduction to salt dilution gauging for streamflow measurement: part 1. Streamline Watershed Management Bulletin. 7(4): 20-23.

Moore, R. 2005. Slug injection using salt in solution. Streamline Watershed Management Bulletin. 8: 1-6.

Shepard, J.P. 2006. Water quality protection in bioenergy production: the US system of forestry best management practices. Biomass & Bioenergy. 30: 378-384.

Taylor, S.E.; Rummer, R.B.; Yoo, K.H. [and others]. 1999. What we know -- and don't know -- about water quality at stream crossings. Journal of Forestry. 97: 12-17.

Wynn, T.M.; Mostaghimi, S.; Frazee, J.W. [and others]. 2000. Effects of forest harvesting best management practices on surface water quality in the Virginia Coastal Plain. Transactions of the ASAE. 43: 927-936.

FORESTRY BEST MANAGEMENT PRACTICES AND SEDIMENT CONTROL AT SKIDDER STREAM CROSSINGS

Laura R. Wear, W. Michael Aust, M. Chad Bolding, Brian D. Strahm, and C. Andrew Dolloff[1]

Abstract--Stream crossings for skid trails have high sediment delivery ratios. Forestry Best Management Practices (BMPs) have proven to be effective for erosion control, but few studies have quantified the impact of various levels of BMPs on sedimentation. In this study, three skid-trail stream-crossing BMP treatments were installed on nine operational stream crossings (three replications) to evaluate the degree of sediment control associated with the different treatments. Treatments were: (1) slash, (2) mulch, and (3) mulch plus silt fence. Upstream and downstream water samples were collected daily at the stream crossings for 1 year following BMP installation. Samples were evaluated for total suspended solids. Both slash and mulch treatments applied to the stream-crossing approaches after removal of temporary skidder bridges were effective at reducing the amount of sediment entering the stream after harvest. The mulch plus silt-fence treatment was the most expensive treatment, yet it allowed more sediment to enter the stream at the approach due to silt-fence installation disturbances. Thus BMP related disturbances should be minimized adjacent to a stream bank.

INTRODUCTION

Forest roads and skid trails can cause significant increases in erosion and sedimentation (McBroom and others 2008, Patric 1976, Swift and Burns 1999). Therefore most forestry best management practices (BMPs) were developed with a focus on erosion associated with silvicultural transportation networks, including roads, decks, skid trails, and stream crossings (Anderson and Lockaby 2011, Aust and Blinn 2004). Typical BMPs for roads, skid trails, and logging decks include proper planning and location, use of streamside management zones (SMZs) or buffer strips, control of grade, control of water, surfacing, road or trail closure to minimize soil disturbance, and revegetation following harvesting (Aust and Blinn 2004, Ice and others 2010, Shepard 2006, Swift 1985).

Stream crossings are a particularly important potential source of sediment because stream crossings interrupt SMZs and may serve as channels for sediment to enter streams (Aust and others 2011, Litschert and MacDonald 2009, MacDonald and Coe 2007, Swift 1985, Witmer and others 2009). Therefore, sediment concentrations are often increased below stream crossings (Croke and others 2005, Lane and Sheridan 2002). Sediment contributions from stream crossings have been associated with road densities (Schoenholtz 2004), time since road construction (Luce and Black 1999, Schoenholtz 2004), stream-crossing types, and adequacy of approach BMPs (Aust and others 2011).

During annual BMP audits, the Virginia Department of Forestry (VDOF) has repeatedly identified stream crossings as an area where BMP compliance could be improved (VDOF 2008). However, methods of stabilization are not explained in the Virginia BMP manual, and closure techniques are not specified for stream crossings in many of the state BMP manuals in the South.

Additionally, forest stream crossings and associated BMPs have been the central issues in court cases appearing before the U.S. Ninth Circuit Court of Appeals and the U.S. Supreme Court (Boston 2012). The issue is not resolved, yet it does emphasize the need for additional research regarding the effects of forest-road stream-crossing BMPs on sediment (Anderson and Lockaby 2011). The objectives of this research were to evaluate three levels of skid-trail stream-crossing closure BMPs (slash, mulch, and mulch + silt fence) on stream sediment levels and to quantify the costs of the BMP treatments.

MATERIALS AND METHODS

Study Sites

Nine operational skidder stream crossings that used steel-panel skidder bridges to span Piedmont streams were evaluated for 1 year

[1]Graduate Research Assistant, Professor, Associate Professor, and Assistant Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061; and Team Leader, USDA Forest Service, Southern Research Station, Blacksburg, VA 24061.

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after the temporary crossings were closed. Stands were MeadWestvaco-managed 18- to 25-year-old loblolly pine (*Pinus taeda* L.) plantations that were clearcut between fall 2010 and spring 2011. All stream-crossing locations were specified before harvesting by a professional forester in order to minimize the number of crossings needed. The steel-paneled bridges ranged from 7.3 to 9.7 m in length, and three 1-m-wide panels (3-m wide total) were used on each crossing. Panels were installed and removed with rubber-tired grapple skidders, as is common operationally. Standard 15-m SMZs were flagged for each side of the streams, but actual SMZs ranged from 13 to 45 m in width.

Mean annual precipitation values ranged from 1070 to 1140 mm year^{-1} (NRCS 2013). Rolling topography had average side slopes of 15 percent ranging up to 30 percent. Stream crossings were on first- and second-order intermittent streams having watershed sizes from 3 to 39 ha above the crossing points. Sites had similar soils, Hapudults and ultic Hapludalfs (NRCS 2013). All sites had a history of prior agricultural disturbance as is typical of the region (Jackson and others 2005, Nutter and Douglass 1978).

Treatments

After harvesting, skidder bridges were removed, and three BMP closure treatments were replicated three times, for a total of nine stream crossings having 18 approaches; i.e., BMP treatments were the same on each side of the stream. All stream crossings had waterbars, which is the minimal recommended BMP level in Virginia (VDOF 2011). The stream-crossing closure treatments were: (1) Slash: a rubber-tired grapple skidder placed logging slash (tree limbs and tops) from slash piles on skid trail approaches to a depth of 0.25 to 1 m; (2) Mulch: fescue seed, fertilizer, lime, and straw mulch were spread on the approaches (not in the stream), with the mulch providing 100 percent coverage of bare soil. Each approach was covered with 10 bales of straw mulch, equating to 20 bales per crossing; and (3) Mulch + silt fence: silt fences were installed in trenches < 1 m from the stream bank on both sides of the stream channel. Fescue seed, fertilizer, lime, and straw mulch were spread on the approaches with the mulch providing 100 percent coverage of bare soil. Each approach was covered with 10 bales of mulch, equating to 20 bales per crossing.

Sediment Sampling

At each stream crossing, two automated water samplers, either ISCO 3700 (Teledyne Isco, Inc., Lincoln, NE) or Sigma 900MAX (Hach Company, Loveland, CO) were installed. One automated sampler was positioned approximately 10 m upstream, and the second was positioned 10 m downstream from the crossing. Equipment safety and logistics required that water samplers were installed after harvesting but before the BMP closure treatments were applied (which ranged from a period of 1 to 10 days depending on the location). All automated water samplers collected one 500-mL sample per day. Samples were retrieved every 3 weeks and analyzed for total suspended solids (TSS) using the method outlined by Eaton and others (2005). Data collection continued for 1 year following harvesting. Daily precipitation data were collected from National Oceanic and Atmospheric Administration (NOAA) weather stations near each tract.

Treatment costs were recorded and reported by the MeadWestvaco forester responsible for overseeing the BMP installation. Costs included both materials and labor. The slash treatment did not require a material cost, so costs were based on labor and machine time only. Costs were reported as averages for each treatment.

Statistical Analysis

Statistical analyses used rain events as statistical blocks in order to control TSS variation at different rainfall intensities as suggested by Clinton and Vose (2003) in a similar forest road study. Four rainfall categories were established by dividing the daily rainfall data into quartiles above zero and then combining the lowest category with the days with no rain. The categories were: low = 0.00-1.0 mm; medium = 1.1-4.0 mm; high = 4.1-10.00 mm; and maximum > 10.0 mm. A daily TSS percent change value was calculated for analysis using the following equation:

Daily TSS percent change = [(Downstream TSS – Upstream TSS)/Upstream TSS] x 100 (1)

Data were analyzed for statistical significance using JMP Statistical Discovery Software (JMP Version 9, SAS Institute, Cary, NC). Data were not normally distributed; thus, non-parametric tests were used. Both the Kruskal-Wallis test (Ott and Longnecker 2010a) and the Wilcoxon test (Ott and Longnecker 2010b) were used to detect treatment differences. Physical features of the stream-crossing approaches were also measured and analyzed for significance with a Pearson's correlation matrix.

RESULTS AND DISCUSSION

Total Suspended Solids

Treatment performance rank is indicated by the Kruskal-Wallis statistical test (table 1). Higher scores (score mean values) indicate higher sediment values downstream relative to upstream values. The Wilcoxon tests indicate treatment differences between each paired treatment at each rainfall category (table 2). The rainfall categories that displayed significant differences between treatments were low, medium, and high (in the Kruskal-Wallis test). Slash performed better than the other two treatments with regard to sediment reduction at the low rainfall category. However, the medium, high, and maximum rainfall categories indicated that the slash and mulch treatments were statistically the same, while they both were different than the mulch + silt-fence treatment. These results indicate that the slash and mulch treatments performed better than the mulch + silt-fence treatment. Although silt-fence installation is a proven BMP for reducing silt-sized and larger sediment (Robichaud and Brown 2002), its installation requires disturbance. Silt fences were installed adjacent to streams, and the installation disturbances apparently introduced sediment. It should also be noted that silt-fence failure could be related to the high clay content commonly found in the Piedmont of Virginia. Clay soil particles are smaller than silt particles and therefore have the ability to pass through silt fence. These results indicate the need to minimize disturbances within the SMZs even while installing BMPs.

BMP Treatment Costs

BMP treatment costs were reported by the forester responsible for overseeing the BMP treatment installation (table 3). The slash treatment average costs were $120 per stream crossing. This assumes that logging slash is available on site, and that it is moved after harvest has been completed. The costs are based on 2 hours of operator and machine time for slash application. This cost could be reduced if slash was spread on stream-crossing approaches during normal logging operations. The mulch treatment average costs, including material and labor, were $280 per stream crossing. Mulch + silt-fence applications were the most expensive treatment, costing an average of $345 per stream crossing, including materials and labor. These costs are lower but in the same order of magnitude as those reported recently by McKee and others (2012), who surveyed 70 logging contractors and reported the costs of stream-crossing BMPs ranged from $533 to $655 across Virginia.

CONCLUSIONS

Practically all forestry BMP recommendations recognize that stream-crossing portions of skid trails are where sediment delivery has the greatest potential to occur. However, few studies have specifically addressed BMP efficacy for closing stream crossings (Anderson and Lockaby 2011). Our results indicate that sedimentation is reduced by applications of the slash or seed and mulch treatment to temporary skidder stream-crossing approaches. On these sites, slash treatments cost less and would be more desirable. Mulch and seed is another viable option where slash is less readily available, but it can cost more. Either slash or mulch provided immediate coverage and erosion control at the stream-crossing approach. This study indicates that the nearly 100 percent soil coverage provided by the slash or mulch treatments were more important for erosion control than the slope of the approach (up to 18 percent). Slash was the most cost-effective option. These results correspond well to the bladed skid trail and overland skid trail closure results found by Wade and others (2012) and Sawyers and others (2012).

Table 1--Results of the Kruskal-Wallis Test. The score mean values show the rank in which the treatments performed. Higher scores (score mean values) indicate a higher percentage of sediment downstream, compared to other treatments. The asterisk (*) in the P-value column denotes significant differences between treatments at the respective rainfall category, at α = 0.05. Score means not connected by the same letter are significantly different

Daily rainfall category	Chi square	P-value	Treatment	N	Score mean
Low	14.9433	0.0006*	Slash	245	193.27 a
0.0-1.0 mm			Mulch	96	231.95 b
			Mulch + silt fence	83	246.77 b
Medium	9.0407	0.0109*	Slash	27	24.14 a
1.11- 4.0 mm			Mulch	16	26.25 a
			Mulch + silt fence	13	40.30 b
High	11.7111	0.0029*	Slash	37	38.00 a
4.1-10.0 mm			Mulch	31	43.90 a
			Mulch + silt fence	23	61.69 b
Maximum	4.2202	0.1212	Slash	43	42.25 a
> 10 mm			Mulch	24	40.95 a
			Mulch + silt fence	22	54.77 a

Table 2-- Results of the Wilcoxon test. Each treatment was compared with all other treatments within each rainfall category. The asterisk (*) in the P-value column denotes significant differences between the two treatments being compared at α = 0.10. Score mean difference is the difference between the score means from the Kruskal-Wallis test

Daily rainfall category	Treatment	vs.	Treatment	Score mean difference	Standard error difference	Z	P-value
Low	Mulch		Slash	30.567	11.870	2.575	0.0100*
0.00 – 1.0 mm	Mulch + silt fence		Slash	41.969	12.044	3.485	0.0005*
	Mulch + silt fence		Mulch	5.425	7.766	0.699	0.4848
Medium	Mulch + silt fence		Slash	10.826	3.946	2.743	0.0061*
1.1 – 4.0 mm	Mulch + silt fence		Mulch	8.016	3.179	2.521	0.0117*
	Mulch		Slash	2.140	3.961	0.540	0.5891
High	Mulch + silt fence		Slash	15.440	4.637	3.329	0.0009*
4.1 – 10.0 mm	Mulch + silt fence		Mulch	10.678	4.329	2.466	0.0136*
	Mulch		Slash	4.505	4.814	0.935	0.3494
Maximum	Mulch + silt fence		Slash	9.378	4.956	1.892	0.0584*
> 10.0 mm	Mulch + silt fence		Mulch	6.751	3.961	1.704	0.0883*
	Mulch		Slash	-1.201	4.964	-0.241	0.8088

Table 3-- Treatment costs per stream crossing as reported by the logging contractors

Treatment	Materials	Material cost	Labor	Labor Cost	Total cost per stream crossing
		$		*$*	*$*
Slash	Logging slash	n/a	Skidder machine time (2 hours)	120	120
Mulch	Straw mulch (20 bales)	100	Dozer machine time	90	
	Lime	5	Manual labor (2 hours)	80	
	Fertilizer and seed	5			280
Mulch + silt fence	Straw mulch (20 bales)	100	Dozer machine time	90	
	Lime	5	Manual labor (3 hours)	120	
	Fertilizer and seed	5			
	Silt fence	25			345

Applying either slash or mulch with seed to the stream-crossing approaches during harvest closure will reduce the amount of sediment that could otherwise enter the stream at these sensitive areas. Skidder stream crossings can be effectively closed, as long as coverage of bare soil is completed immediately following (or during) harvest. Minimal stream sedimentation can be achieved with the appropriate combination of stream-crossing BMPs.

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LITERATURE CITED

Anderson, C.J.; Lockaby, B.G. 2011. Research gaps related to forest management and stream sediment in the United States. Environmental Management. 47(1): 303-313.

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: 5-36.

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

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.

Clinton, B.D.; Vose, J.M. 2003. Differences in surface water quality draining four road surface types in the southern Appalachians. Southern Journal of Applied Forestry. 27(2): 100-106.

Croke, J.; Mockler, S.; Fogarty, P.; Takken, I. 2005. Sediment concentration changes in runoff pathways from a forest road network and the resultant spatial pattern of catchment connectivity. Geomorphology. 68(3-4): 257-268.

Eaton, A.D.; Clesceri, L.S.; Rice, E.W. [and others]. 2005. Total suspended solids dried at 103-105 °C. In: Standard methods for the examination of water and wastewater. Washington, DC: American Public Health Association. 1368 p.

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

Jackson, C.R.; Martin, J.K.; Leigh, D.S.; West, L.T. 2005. A southeastern piedmont watershed budget: evidence for a multi-millennial agricultural legacy. Journal of Soil and Water Conservation. 60(6): 298–310.

Lane, P.N.J.; Sheridan, G.J. 2002. Impact of an unsealed forest road stream crossing: water quality and sediment sources. Hydrologic Processes. 16(13): 2599–2612.

Litschert, S.E.; MacDonald, L.H. 2009. Frequency and characteristics of sediment delivery pathways from forest harvest units to streams. Forest Ecology and Management. 259(2): 143-150.

Luce, C.H.; Black, T.A. 1999. Sediment production from forest roads in western Oregon. Water Resources Research. 35(8): 2561-2570.

MacDonald, L.H.D.; Coe, D. 2007. Influence of headwater streams on downstream reaches in forested areas. Forest Science. 53(2): 148-168.

McBroom, M.W.; Beasley, R.S.; Chang, M.; Ice, G.G. 2008. Storm runoff and sediment losses from forest clearcutting and stand re-establishment with best management practices in East Texas, USA. Hydrologic Processes. 22(10): 1509-1522.

McKee, S.E.; Shenk, L.A.; Bolding, M.C.; Aust, W.M. 2012. Stream crossing methods, costs, and closure best management practices for Virginia loggers. Southern Journal of Applied Forestry. 36(1): 33-37.

NRCS [Natural Resources Conservation Service]. 2013. Web soil survey. Washington, DC: U.S. Department of Agriculture Natural Resources Conservation Service. http://websoilsurvey.nrcs.usda.gov/. [Date accessed: December 19, 2013].

Nutter, W.L.; Douglass, J.E. 1978. Consequences of harvesting and site preparation in the Piedmont. In: Tippin, T., ed. Proceedings: symposium on principles of maintaining productivity on prepared sites. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 65-72.

Ott, R.L.; Longnecker, M. 2010a. The Kruskal-Wallis test. In: An introduction to statistical methods and data analysis. 6th ed. Belmont, CA: Brooks/Cole: 428-431.

Ott, R.L.; Longnecker, M. 2010b. The Wilcoxon rank sum test. In: An introduction to statistical methods and data analysis. 6th ed. Belmont, CA: Brooks/Cole: 305-314.

Patric, J.H. 1976. Soil erosion in the eastern forest. Journal of Forestry. 74(10): 671-677.

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

Sawyers, B.C.; Bolding, M.C.; Aust, W.M.; Lakel, W.A., III. 2012. Effectiveness and implementation costs of overland skid trail closure techniques in the Virginia Piedmont. Journal of Soil and Water Conservation. 67(4): 300-310.

Schoenholtz, S.H. 2004. Impacts of forest management on water quality. In: Burley, J.; Evans, J.; Youngquist, J., eds. Encyclopedia of forest sciences. Oxford: Elsevier: 377-388

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.

Swift, L.W., Jr. 1985. Forest road design to minimize erosion in the southern Appalachians. In: Blackmon, B.G. Proceedings of forest and water quality: a mid-south symposium. Little Rock, AR: University of Arkansas: 141-151.

Swift , L.W., Jr.; Burns, R.G. 1999. The three Rs of roads: redesign, reconstruction, restoration. Journal of Forestry. 97(8): 40-44.

VDOF [Virginia Department of Forestry]. 2008. Silvicultural best management practices implementation monitoring for Virginia: 2007-2008. Charlottesville, VA: Virginia Department of Forestry. 8 p.

VDOF [Virginia Department of Forestry]. 2011. Virginia's forestry best management practices for water quality. Technical Manual. 5th ed. Charlottesville, VA: Virginia Department of Forestry. 216 p.

Wade, C.R.; Bolding, M.C.; Aust, W.M. [and others]. 2012. Comparing sediment trap data with the USLE-Forest, RUSLE2, and WEPP-Road erosion models for evaluation of bladed skid trails BMPs. Transactions of the American Society of Agricultural and Biological Engineers. 55(2): 403-414.

Witmer, P.L.; Stewart, P.M.; Metcalf, C.K. 2009. Development and use of a sedimentation risk index for unpaved road-stream crossings in the Choctawhatchee watershed. Journal of the American Water Resources Association. 45(3): 734-747.

Pine Regeneration and Genetics

Moderator:

Gregory Ruark
USDA Forest Service
Southern Research Station

DEVELOPMENT OF UNDERSTORY TREE VEGETATION AFTER THINNING NATURALLY OCCURRING SHORTLEAF PINE FORESTS

Anup K.C., Thomas B. Lynch, Douglas Stevenson, Duncan Wilson, James M. Guldin, Bob Heinemann, Randy Holeman, Dennis Wilson, and Keith Anderson[1]

During the 25 years since establishment of more than 200 growth study plots in even-aged, naturally regenerated shortleaf pine (*Pinus echinata* Mill.) forests, there has been considerable development of hardwood understory trees, shrubs, and some shortleaf pine regeneration. During the period from 1985-1987, even-aged shortleaf pine growth-study plots were established on the Ozark and Ouachita National Forests in western Arkansas and eastern Oklahoma. Plots were established in combinations of four levels of site index (50, 60, 70, and 80 feet at base age 50 years), four levels of age (20, 40, 60, and 80 years), and thinned to four levels of basal area per acre (30, 60, 90, and 120 square feet per acre). At plot establishment, hardwoods were controlled by application of chemical herbicide by stem injection or equivalent method. In subsequent years, hardwood regrowth has occurred, and some plots contain shortleaf pine regeneration.

At the second remeasurement, 10 years after plot establishment, understory tree development was measured by establishing two 5-milacre plots within each 0.2-acre shortleaf pine growth-study plot. At the third and subsequent remeasurement, 15 years after plot establishment, four 5-milacre plots were established within each 0.2-acre shortleaf pine growth-study plot to assess understory tree development. The statistical analysis has shown that red maple (*Acer rubrum* L.), flowering dogwood (*Cornus florida* L.), winged elm (*Ulmus alata* Michx.), northern red oak (*Quercus rubra* L.), black oak (*Q. velutina* Lam.) and shortleaf pine are the most dominating understory species. An annual total understory basal area growth model shows that the understory vegetation basal area growth is negatively affected by the upper story basal area and upper story plot age and positively affected by the site index (p-value < 0.05). The proportion of total shortleaf pine understory basal area to understory hardwood basal area has decreased with time elapsed from initial thinning treatments. However, there are relatively good numbers of shortleaf regeneration stems in the understory on some plots. The study results have clearly shown that the growth and development of understory hardwood and shortleaf pine regeneration depend on the actual condition and density of upper-story shortleaf trees in the stand of interest.

[1]Graduate Research Assistant, Professor, Senior Research Specialist, and Assistant Professor, respectively, Oklahoma State University, Department of Natural Resource Ecology and Management, Stillwater, OK 74078; Supervisory Ecologist, USDA Forest Service, Southern Research Station, Hot Springs, AR 71902; and Senior Superintendent, Research Specialist, Research Specialist, and Forestry Technician, respectively, Oklahoma State University Kiamichi Forest Resources Center, Idabel, OK 74745.

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GROWTH PERFORMANCE OF LOBLOLLY, SHORTLEAF, AND PITCH X LOBLOLLY PINE HYBRID GROWING ALONG THE WESTERN MARGIN OF COMMERCIAL PINE RANGE

Dipesh K.C., Rodney E. Will, Thomas C. Hennessey, Thomas B. Lynch, Robert Heinemann, and Randal Holeman[1]

Expansion of the commercial pine range is one of the opportunities to improve forest production and counterbalance the loss of forest land to other uses. The potential genotypes for the purpose are fast-growing loblolly pine (*Pinus taeda* L.), the slower growing, but more drought tolerant shortleaf pine (*P. echinata* Mill.), and the more cold tolerant pitch x loblolly pine hybrid (*P. rigida* x *taeda)*. An uncertainty regarding the effect of potential climate change on the future commercial range of pine plantation also calls for the detailed information on survival and growth performance among the potential genotypes planted at the margin of the range.

Our objectives were to compare survival and growth of loblolly pine, shortleaf pine, and pitch x loblolly hybrid pine planted in southeastern Oklahoma. The study was established in spring 2002 at 4 sites in southeastern Oklahoma within the natural range of shortleaf pine and loblolly pine, as well as west and north of the loblolly pine natural range. Three replications were planted with 1-0 seedlings of loblolly pine, shortleaf pine and pitch x loblolly pine (F1 hybrids) at each location at an approximate density of 1,343 trees ha^{-1}. Height and diameter were measured periodically until age 10, except after 6th, 7th, and 8th growing seasons. After the 8th growing season, every other tree was selected in the loblolly and pitch x loblolly pine stands at the Idabel site for thinning. Survival of 8-year-old trees from Idabel were compared with 9-year-old trees from other sites; height and diameter of the genotypes across all sites were compared after the 10th growing season.

Average survival of loblolly, shortleaf, and pitch x loblolly hybrids were 70, 59, and 71 percent respectively. After 10 years, the loblolly pine was tallest (9.4 m), pitch x loblolly pine hybrid second tallest (8.3 m), and shortleaf the shortest (6.8 m). Diameter at breast height followed the same pattern as height with loblolly pine 16.5 cm, pitch x loblolly pine hybrid 14.2 cm, and shortleaf pine 11.8 cm. These results indicate that loblolly pine outperformed both shortleaf and pitch x loblolly pine hybrids when planted at the western edge of the commercial southern pine range.

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[1]Graduate Student, Associate Professor, Professor, and Professor, respectively, Oklahoma State University, Department of Natural Resource Ecology and Management, Stillwater, OK 74078; and Senior Superintendent and Research Specialist, respectively, Oklahoma State University Kiamichi Forest Resources Center, Idabel, OK 74745.

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LONGLEAF PINE GROWN IN VIRGINIA: A PROVENANCE TEST

Kurt H. Johnsen, Jerre L. Creighton, and Chris A. Maier[1]

In 2006 the Virginia Department of Forestry established a longleaf pine (*Pinus palustris* Mill.) provenance test on three sites near Richmond, VA, near the most northern native range of longleaf pine. Seedlings were grown in containers at the Virginia Department of Forestry New Kent Forestry Center during the 2005 growing season. The provenances originated from native trees in Virginia, a natural stand and a seed orchard in North Carolina, and natural stands in South Carolina, Georgia, Florida, Alabama, and Mississippi. The provenances were grown on three sites, two previous nursery sites that were treated with herbicide and watered periodically during year one (New Kent -- 37° 25' N, 77° 01' W and Garland Grey -- 36° 51' N, 77° 10' W) and a cut-over site (Sandy Point -- 37° 40' N, 76° 55 W) that was mowed once in year one and not watered. Trees were measured for survival and height growth in years three, five and seven.

The sites varied widely in survival at age seven: New Kent (94 percent), Garland Grey (67 percent), and Sandy Point (57 percent). Height growth at Sandy Point was lowest at all ages: at age 7, mean height was 3.9, 3.9, and 2.7 m for trees at New Kent, Garland Grey, and Sandy Point, respectively. At age three, growth was greatest for the VA trees and decreased in a mostly clinal pattern (generally decreasing height growth from north to south). By age 5, the heights of the provenances on the nursery sites (New Kent and Garland Grey) had largely converged while the clinal pattern was maintained on the lower productivity Sandy Point site. After the seventh growing season, no provenance differences in height were observed on any site.

At the end of the fifth growing season, foliage was collected for analysis of carbon isotope discrimination ($\Delta^{13}C$). Measurement of the carbon isotope ratio of leaf tissue provides an assimilation-weighted average of the ratio of leaf intercellular CO_2 partial pressure (p_i/p_a) to atmospheric CO_2 partial pressure (Farquhar et al. 1989). This ratio (p_i/p_a) is important because it is a function of photosynthetic capacity and stomatal conductance. Changes in p_i/p_a are a function of changes in either, or both, photosynthetic capacity or stomatal conductance. Since leaf carbon isotope ratio provides information about processes integrated over the whole life of the leaf, it is particularly useful for examining subtle genetic differences in photosynthetic characteristics.

Water use efficiency (assessed via $\Delta^{13}C$) was greatest in the VA trees on all sites. A relationship between height and $\Delta^{13}C$ was observed (r = -0.55) only on the cut-over site (Figure 1), the site that had not been irrigated and had the lowest productivity. Theoretical (Farquhar et al. 1989) and empirical (Johnsen 1999) evidence indicates the faster growth associated with lower $\Delta^{13}C$ values was due to variation in photosynthesis. When the 25% best performing trees for height growth at age 5 and water use efficiency are identified (Figure 1), approximately half are from the Virginia seed source and the other half are from trees derived from more southern sources.

The current area of longleaf pine ecosystems in the southern United States is only approximately 3 percent of its former (pre-European) extent, and there is increasing interest in its restoration (Gilliam and Platt 2006). Such interest includes restoring such ecosystems in Virginia which

[1]Research Plant Physiologist, USDA Forest Service, Southern Research Station, Research Triangle Park, NC 27709; Research Program Manager, Virginia Department of Forestry, Charlottesville, VA 22903; and Research Biological Scientist, USDA Forest Service, Southern Research Station, Research Triangle Park, NC 27709.

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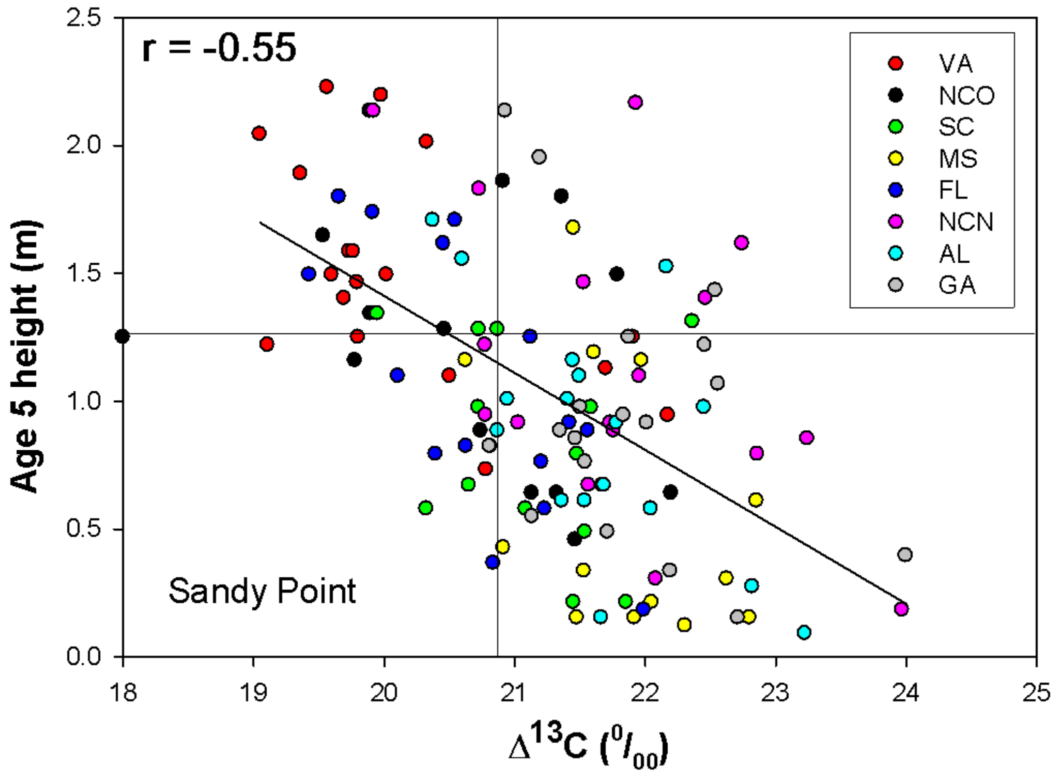

Figure 1--The relationship between height at age 5 and $\Delta^{13}C$ for all sample trees at the Sandy Point site. Water use efficiency decreases with increasing $\Delta^{13}C$. The top left quadrant contains data from the 25 percent of trees with the greatest height growth and lowest water use efficiency.

contains what are thought to be populations near or at the most northern part of its historical range. Clearly, the first step in restoring longleaf pine ecosystems where no longleaf currently exists is planting longleaf pine seedlings. Schmidtling and White (1990) showed that a Virginia seed source of longleaf pine performed poorly, relative to other sources, even at their most northern planting site. Here we found the Virginia source to display the fastest early growth and equal growth to the other sources at age 7. However it must be noted that our Virginia source was the result of seed collection from only 10 trees from one small population. In addition, molecular genetic analyses indicate that the 40% of the Virginia trees in our study may have one maternal parent (Craig Echt, personal communication). Thus, our results may not be representative.

Schmidtling (2001) indicated that longleaf pine seed sources can safely be moved so that the average annual minimum temperature of the planting site is within 2.8° C (5° F) of the seed sources' and that transfers up to 5.6° C (10° F) are possible although some growth potential may be lost. In our small study with extremely divergent seed sources grown at the northern edge of its native range, although southern sources grew slower initially, by age seven they were as tall as northern sources. These results may have ramifications with respect to restoring the species in light of climate change. Although it would likely be unwise to totally ignore native seed sources, a useful strategy may be to inter-mix a percentage of more southern seed sources, perhaps using or even exceeding Schmidling's (2001) more liberal suggestion, in longleaf pine restoration efforts in the north to increase adaptive potential. Of course, variation in flowering phenology will need to be understood so that gene flow can be maintained across the seed sources via something approximating random mating.

LITERATURE CITED

Farquhar, G.D; Ehleringer, J.R; Hubick, K.T. 1989. Carbon isotope discrimination and photosynthesis. Annual Review of Plant Physiology and Plant Molecular Biology. 40: 503-537.

Gilliam, F.S.; Platt, W.J. 2006. Conservation and restoration of the *Pinus palustris* ecosystem. Applied Vegetation Science. 9: 7-10.

Johnsen, K.H.; Flanagan, L.B.; Huber, D.A ; Major, J.E. 1999. Genetic variation in growth and carbon isotope discrimination in *Picea mariana*: analyses from a half-diallel mating design using field grown trees. Canadian Journal of Forest Research. 29: 1727-1735.

Schmidtling, R.C. 2001. Southern pines seed sources. Gen Tech . Rep. SRS-44. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 25 p.

Schmidtling, R.C.; White, T.L. 1990. Genetics and tree improvement of longleaf pine. In: Farrar, R.M., Jr., ed. Proceedings of the symposium on the management of longleaf pine. Gen. Tech. Rep. SO-75. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 114-127.

Snyder, E.B.; Namkoong, G. 1978. Inheritance in a diallel crossing experiment with longleaf pine. Res. Pap. S0-140. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 38 p.

EFFECTS OF GENETICS, MANAGEMENT INTENSITY, AND SEEDLING DENSITY ON EARLY STOCKING IN LOBLOLLY PINE

Scott D. Roberts, Randall J. Rousseau, and B. Landis Herrin[1]

Abstract--Rapid establishment and early tree growth can be key factors in successful plantation management. This generally entails planting good quality planting stock at a seedling density appropriate for the management objectives and then managing at an appropriate intensity with a goal of fully occupying the site as quickly as possible within the context of those objectives. We established a study to examine the performance of two varietal lines of loblolly pine (*Pinus taeda* L.) planted at three spacings and managed at two levels of intensity. After five growing seasons, management intensity and genetic variety were both significantly affecting tree height and diameter growth. This has resulted in significant differences in stocking, with relative density (%SDI_{max}) ranging from 4 to 31 percent. Initial tree spacing has yet to begin affecting individual tree growth, but higher relative densities in the tighter spacing suggest that these plots will soon start experiencing the effects of inter-tree competition.
Robinson, C.; Duinker, P.N.; Beazley, K.F. 2010. A conceptual framework for understanding, assessing, and mitigating ecological effects of forest roads. Environmental Reviews. 18: 61-86.

INTRODUCTION

Many of the decisions made at the time of stand establishment and early in the stand's rotation have important long-term consequences. For example, decisions concerning what planting stock to use, what seedling density to plant, and what silvicultural practices to employ prior to or immediately after planting can all have substantial effects on early tree and stand growth and set the stage for the entire silvicultural regime. These decisions affect rates of tree growth and therefore the time to reach full site occupancy, the timing of thinnings and final harvest, the quality of products produced, and other operational logistics of stand management. All of these decisions ultimately affect the overall economics of the management regime.

Growth rates of southern pine plantations have increased substantially over the past 30 to 40 years. Yields from loblolly pine (*Pinus taeda* L.) plantations now often exceed 300- and in some cases 400-cubic feet per acre per year (Fox and others 2007). A significant portion of this greater productivity is due to genetic gains that have been made in loblolly pine over the past half century. Achieving these gains, however, requires greater plantation management intensity involving improved site preparation, more effective competition control, better understanding of forest nutritional requirements, and greater attention to density management.

Formal efforts at genetic improvement of southern pines began in the early 1950s, and by the mid-1980s virtually all southern pine plantations were being established with seedlings produced from genetically improved seed. Volume gains from plantations established using first-generation improved seedlings generally ranged from 7 to 12 percent (Li and others 2000), with gains in harvest value estimated to be > 20 percent (Fox and others 2007). Over half of all southern pine plantations planted in the early 2000s were established with second-generation seedling stock, with volume gains estimated to range from 7 to 23 percent over first-generation plantations (Fox and others 2007; Li and others 1997, 1999, 2000; McKeand and others 2003, 2006a). Additional genetic gains have been realized through the deployment of the single half-sib family blocks (Duzan and Williams 1988, McKeand and others 2006b), and even further gains are possible through the utilization of full-sib families produced using mass-controlled pollination (MCP) techniques (Bramlett 2007). Volume gains from full-sib families over unimproved stock may be as high as 60 percent (Jansson and Li 2004).

The development of techniques for mass production of varietal (i.e., clonal) loblolly pine planting stock has led to the potential for even further plantations gains. The use of varietal planting stock has grown over the past decade,

[1]Professor, Professor, and Research Associate, respectively, Mississippi State University, College of Forest Resources, Department of Forestry, Mississippi State, MS 39762.

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although it is still not widely planted due to a variety of issues that need further investigation. The cost of varietal planting stock is high, currently much higher than other planting stock options. Field testing is needed to compare the performance of varietal stock to that of other planting stock options. In addition, studies are needed to identify the appropriate silvicultural regimes for maximizing gains when using varietal planting stock.

In 2008, we installed a study in central Mississippi to examine the effects of different silvicultural practices on the performance of varietal loblolly pine. The specific objectives of this study were to examine the performance of two selected loblolly pine varieties established at different initial seedling spacing and subjected to different levels of management intensity.

METHODS

In 2008, a study was established at the Mississippi State University Coastal Plain Branch Experiment Station Experiment Station near Newton, MS (32°20'19" N, 89°05' 51" W) to examine the effects of spacing and management intensity on the performance of two contrasting genotypes of loblolly pine. Soils at the site are a Prentiss very fine sandy loam. In September 2007, the site was treated with broadcast application of Glyphosate (64 ounces per acre) and in October was subsoiled to a depth of approximately 14 inches. The site received a second broadcast application of Glyphosate (32 ounces per acre) in March 2008 prior to planting of the seedlings. The site was hand planted with containerized stock in late April/early May 2008.

A 2x2x3 factorial split plot design was used. Main-effects treatments included two genetic varieties of loblolly pine and two levels of management intensity. Main-effects treatment plots were randomly applied and were split by three randomly applied planting spacings with trees within the spacing subplots planted in 64 tree blocks (8 by 8 tree spacing). The inner 36 trees constituted the measurement plots. Each combination of genotype, management intensity, and spacing was replicated four times.

Two varietal genotypes (i.e., clones) of loblolly pine were included in the study based on their putative divergent crown architectures. Produced by ArborGen, LLC, the varieties included one competitor ideotype (comp) characterized by a relatively wide crown form and a crop-tree ideotype (crop) with a more narrow, compact crown form. The management-intensity treatments included a standard intensity (low) and a high intensity (high). Along with the chemical site preparation and subsoiling, both high- and low-intensity plots received a broadcast application of Oustar® (10 ounces per acre) in year 1 for herbaceous competition control. The high-management-intensity plots also received tipmoth control in the form of a single SilvaShieldTM tablet (Bayer Environmental Science) in the planting hole at time of planting, herbaceous competition control in year 2 (1 ounce per acre of Escort®, 16 ounces per acre of Arrow®, 32 ounces per acre of Goal®), PTM™ insecticide (BASF Corp.) injected 3 to 6 inches deep in the soil adjacent to each tree (0.05 ounces active ingredient per tree) in years 2 and 3 for additional tipmoth control, and mowing between rows in year 3. The three initial tree spacings were 6 by 14 feet [519 trees per acre (tpa)], 9 by 14 feet (346 tpa), and 16 by 14 feet (194 tpa).

Seedling heights were measured immediately following planting. Survival has been assessed and heights measured annually through the first five growing seasons (2008-2012). Diameter at breast height (d.b.h.) has been measured on each tree starting in year 4. Reineke's Stand Density Index (SDI; Reineke 1933) was calculated for each plot at year 5 and expressed as a percentage of 450, the maximum SDI for loblolly pine (Dean and Baldwin 1993, Reineke 1933). A simple volume index was calculated for each stem from d.b.h. and total height using the formula for a cone. Plot volume was expressed as cubic feet per acre. We tested for treatment differences in mean tree size, relative density, and early volume production following five growing seasons. All reported treatment differences are based on a critical value of $\alpha = 0.05$.

RESULTS AND DISCUSSION

First-year survival was high across all treatments (table 1), with overall seedling survival over 94 percent. There were no differences in survival associated with the spacing treatment. Surprisingly, the low-intensity

Table 1--Survival at years 1 and 3, and height at years 1, 3, and 5 for two genetic varieties of loblolly pine planted at three different spacings and managed at two different management intensities on a site previously managed for agricultural production in central Mississippi

	---------Survival[a]---------		--------------------Mean height[a]-------------------		
	Year 1	Year 3	Year 1	Year 3	Year 5
	-------------*percent*------------		-------------------------*feet*------------------------		
Overall average	94.1	92.2	1.69	7.89	14.8
Initial spacing					
6 x 14 feet	94.8	93.6	1.69	8.07	15.1
9 x 14 feet	95.7	94.8	1.71	8.07	14.9
16 x 14 feet	91.8	88.2	1.66	7.52	14.5
Management intensity					
Low	96.3*	92.7	1.66	6.53	12.1
High	91.9	91.7	1.71	9.25*	17.5*
Genetic variety					
Crop	98.5*	98.0*	1.68	8.39*	15.9*
Competitor	89.7	86.3	1.69	7.38	13.7

[a]Values followed by an asterisk are significantly different from other values in the group at $\alpha = 0.05$.

management treatment had slightly greater survival than the high-intensity treatment. The crop-tree ideotype had better survival that the competitor ideotype. Little additional mortality was observed by year 3, with overall survival still exceeding 92 percent. The differences in survival associated with the management-intensity treatments that had been observed in year 1 had disappeared by year 3. However, the survival differences observed between genotypes remained and increased to nearly 12 percent. Initial spacing was still having no effect on survival through age 3, although one plot at the 16-foot spacing did experience excessive mortality due primarily to sawfly damage.

At age 1, seedlings averaged 1.7 feet in height with no significant treatment-related differences (table 1). By age 3, overall seedling heights averaged nearly 7.9 feet. Significant differences were observed between the management-intensity treatments, with the high-intensity plots averaging nearly 3 feet taller than the low-intensity plots. In addition to having better survival, the crop-tree ideotype had significantly greater mean heights at age 3, averaging about 1 foot taller than the competitor ideotype. Within each of the genotypes, differences in year-3 mean heights were four times greater on the high-intensity plots (1.6 feet) than on the low-intensity plots (0.4 feet). The spacing treatments were still having no effect on mean heights at age 3. By age 5, mean tree height across all treatments averaged nearly 15 feet. Genotype-related differences in height continued, with the advantage for the crop-tree ideotype increasing to over 2 feet. The biggest treatment-related effect on height continued to be associated with management intensity. Trees in the high-intensity treatment, with an average age 5 height of 17.5 feet, were over 5 feet greater than that of the low-intensity plots.

Average tree diameter across all treatments at age 5 was 3.3 inches, with treatments showing the same trends as observed for heights (table 2). Mean diameters for the crop-tree ideotype was nearly 1 inch greater than that of the competitor ideotype. Management intensity again showed the greatest impact on mean diameter. The high-intensity treatment had mean d.b.h. of 4.0 inches, nearly 1.5 inches greater than the low-intensity plots. Initial spacing has yet to start showing an effect on stem diameters.

Relative density (%SDI_{max}), which takes into account both stem density and mean tree size, showed significant treatment-related differences across all treatments (table 2). The mean relative density across all plots was 14 %SDI_{max}, with individual plots ranging from a low of 3 %SDI_{max} to a high of 38 %SDI_{max}. While the

Table 2--Mean d.b.h., relative density, and cubic foot volume per acre at year 5 for two genetic varieties of loblolly pine planted at three different spacings and managed at two different management intensities on a site previously managed for agricultural production in central Mississippi[a]

	D.b.h.	Rel. density[b]	Volume
	inches	*%max SD*	*ft^3 $acre^{-1}$*
Overall average	3.3	14	179
Initial spacing			
6 x 14 feet	3.2	20[a]	252[a]
9 x 14 feet	3.3	14[b]	177[b]
16 x 14 feet	3.3	8[c]	102[c]
Management intensity			
Low	2.6	10	89
High	4.0*	19*	264*
Genetic variety			
Crop	3.7*	17*	230*
Competitor	2.9	12	125

[a]Values followed by an asterisk or by different letters are significantly different from other values in the group at $\alpha = 0.05$.
[b]Relative density is expressed as the percentage of maximum SDI for loblolly pine.

spacing treatments did not differ in mean tree size, due to differences in stem density there were large differences in relative density. The average relative density of the 6-foot treatment plots was over 20 $\%SDI_{max}$, while the 16-foot plots averaged only 8 $\%SDI_{max}$. The crop-tree ideotype not only had slightly better survival but also better individual tree growth through 5 years and therefore had higher stocking. Relative density of the crop-tree ideotype plots averaged nearly 17 $\%SDI_{max}$ compared to less than 12 $\%SDI_{max}$ for the competitor ideotype. Once again, management intensity had a noticeable effect on stocking with the high-intensity plots averaging nearly 19 $\%SDI_{max}$ while the low-intensity plots averaged only 10 $\%SDI_{max}$.

Age 5 is earlier than most studies start assessing volume production, but it does provide another indication of how these stands are developing under these treatment combinations. Again, all treatments had significant impacts on volume production through age 5 (table 2). There was a significant interaction between spacing and management intensity for age 5 volume, ranging from an average of about 47 cubic feet per acre for the low-intensity treatment plots at the 16-foot spacing to as high as 380 cubic feet per acre for the high-intensity plots at the 6-foot spacing (fig. 1). There was also a significant interaction between genotype and management intensity (fig. 2). At low-management intensity, age-5 volume for the crop-tree ideotype average nearly 114 cubic feet per acre, over 50 cubic feet per acre higher than the competitor ideotype. Under the high-intensity treatment, volume averaged 346 cubic feet per acre for the crop-tree ideotype compared to less than 183 cubic feet per acre for the competitor ideotype.

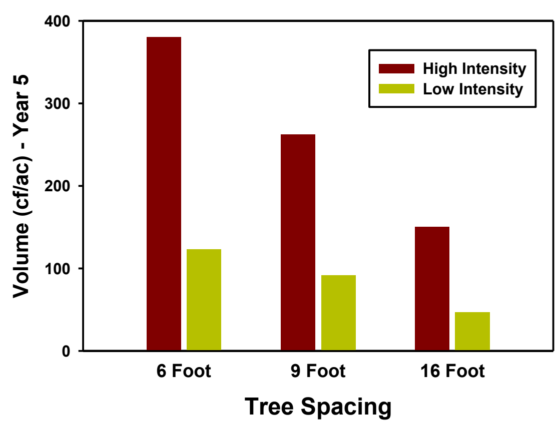

Figure 1--Volume per acre at year 5 for loblolly pine planted at three different spacings and managed at two levels of intensity. All trees were planted in rows 14 feet apart. Tree spacing refers to the distance between trees within rows. Values are means for two different loblolly pine varieties (i.e., clones).

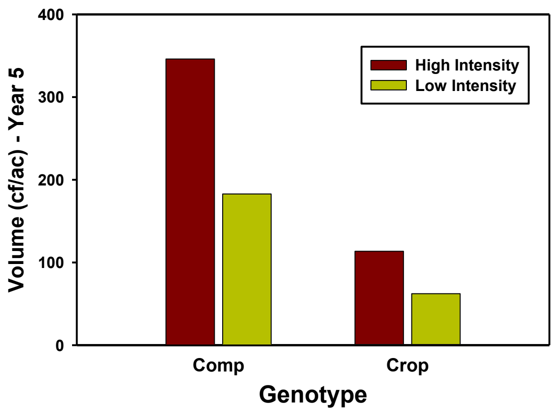

Figure 2--Volume per acre at year 5 for two loblolly pine genotypes managed at two levels of intensity. Values are means across three different initial tree spacing treatments. The 'Comp' genotype refers to a competitor ideotype. The 'Crop' genotype refers to a crop-tree ideotype.

Our results show that, through 5 years, the spacing treatments are not significantly affecting average tree heights or diameters although with initial density only ranging from 194 to 519 trees per acre, it is likely too early to expect much of a density effect on the trees. Significant differences in growth rates have been observed in the genotype treatment, with the crop-tree ideotype demonstrating greater individual tree and stand-level growth. Management intensity has clearly had the biggest impact to date on both tree- and stand-level growth through the first five growing seasons.

Treatment effects on tree growth rates and stem densities have resulted in tremendous variation in average stocking at age 5, ranging from a low of 4 $\%SDI_{max}$ for the competitor ideotype at 16-foot spacing with low-management intensity to a high of 31 $\%SDI_{max}$ for the crop-tree ideotype on 6-foot spacing with high-management intensity. Long (1985) suggests that for many species the onset of competitive interactions between trees begins at around 25 percent of maximum SDI, and that full site occupancy is achieved at about 35 percent of maximum SDI. If these values are accurate for loblolly pine, then a few of our treatment combinations are approaching the onset of inter-tree competition, and one set of treatments has already exceeded that level and is approaching full-site occupancy.

Differences in average tree growth rates and stocking rates have led to a 16-fold difference across treatment combinations in cubic foot volume production through age 5. Current volume approximations range from a low of 30- to a high of 480-cubic feet per acre. However, experience from numerous spacing studies suggests that the observed differences in stocking and volume production will begin to close as tree growth in the high-density stands begins to slow due to increased competition and the earlier onset of density-related mortality.

Our results show, as have many other studies, the potential growth benefits of improved genetics and intensive forest management. Perhaps just as important, even at this early age our study illustrates the importance, when achieving these benefits, of recognizing that stand development can be greatly accelerated and thus stocking rates and the timing of silvicultural activities need to be adjusted accordingly. Our treatment combinations exhibiting the lowest stocking rates are still several years from reaching the point where competitive interactions between trees will become noticeable. Conversely, stocking rates in our most rapidly developing treatment combinations are approaching, and in some cases have exceeded, levels where we might expect to see competitive effects on tree growth, although we were unable to detect any.

This study also illustrates another seemingly obvious point that is often overlooked. Landowners that are unwilling to invest in high-quality genetic planting stock and intensive management should not expect the same results as landowners that do. Even in our study, where both of the genotypes tested were highly selected, the results show that achieving the highest yields will not be possible if investments are not made in appropriate management intensity. Non-industrial private landowners in particular should not be lulled into believing that they are going to achieve the same yields and rotation lengths attained on intensively managed industrial plantations without a comparable investment in management inputs.

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LITERATURE CITED

Bramlett, D.L. 2007. Genetic gain from mass controlled pollination and topworking. Journal of Forestry. 105: 15-19.

Dean, T.J.; Baldwin, V.C., Jr. 1993. Using a density-management diagram to develop thinning schedules for loblolly pine plantations. Res. Pap. SO-275. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 7 p.

Duzan, H.W., Jr.; Williams, C.G. 1988. Matching loblolly pine families to regeneration sites. Southern Journal of Applied Forestry. 12: 166-169.

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.

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

Li, B.; McKeand, S.E.; Hatcher, A.V.; Weir, R.J. 1997. Genetic gains of second generation selections from the North Carolina State University-Industry Cooperative Tree Improvement Program. In: White, T.; Huber, D.; Powell, G., comps. Proceedings of the 24th biennial southern tree improvement conference. Sponsored publication no. 46: 234-238.

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

Li, B.; McKeand, S.; Weir, R. 2000. Tree improvement and sustainable forestry - results from two cycles of loblolly pine breeding in the U.S.A. Journal of Sustainable Forestry. 10: 79-85.

Long, J.N. 1985. A practical approach to density management. Forestry Chronicles. 61: 23-27.

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

McKeand, S.E.; Abt, R.C.; Allen, H.L. [and others]. 2006a. What are the best loblolly pine genotypes worth to landowners? Journal of Forestry. 104: 352-358.

McKeand, S.E.; Jokela, E.J.; Huber, D.A. [and others]. 2006b. Performance of improved genotypes of loblolly pine across different soils, climates, and silvicultural inputs. Forest Ecology and Management. 227: 178-184.

Reineke, L.H. 1933. Perfecting a stand-density index for even-aged forests. Journal of Agricultural Research. 46: 627-638.

COMPARISON OF SECOND GENERATION OPEN-POLLINATED, MASS CONTROL-POLLINATED, AND VARIETAL PINE PLANTING STOCK THROUGH 6 YEARS ON A NORTH MISSISSIPPI SITE

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

Abstract--Landowners face a wide array of loblolly pine genetic material to choose from at the time of regeneration. In general, most opt to plant open-pollinated second-generation stock (second-Gen OP) as previously recommended by either consultants or industry personnel. The goal of this study is to evaluate a selected second-Gen OP family, a selected mass control-pollinated family (MCP), and varietal stock in terms of performance as well as form characteristics. This study was established on an old pasture site near Holly Springs, MS, in a nested randomized complete block design with six blocks, arranged by genetic stock and planted in 100-tree block plots at a spacing of 12 by 9 feet. The study was measured annually through the first 4 years. At age 6, the MCP family was significantly outperforming the second-Gen OP family and the varietal stock for both diameter and volume. The MCP family exhibited overall better performance than the second-Gen OP family and the overall means of the varietal plots. However, the comparison between the MCP family and the top-performing varietals revealed that, while the selected varietals were taller, their diameter and volume were less than the MCP. What was striking was that some of the best-growing varietals also exhibit exceptional stem form and limb characteristics making them highly suitable for higher end products.

INTRODUCTION

Currently, the majority of non-industrial private forest landowners (NIPF) in Mississippi have few options for assistance in deciding upon the best level of genetic quality for their loblolly pine (*Pinus taeda* L.) plantations. In the past, landowners relied primarily on the Mississippi Forestry Commission (MFC) not only for their pine seedlings but also for recommendations of what type of genetic stock they should be planting. Due to budget cuts, the MFC, which was a member of the Western Gulf Tree Improvement Program at Texas A&M University, made the decision to vacate this tree improvement program and to mothball the Commission's seed orchards and nurseries. With the landowners' primary source of information and seedlings gone, there is a void in terms of recommendation for the most current planting stock. Since they had become accustomed to earlier recommendations of second-generation (second-Gen) planting stock, most landowners have continued to follow this advice when ordering new seedlings for regenerating newly harvested stands. The variety of genetically improved loblolly pine seedlings available today to the landowner spans a wide range from open-pollinated 1.5-generation seedlings to clonal selections known as varietals. The high cost of membership likely restricts most small landowners or individual consultants from participating in such programs.

Genetic gains from open-pollinated seedlings can be increased by planting seedlings in single half-sib family blocks, allowing selection of parents that exhibit greater breeding values (Duzan and Williams 1988, McKeand and others 2006). As of 2002, nearly 60 percent of all southern pine plantations and 80 percent of industrial plantations were deploying seedlings in single half-sib family blocks (McKeand and others 2006, McKeand and others 2007). Further genetic gains can be achieved by using full-sib families, produced through mass controlled-pollination (MCP) techniques, also known as supplemental mass-pollinations (SMP) (Bramlett 1997). Jansson and Li (2004) showed potential volume gains from full-sib families of up to 60 percent over unimproved stock, with realized gains dependent on the selection intensity of the specific cross.

Indications are that clonal forestry (i.e. varietals) will provide even greater genetic gains in forestry through mass propagation of highly selected genotypes. The most commonly used technique for most conifers has been rooted cuttings. Mass production of planting stock via tissue culture or somatic embryogenesis (SE) techniques, while common with some hardwood species, has until recently been impractical in southern pines due to lack of an efficient propagation system. Advances in techniques of SE and cryopreservation have increased the potential for clonal, or varietal, southern pine planting stock (Park 2002). An important

[1]Professor, Professor, and Research Associate, respectively, Mississippi State University, College of Forest Resources, Department of Forestry, Mississippi State, MS 39762.

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advantage of clonal propagation via SE is that the embryonic tissue can be cryopreserved while the varietal lines are tested for genetic superiority, thus overcoming the problem of tissue maturation (Park 2002, Sutton 2002). Despite the tremendous promise of this technology, studies have yet to confirm that the enhanced growth and quality produced by these trees is economically justifiable at current varietal costs. Currently we view the end products of varietal material as being those that have the greatest potential of every seedling, combining excellent growth, stem form, and wood characteristics. Current goals are to determine the best use of varietals in relationship to advanced open-pollinated seedlings from third-cycle parents and MCP seedlings.

The objectives of this specific trial are to examine three genetic types of planting stock on a north Mississippi site and to provide on-site demonstration to landowners of the three stock types. The three types included a select open-pollinated second-generation (second-Gen OP) family, a select MCP family, and a number of varietals. In addition to this first objective, the performance of the various genotypes included in the varietal plots are a portion of an overall series of varietal trials by ArborGen examining varietal selection across the southern United States.

MATERIALS AND METHODS

This study was established in the spring of 2007 at a Mississippi State University's North Branch Experiment Station near Holly Springs, MS. Soils on the site are a Loring silt loam. The site had previously been in hay production and pasture, thus the soils were somewhat compacted.

In January 2007, prior to planting, the site was sub-soiled at a 12-foot spacing to a depth of approximately 18 inches, with glyphosate banded at a rate of 2 quarts per acre over the sub-soiled slits in March to eliminate existing herbaceous vegetation. The study was hand-planted on March 23-24, 2007 at a spacing of 12- by 9-feet, with each treatment plot consisting of 100-tree block plots arranged as 10- by 10-tree plots. As each seedling was planted, a single 20 mg. SilvaShield tablet was inserted into each dibble hole as a precautionary method for tip moth control. In May 2008, the site received a broadcast application of Oustar® at a rate of 14 ounces per acre. At the end of both the first and second growing seasons, stem heights were measured on the 64-tree internal measurement plots within each treatment plot.

The study is a nested design consisting of six blocks with three treatments nested in each block. The three treatments were three distinct levels of genetic improvement of the loblolly pine planting stock. These were second-Gen OP, MCP, and varietal (SE) stock produced using SE techniques. The second-Gen OP seedlings were 1-0 bareroot stock while the MCP and SE material were both produced in containers. Each of the six blocks contained a single 100-tree plot of each genetic type.

The second-Gen OP and MCP material was selected from MeadWestvaco based on their performance in tests located in southwest Tennessee and provided by ArborGen. The SE material was not a single clone but rather a composite of 56 SE varieties, with one ramet of each variety included in each varietal treatment plot. The remaining trees in the 64-tree varietal plots included checks and filler trees. All of the SE material was provided by ArborGen and was included into the ArborGen Testing Service. Only varieties with at least four of the original six ramets surviving after the second growing season were included in this analysis.

Standard GLM techniques in SAS 9.3 were used to compare the mean height of the second-Gen OP and MCP material against that of the SE genetic type through age 6. In addition, the SE varieties were ranked by age-6 height, diameter, and volume, with the average height and volume of the three tallest varieties examined relative to the MCP stock types.

RESULTS

Overall test survival at age 6 continued to remain high, 94 percent, with a drop of only 1 percent between ages 4 and 6. Survival of the three genetic types showed the second-Gen OP was the highest at 97 percent followed by the MCP at 95 percent, and then the SE type at 90 percent. There was no significant difference between the second-Gen OP and the MCP types through age 6, but both were significantly different from the SE type.

Average height among the three genetic types showed that the MCP type was taller than the second-Gen OP and SE types from age 2 through 6. The difference in total height between the MCP and the second-Gen OP types increased to approximately 1.2 feet at age 6, whereas the difference between the MCP and SE types remained approximately the same at 1.5 feet (table 1).

Examination of height growth between age 2 and 6 showed that growth rates increased for all three genetic types between ages 2 and 4 but decreased between ages 4 and 6 (table 2). Volume growth between ages 3 and 4 and ages 4 and 6 showed that second-Gen OP and MCP were similar while the SE type was slightly lower. However, while there was no significant difference between the second-Gen OP and MCP types across either age grouping there is a trend for the MCP to be increasing its numerical difference over the second-Gen OP type (table 2). Because of the great diversity of genotypes that make up the SE type, it is difficult to compare this type to either the second-Gen OP or MCP types. While the SE type was statistically smaller in volume growth than either of the other two genetic types, the difference reflects that the vast majority of the genotypes were poorly adapted to the site.

The majority of the MCP trees at age 6 were between 22- and 29-feet tall whereas majority of the second-Gen OP trees were between only 15 and 22 feet. The same trend is also true for age-6 d.b.h., with the MCP type having a greater number of trees in the larger diameter classes compared to the second-Gen OP type

As previously mentioned, one of the objectives of this particular study was to identify the best performing planting stock at this specific site. Significant varietal (SE) differences were shown for all traits from age-1 height to age-6 height, d.b.h., and volume. Table 3 provides a comparison between the MCP type means for height at ages 3, 4, and 5 years for the top three performing varietal means. Mean height of the top three genotypes were 11.2, 16.4, and 24.0 feet at ages 3, 4, and 6 years, respectively as compared to 10.4, 15.8, and 23.2 feet for the MCP type. The mean height of the best genotype was 11.6, 17.0, and 24.1 feet at ages 3, 4, and 6 years, respectively (table 3). The top three volume-producing genotypes showed mean values of 0.45, 0.77, and 2.13 cubic feet at ages 3, 4, and 6 years, respectively. The best volume producing genotype showed a mean value of 0.45, 0.78, and 2.14 cubic feet at ages 3, 4, and 6 years, respectively (table 3).

DISCUSSION

The overall test survival has been quite good through age 6, with the majority of the mortality concentrated in one block due to intense Bermuda grass (*Cynodon dactylon* L.) competition. The SE seedlings were the only actively growing stock when the test was established; these seedlings were shipped to us just prior to planting. The other two genetic types were either bareroot, as the case of the second-Gen OP seedlings, or dormant container grown seedlings as were the MCP seedlings. In addition, the SE seedlings were less developed than the other two seedling types (in terms of height and root systems), likely accounting for the higher mortality rates and quite possibly the slower growth rates (Rousseau and others 2012). Taking into account the problem incurred with the SE seedlings and the intense grass competition, the lower survival rates of this genetic type should not be considered a genetic problem but rather one associated with planting stock quality.

Although the MCP stock showed better height, diameter, and volume performance compared to the second-Gen OP stock at both ages 4 and 6, the difference between these three traits either remained the same or increased slightly with increasing age. While it was expected that the MCP type would exhibit better performance than the second-Gen OP type, the small difference between the two types at age 6 was unexpected. Because the genetic make-up of a MCP family is based on known selected parents, the resulting progeny of a cross between two high performing parents will hopefully produce

Table 1--Least square means for d.b.h., total height, and volume for the three genetic types at ages 4 and 6 of the 2007 Loblolly Pine Genetic Comparison and Varietal Study located near Holly Springs, MS

	---------------Age 4[a]---------------			---------------Age 6[a]---------------		
Genetic type	D.b.h.	Height	Volume	D.b.h.	Height	Volume
	inches	*feet*	*feet*^{3}	*inches*	*feet*	*feet*^{3}
2nd-Gen OP	3.1a	14.7b	0.70b	5.5b	21.9a	1.95ab
MCP	3.5a	15.8a	0.81a	5.9a	23.2a	2.24a
Varietal (SE)	2.7b	14.2b	0.61b	5.0c	21.6a	1.69b

[a]Mean values for the same column are significantly different at the 0.05 level of the Duncan's Test if the letters are different.

Table 2--Early-age height and volume growth of the three genetic types tested in the 2007 Loblolly Pine Genetic Comparison and Varietal Study located near Holly Springs, MS

	-----------------Height (*feet*)[a]---------------			------Volume (*feet*^{3})[a]------	
Genetic type	2-3 years	3-4 years	4-6 years	3-4 years	4-6 years
2nd-Gen OP	4.5a	5.1a	3.6a	0.29a	0.62ab
MCP	4.8a	5.4a	3.7a	0.36a	0.71a
Varietal (SE)	4.6b	4.7b	0.6b	0.21b	0.53b

[a]Mean values for the same column are significantly different at the 0.05 level of the Duncan's Test if the letters are different.

Table 3--Least square means for height and volume at ages 3, 4, and 6 years for MCP, the top three varietals, and the best performing varietal for that specific trait in the 2007 Loblolly Pine Genetic Comparison and Varietal Study located near Holly Springs, MS

	----------Height (*feet*)[a]----------			---------Volume (*feet*^{3})[a]--------		
	Age 3	Age 4	Age 6	Age 3	Age 4	Age 6
MCP	10.4	15.8	23.2	0.45	0.81	2.24
Top 3 varietals	11.2	16.4	24.0	0.45	0.77	2.13
Best varietal	11.6	17.0	24.1	0.45	0.78	2.14
Field ID	(228)	(228)	(14)	(329)	(228)	(228)

[a]Mean values for the same column are significantly different at the 0.05 level of the Duncan's Test if the letters are different.

progeny that are less variable as well as exhibiting greater gains in growth, as long as the resulting progeny are adapted to the site conditions (Rousseau 2010). In comparison, in the second-Gen OP family, there is no control of the pollen parents. Thus we expect greater variability among the progeny and the resulting growth to be less than the designed cross of an MCP family. The variability of an open-pollinated family is not only due to the variation of the genotypes within the seed orchard but outside pollen contamination as well. Adams and Birkes (1989) showed that for an open-pollinated seed orchard, contamination from outside pollen could be as high as 50 percent in any year, thus adding to the variability among progeny and the reduction in expected gains. Further examination of the MCP and second-Gen OP types showed approximately 63 percent of the MCP test population and 41 percent of the second-Gen OP test population fell into the 22- to 24-height category at age 6. The greater age-6 height variability among the second-Gen OP test population was typical of an open-pollinated family whereas the MCP type showed much less variability among height with the preponderance of the progeny falling into a single category, thus exhibiting less variability as expected of progeny from a full-sib family. Examination of the diameter distributions between the MCP and second-Gen OP progeny showed a higher percentage of second-Gen OP progeny falling into the lower diameter classes (i.e. 3-, 4-, and 5-inch classes). This trend was reversed for the higher diameter classes (i.e. 6-, 7-, and 8-inch classes) where the MCP progeny had the highest percentage. In fact, 63 percent of the MCP progeny fell into the 6-inch diameter class, again showing less variability than the second-Gen OP progeny. Based on the better height and diameter of the MCP progeny, it was evident that the total volume (cubic feet) would be greater than the second-Gen OP type. This was shown to be true since the mean of the MCP type was 900 cubic feet compared to 790 cubic feet for the second-Gen OP type.

Unfortunately it was not possible to directly compare the performance SE material to either the MCP or the second-Gen OP types due to the fact that the SE plots consist of 54 different genotypes as well as some check lots. Because of this, only a maximum of six observations per genotype could be made, which only provides a glimpse of the type of performance that might be expected. With these limitations, the performance of the SE type was compared to the MCP type as a means of gauging the performance of selected SE individuals. As shown in table 3, the top three varietals and the best varietal were taller than the MCP mean, but there was little difference among the better performing SE genotypes. While Frampton and Huber (1995), working with loblolly clonal stock, showed that clones could yield considerably higher gains than full-sib families, this does not seem to be the case in this comparison of a limited numbers of SE genotypes. In this case, a mixture of the best three genotypes was as good as a single genotype, and this mixture would add genetic diversity to a planting. Volume production between the MCP and SE types differed very little, but the trend for higher volume production of the MCP type was due to the larger diameters exhibited by age 6. While yield in terms of volume production certainly provides a greater tonnage production, this may or may not translate into higher quality products that landowners are seeking (McKeand and others 2006, McKeand and others 2007, Sherrill and others 2008). Thus, a more appropriate factor would be to consider the variability of the SE individuals in terms of stem quality and crown characteristics that are exhibited by specific genotypes (Cumbie and others 2012, Dougherty and others 2010). Varietal 567 exhibited exceptional form as well as above average height and diameter. This is likely the type of individual that landowners would favor. However, at the current cost of approximately $400 per thousand seedlings, this material is too expensive for most small landowners (Rousseau 2010).

CONCLUSION

Age-6 results indicated that while the MCP type outperformed the second-Gen OP type, the differences in height, diameter, and volume were not as large as expected. The decreased variability among the progeny of the MCP type in comparison to the open-pollinated second-Gen OP type was seen for both diameter and height at age 6. This decrease in variability may become more important as the stand ages and moves into the various products through the rotation. The result could be higher returns per acre as well as reduced rotation lengths. In addition, the greater cost of MCP seedlings is not significantly different to the point where it would limit the use of this type of genetic material. However, landowners would benefit if there were a larger number of trials that examined a wider variety of MCP families to determine growth and quality on a diversity of Mississippi sites.

The performance of the SE material was quite variable but expected as there were 56 different genotypes. However, there were a limited number of excellent genotypes that showed not only good growth but excellent stem form and branch characteristics. Like the MCP, additional testing of a large number of SE genotypes would provide a greater base of material from which to choose. If suitable, including a number complimentary growth genotypes would add genetic diversity to a plantation establishment.

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LITERATURE CITED

Adams, W.T.; Birkes, D.S. 1989. Mating patterns in seed orchards. In: Proceedings of the 20th southern forest tree improvement conference. Sponsored Publ. No. 42: 75-86.

Bramlett, D.L. 1997. Southeastern conifers: genetic gains from mass control pollination and topworking. Journal of Forestry. 95(3): 15-19.

Cumbie, W.P.; Isik,F.; McKeand, S.E. 2012. Genetic improvement of sawtimber potential in loblolly pine. Forest Science. 58(2): 168-177.

Dougherty, D.; Bryant, R.; Burkhart, H. [and others]. 2010. Valuing tomorrow's loblolly pine plantations today. Forest Landowner. Jan./Feb.: 19-21.

Duzan, H.W., Jr.; Williams, C.G. 1988. Matching loblolly pine families to regeneration sites. Southern Journal of Applied Forestry. 12: 166-169.

Frampton, L.J., Jr.; Huber, D.A. 1995. Clonal variation in four-year-old loblolly pine in coastal North Carolina. In: Weir, R.J.; Hatcher, A.V., comps. Proceedings of the 23rd southern forest tree improvement conference. Sponsored Publ. 45: 254-284.

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

McKeand, S.E.; Abt, R.C.; Allen, H.L. [and others]. 2006. What are the best loblolly pine genotypes worth to landowners? Journal of Forestry. 104 (7): 352-358.

McKeand, S.E.; Zobel, B.J.; Byram, T.D.; Huber, D.A. 2007. Southern pine tree improvement – a living success story. In: McKeand, S.E.; Li, B., eds. Proceedings of the 28th southern forest tree improvement conference. Sponsored Publ. No. 50: 3-6.

Park, Y.S. 2002. Implementation of conifer somatic embryogenesis in clonal forestry: technical requirements and deployment considerations. Annals of Forest Science. 59: 651-656.

Rousseau, R.J. 2010. What are genetically improved seedlings? Mississippi State University Extension Service Publ. No. 2617. Starkville, MS: Mississippi State University. 5 p.

Rousseau, R.J.; Roberts, S.D.; Herrin, B.L. 2012. Comparison of advanced genetic loblolly pine planting stock. 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 Agriculture Forest Service, Southern Research Station: 269-272.

Sherrill, J.R.; Mullin, T.J.; Bullock, B.P. [and others]. 2008. An evaluation of selection for volume growth in loblolly pine. Silvae Genetica. 57(1): 22-28.

Sutton, B.C. 2002. Commercial delivery of genetic improvement to conifer plantations using somatic embryogenesis. Annals of Forest Science. 59: 657–661.

SOIL PHYSICAL EFFECTS ON LONGLEAF PINE PERFORMANCE IN THE WEST GULF COASTAL PLAIN

Mary Anne S. Sayer, James D. Haywood, and Shi-Jean Susana Sung[1]

Abstract--We summarize 8 years of soil physical property responses to herbicide manipulation of the understory in two young longleaf pine stands growing on either Ruston fine sandy loam or Beauregard silt loam soils. We also describe relationships between pine sapling vigor and the soil physical environment across a 3-year period on the Ruston soil and a 2-year period on the Beauregard soil. It is hypothesized that understory control affects soil porosity, bulk density, and the ability to store plant-available water by a change in the amount and distribution of non-pine roots. Furthermore, *Pinus* vigor may be reduced when the inherent physical nature of a soil limits pine root elongation. We observed temporal changes in soil porosity fractions and bulk densities, possibly representing natural soil recovery after disturbance. Near the surface of the soil, soil perturbation by grass roots may have aided pine vigor by increasing the water-holding capacity of soil micropores. In the subsoil, pine vigor was correlated with bulk density and microporosity. Relationships between pine vigor and subsoil physical properties were different between the two soil types. Clay illuviation and sand content in the two soil types may have played a role in these relationships. Our results provide insight regarding soil variables that impart some degree of control on pine root system expansion and tree vigor on the West Gulf Coastal Plain.

INTRODUCTION

Southern pine root systems normally supply adequate water to sustain pine vigor across the southeastern United States. Subsoil water may be accessed by roots growing along interped spaces and in pores created by roots and soil fauna (Van Lear and others 2000). This means of water acquisition is vital in the West Gulf Coastal Plain, where many forest soils are characterized by root-growth-limiting subsoil bulk densities (Patterson and others 2004, Scott and others 2007). The proliferation of pine ectomycorrhizae and rhizomorphs near the surface of the soil and the elongation of deep pine roots also enable water to be hydraulically redistributed near the soil surface by its nocturnal, deep-root absorption, ascension, and shallow-root release (Dawson 1993; Warren and others 2006, 2008).

In addition to root system expansion, soil porosity controls water uptake by roots. Soil micropores with surface tensions > 1.5 MPa reduce the volume of water available for plant uptake, while those with surface tensions > 0.3 MPa and < 1.5 MPa favor water accessibility to plant roots (Kramer and Boyer 1995). Furthermore, the movement and decomposition of plant roots and soil fauna create macropores that serve as conduits for root elongation, and over time these macropores accelerate the development of large micropores capable of supplying plant-available water.

Herbicide application and prescribed fire change the amount and composition of competing vegetation above the soil surface (Haywood 2009, 2011). It is likely that parallel changes in rooting occur belowground. We found, for example, that repeated prescribed fire in March and July increased non-pine rooting in the upper 5 cm of the A horizon compared to no prescribed burning or prescribed fire in May (Sword Sayer and Haywood 2012). This effect was attributed to greater grass and forb competition when prescribed fire was applied in March or July compared to no prescribed fire or its application in May.

We hypothesize that vegetation management treatments indirectly affect soil porosity and bulk density by changing the amount and distribution of non-pine rooting. Also, where the inherent physical nature of a soil type has the potential to limit pine root elongation, negative effects on soil porosity and bulk density caused by vegetation management treatment may be seen as a reduction in *Pinus* vigor. The first objective of this analysis was to summarize long-term soil physical property responses to herbicide application during establishment of young longleaf pine (*Pinus palustris* Mill.) plantations on two West Gulf Coastal Plain forest loam soils. Our second objective at these two sites was to assess relationships between two physiological variables representative of *Pinus* vigor and soil physical properties.

MATERIALS AND METHODS

Study Site

The study is located at two sites in the Kisatchie National Forest in central Louisiana. Site 1 supports two replications on a Ruston fine sandy loam (fine-loamy, siliceous, semiactive, thermic Typic Paleudults) containing some Malbis fine sandy loam (fine-loamy, siliceous, subactive, thermic Plinthic

[1]Research Plant Physiologist, Supervisory Research Forester, and Research Plant Physiologist, respectively, USDA Forest Service, Southern Research Station, Pineville, LA 71360.

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Paleudults) and Gore very fine sandy loam (fine, mixed, active, thermic Vertic Paleudualfs) (31^{o} 6' N, 92^{o} 36' W). Site 2 supports three replications on a Beauregard silt loam (fine-silty, siliceous, superactive, thermic Plinthaquic Plaeudults) and Malbis fine sandy loam complex (31° 1' N, 92^{o} 37' W). A mixed pine-hardwood forest originally occupied both sites. In 1996, site 1 was clearcut harvested and roller-drum chopped, followed by burning in 1997. In 1991, site 2 was clearcut harvested, sheared, and windrowed, and burned in 1993 and 1996. Container-grown longleaf pine seedlings from genetically improved Louisiana (site 1) and Mississippi (site 2) sources were planted (1.8 by 1.8 m) in November 1997 and March 1997, respectively. Treatment plots, 22- by 22-m (0.048 ha), were established at each location, and blocks were delineated by soil drainage and topography. Treatment plots contained 12 rows of 12 seedlings, and measurement plots were the internal 8 rows of 8 seedlings in each treatment plot.

In each block, the competing vegetation of one plot was not treated (C, control), while that of a second plot was chemically treated (H, herbicide). Herbicides were applied after planting longleaf pine seedlings to control herbaceous and arborescent plants. On site 2, sethoxydim and hexazinone in aqueous solution were applied in bands centered over the rows of unshielded seedlings in May 1997 and April 1998. The rate of sethoxydim application was 0.37 kg active ingredient (ai)/ha, and the rate of hexazinone application was 1.12 kg ai/ha. At site 1, hexazinone was similarly banded in April 1998 and 1999. At both sites in April 1998 and May 1999, triclopyr at 0.0048 kg acid equivalent/liter was tank-mixed with surfactant and water and applied as a directed foliar spray to competing arborescent vegetation. Brush that recovered by February 2001 was cut by hand.

Diameters at breast height (d.b.h., cm) of the measurement trees were measured annually during the dormant season (December through February). Basal areas (BA_{tree}) of measurement trees were calculated. In early 2002, 2004, 2006, and 2009, three sample trees from among those of average height per measurement plot were randomly identified on site 1 for additional physiology and soil measurements. Similarly at site 2, three sample trees per plot were identified in early 2004, 2006, and 2009.

Soil Physical Property Measurements
A tractor-mounted hydraulic probe was used to extract one long (5.1 cm diameter by 61 cm long) and one short (5.1 cm diameter by 30.5 cm long) soil core about 1 m from the base of each sample tree in 2002, 2004, 2006, and 2009. Cores were placed in capped plastic liners and refrigerated until processing. Intact core increments 10 cm in length were excised from the A and upper and lower Bt1 horizons of long soil cores and the A horizon of short soil cores. First, the depth to the top of the argillic (i.e., Bt1) horizon (DAH) of the long cores was estimated by soil color and texture. From the long cores, two 1-cm sections were excised from the A horizon core increment (2 to 12 cm), the upper Bt1 core increment, and the lower Bt1 core increment (50 to 60 cm) using a band saw. The upper Bt1 horizon core increment was defined as the 2- to 12-cm soil core section below the top of the Bt1 horizon. From short soil cores, two 1-cm sections were excised from the 2- to 12-cm depth using a band saw. In each year, 60 soil cores were processed for A horizon information, and 30 soil cores were processed each for upper and lower Bt1 horizon information.

One of each pair of 1-cm sections was positioned on an equilibrated -0.1 MPa or -1.5 MPa ceramic pressure plate. The water retention method was used to estimate total porosity fraction (TOP), microporosity fraction (MIP), macroporosity fraction (MAP), and plant-available water holding capacity (PAWHC) (Klute 1986). The water content of 1-cm core sections at -0.03 MPa was defined as soil water content at field capacity (WATFC), and that at -1.5 MPa was defined as soil water content at permanent wilting point (WATWP). The core bulk density method was used to measure bulk density (BD) (Blake and Hartge 1986).

Predawn Needle Water Potential and Net Photosynthesis Measurements
The fascicle gas exchange of each sample tree was measured in 2003, 2004, and 2005. Measurements were conducted in May, July, and September 2003, April and July 2004, and May and October 2005. Within a consecutive 3-day period, two blocks were measured on each of the first two days and the fifth block was measured on the third day.

Between 0500 and 0600 hours on the day of gas exchange measurements, one current-year, mature, mid-crown fascicle was detached, placed in a plastic bag, and stored in darkness on ice. Predawn needle water potentials (PWP) were measured by a pressure chamber (PMS Instrument Co., Corvallis, OR) within 1 hour of detaching.

On each gas exchange day, the light saturated net photosynthesis rate (A_{sat}) in the upper crown of each sample tree was measured in the afternoon (1300-1530 hours) with a portable photosynthesis system (Model 6400, Li-Cor, Inc. Lincoln, NE) and a standard leaf chamber equipped with a light emitting diode light source (Model 6400-02B, Li-Cor, Inc. Lincoln, NE). For each measurement, two fascicles with three needles each from the south side of a sample tree were detached and placed in the leaf chamber. Measurements were an average of 20 one-second readings, taken after the chamber environment had stabilized. Time between fascicle detachment and measurement was approximately 2 minutes. All measurements were conducted at a photosynthetically active radiation value of 1400 umol/m^2/s. After each measurement, fascicles were placed in plastic bags on ice, and needle surface areas in the leaf chamber were determined by the displaced needle volume method (Johnson 1984). Values of A_{sat} were expressed on a total leaf surface area basis as μmol CO_2/m^2/s.

Statistical Analysis

For each site, mean values of BD, MIP, MAP, and PAWHC for each horizon were transformed, as needed, to natural logarithm or square root values to establish normally distributed experimental errors and evaluated by analysis of variance using a split-plot in time, randomized complete block design. Sites 1 and 2 had two and three blocks, respectively. Year was the whole plot effect, and vegetation management treatment was the subplot effect. Means were compared by the Tukey test and considered significantly different at $\alpha = 0.05$.

For each site, the mean depth to DAH was calculated, and trees with DAH within one standard deviation of mean DAH were partitioned into two subsets by PWP. The first subset contained trees with PWP ≥ -0.6 MPa (moist), and the second subset contained trees with PWP < -0.6 MPa (dry). A PWP value of -0.6 MPa was chosen to distinguish two levels of water status based on the results of Sword Sayer and others (2005) who found that the new root growth of longleaf pine seedlings was significantly reduced when PWP was < -0.6 MPa.

Ordinary least squares regressions between 2 dependent variables (PWP, A_{sat}) and 14 independent variables (year, DAH, and BD, MIP, MAP, and PAWHC of the A, and upper and lower Bt1 horizons) were conducted by site, vegetation management treatment (control and herbicide), and water status (moist and dry) with SAS statistical software (SAS Institute, 9.2 ed., Cary, NC) using the generalized linear model procedure. Original regressions included tree basal area, but this independent variable was excluded from the final analysis because it was not significant. Correlation coefficients (r) were determined for significant regression relationships. The F statistics and coefficients of determination (R^2) were considered significant at $P \leq 0.05$.

RESULTS AND DISCUSSION

Soil Physical Properties

The physical properties of the two soil types changed over time and in response to vegetation management treatment. For example, all measured physical properties (BD, MIP MAP, PAWHC) of the Beauregard A horizon changed significantly by year (table 1). No significant effect of year, however, was observed in the A horizon of the Ruston soil (table 2). Subsoil bulk densities and porosities of both soil types were significantly affected by year in most comparisons, with a tendency for BD to decrease with time after planting (fig. 1). Specifically, the BD of the A and lower Bt1 horizons of the Beauregard soil decreased 7 and 6 percent respectively, and that of the upper and lower Bt1 horizon of the Ruston soil decreased 5 and 6 percent, respectively, between 2004 and 2009. A similar effect was observed in the Beauregard soil with 20, 16, and 9 percent lower MIP in the A, and upper and lower Bt1 horizons, respectively, between 2004 and 2009 (fig. 2). The only significant effect of year on Ruston MIP was observed in the lower Bt1 horizon with an 11 percent decrease between 2004 and 2009. As MIP decreased over time, significant increases in MAP were observed in the A and lower Bt1 horizons of the Beauregard soil and the upper and lower Bt1 horizons of the Ruston soil.

Table 1--Analyses of variance of mean soil physical properties at three depths of the Beauregard silt loam soil[a] in response to no vegetation management or vegetation management by herbicide application in central Louisiana

	----------A horizon----------			-----Upper Bt1 horizon-----			------Lower Bt1 horizon------		
Source of variation	df[b]	MS	Pr > *F*	df	MS	Pr > *F*	df	MS	Pr > *F*
	-----------------------------------Beauregard[a] bulk density (BD)-----------------------------------								
Block (B)	2	0.00370	0.5277	2	0.01720	0.1687	2	0.00785	0.3727
Vegetation management (T)	1	0.00054	0.7518	1	0.00005	0.9195	1	0.00001	0.9618
error a[c]	2	0.00414		2	0.00349		2	0.00466	
Year (Y)	2	0.01682	0.0019	2	0.01529	0.1413	2	0.01682	0.0194
T x Y	2	0.01398	0.0040	2	0.00032	0.9484	2	0.00202	0.5137
error b[c]	8	0.00110		8	0.00606		8	0.00278	
	---------------------------------Beauregard macroporosity (MAP)----------------------------------								
Block (B)	2	0.00247	0.5484	2	0.00388	0.1548	2	0.00240	0.2483
Vegetation Management	1	0.02200	0.1135	1	0.00012	0.7248	1	0.00023	0.6449
error a	2	0.00300		2	0.00071		2	0.00079	
Year	2	0.01485	<0.0001	2	0.00714	0.0609	2	0.00636	0.0107
T x Y	2	0.00014	0.6340	2	0.00013	0.9311	2	0.00023	0.7473
error b	8	0.00029		8	0.00176		8	0.00076	
	----------------------------------Beauregard microporosity (MIP)-----------------------------------								
Block	2	0.00446	0.1779	2	0.00050	0.4625	2	0.00042	0.0977
Vegetation Management	1	0.01962	0.0459	1	0.00005	0.7665	1	0.00034	0.1122
error a	2	0.00097		2	0.00043		2	0.00005	
Year (Y)	2	0.00563	0.0011	2	0.00360	0.0159	2	0.00118	0.0113
T x Y	2	0.00116	0.0716	2	0.000003	0.9946	2	0.00003	0.8139
error b	8	0.00031		8	0.00050		8	0.00014	
	----------------Beauregard plant-available water holding capacity (PAWHC)-----------------								
Block	2	29.5509	0.1453	2	37.6864	0.1957	2	6.60352	0.0297
Vegetation Management	1	135.772	0.0351	1	26.3824	0.2320	1	17.2386	0.0115
error a	2	5.02395		2	9.1713		2	0.2022	
Year	2	35.5328	0.0105	2	29.2363	0.0720	2	6.75891	0.5371
T x Y	2	8.58627	0.1907	2	1.85665	0.7948	2	0.69449	0.9338
error b	8	4.18200		8	7.85483		8	10.05080	

[a]Site 2 is mostly Beauregard silt with some Malbis fine sandy loam.
[b]df = degrees of freedom; MS = mean square; Pr > *F* = probability greater than *F*-value.
[c]error a df = (T-1) x (B-1); error b df = T x (B-1) x (Y-1).

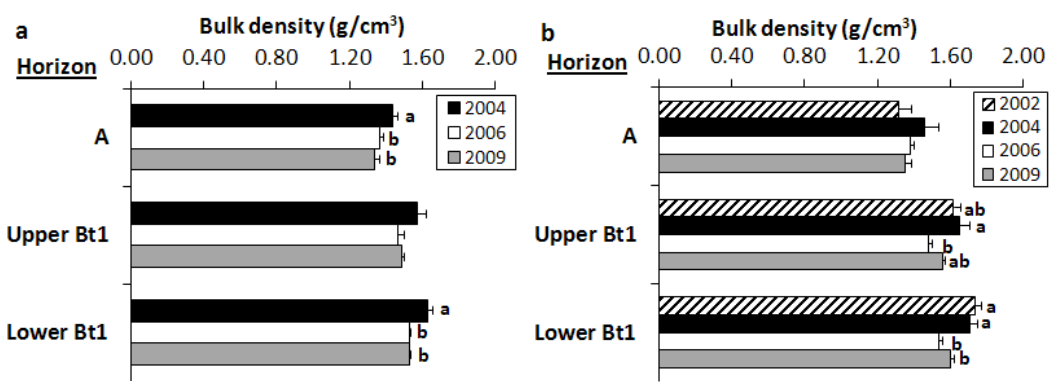

Figure 1--Mean bulk density of the A horizon, the upper Bt1 horizon, and the lower Bt1 horizon at (a) site 2 (Beauregard silt loam) and (b) site 1 (Ruston fine sandy loam) between 2002 and 2009. Bars represent one standard error of the mean. Means within a site and horizon associated with a different lower case letter are significantly different by the Tukey test at α = 0.05.

Table 2--Analyses of variance of mean soil physical properties at three depths of the Ruston fine sandy loam soil[a] in response to no vegetation management or vegetation management by herbicide application in central Louisiana

	----------A horizon----------			-----Upper Bt1 horizon-----			------Lower Bt1 horizon------		
Source of variation	df[b]	MS	Pr > *F*	df	MS	Pr > *F*	df	MS	Pr > *F*
	----------Ruston[a] bulk density (BD)----------								
Block (B)	1	0.04324	0.0491	1	0.00820	0.2690	1	0.01103	0.5783
Vegetation management (T)	1	0.03463	0.0548	1	0.00005	0.8897	1	0.00028	0.9209
error a[c]	1	0.00026		1	0.00166		1	0.01813	
Year (Y)	3	0.01351	0.3693	3	0.02364	0.0350	3	0.03492	0.0040
T x Y	3	0.01044	0.4611	3	0.02273	0.1047	3	0.00115	0.6442
error b[c]	5	0.01033		5	0.00362		5	0.00192	
	----------Ruston macroporosity (MAP)----------								
Block (B)	1	0.01152	0.3756	1	0.00009	0.5699	1	0.00004	0.8240
Vegetation Management	1	0.00249	0.6137	1	0.00039	0.3422	1	0.00191	0.3488
error a	1	0.00517		1	0.00014		1	0.00051	
Year	3	0.00279	0.5285	3	0.00330	0.0113	3	0.00546	0.0059
T x Y	3	0.00166	0.6993	3	0.00191	0.0344	3	0.00050	0.2521
error b	5	0.00333		5	0.00029		5	0.00025	
	----------Ruston microporosity (MIP)----------								
Block	1	0.00080	0.7691	1	0.00129	0.2616	1	0.00006	0.6419
Vegetation Management	1	0.00059	0.8001	1	0.000002	0.9400	1	0.00171	0.1826
error a	1	0.00557		1	0.00025		1	0.00015	
Year (Y)	3	0.00087	0.2298	3	0.00194	0.0664	3	0.00185	0.0419
T x Y	3	0.00012	0.8370	3	0.00089	0.2158	3	0.00019	0.6346
error b	5	0.00043		5	0.00042		5	0.00031	
	----------Ruston plant-available water holding capacity (PAWHC)----------								
Block	1	7.67860	0.7754	1	3.83116	0.6436	1	31.26675	0.5990
Vegetation Management	1	1.22113	0.9072	1	2.03810	0.6759	1	1.36180	0.9039
error a	1	56.6397		1	9.75411		1	58.83439	
Year	3	7.55600	0.1444	3	4.56002	0.6785	3	3.54153	0.0772
T x Y	3	1.24326	07169	3	4.44498	0.6862	3	0.86217	0.4541
error b	5	2.65142		5	8.53021		5	0.83685	

[a]Site 1 is mostly Ruston fine sandy loam with some Malbis fine sandy loam and Gore very fine sandy loam.
[b]df = degrees of freedom; MS = mean square; Pr > *F* = probability greater than *F*-value.
[c]error a df = (T-1) x (B-1); error b df = T x (B-1) x (Y-1).

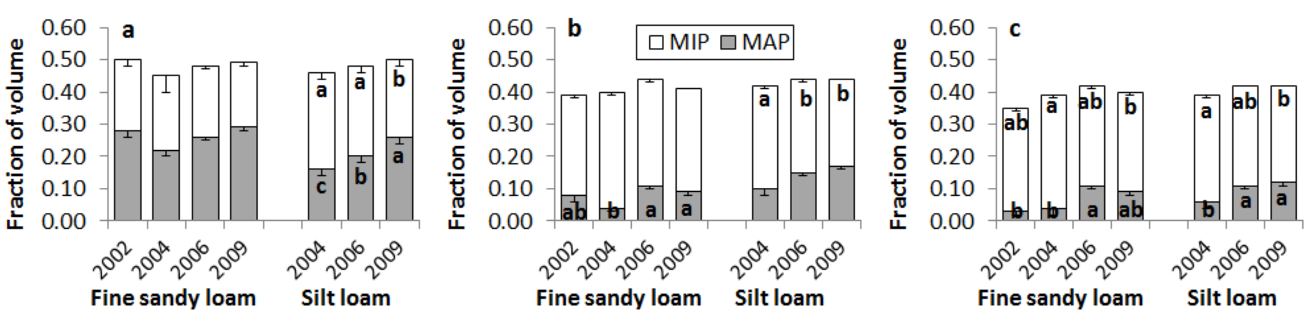

Figure 2--Mean macroporosity (MAP) and microporosity (MIP) of the A (a), upper Bt1 (b), and lower Bt1 (c) horizons at site 1 (Ruston fine sandy loam) and site 2 (Beauregard silt loam) between 2002 and 2009. Bars represent one standard error of the mean. Means within a site and horizon associated with a different lower case letter are significantly different by the Tukey test at α = 0.05.

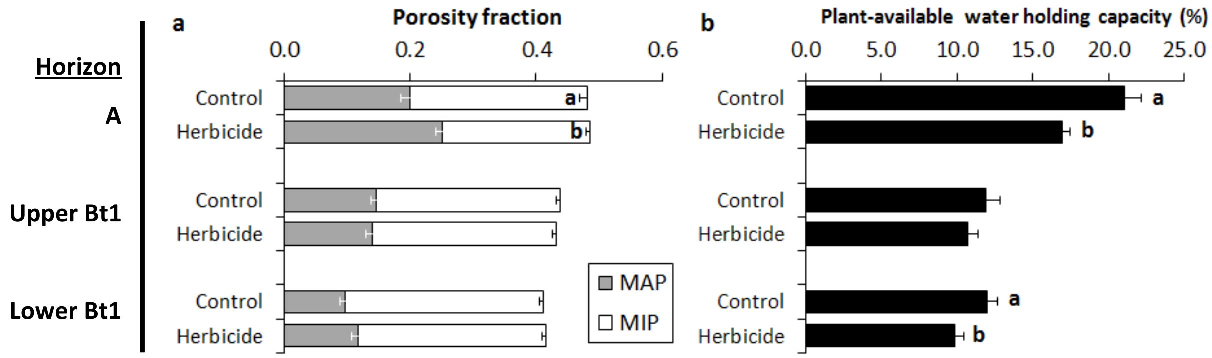

Figure 3--(a) Mean macroporosity (MAP) and microporosity (MIP) and (b) plant-available water holding capacity (PAWHC) at site 2 (Beauregard silt loam) in response to no vegetation management or vegetation management by herbicide application at the time of longleaf pine establishment. Bars represent one standard error of the mean. Means within a variable and horizon associated with a different lower case letter are significantly different by the Tukey test at α = 0.05.

Vegetation management treatment did not impact physical properties of the Ruston soil (table 2) but had a significant effect on the surface soil MIP and PAWHC of the Beauregard soil (table 1). Specifically, herbicide application resulted in 20 and 24 percent reductions in A horizon MIP and PAWHC, respectively, and a 16 percent drop in PAWHC in the lower Bt1 horizon (fig. 3). Two significant interactions between vegetation management treatment and year were also observed, with one indicating that the significant effect of year on the A horizon BD of the Beauregard soil was driven by a decrease in BD between 2004 and 2009 from 1.5 to 1.3 g/cm^3 for the H plots but no change in BD between 2004 (1.4 g/cm^3) and 2009 (1.4 g/cm^3) for the C plots. The second significant interaction was observed for upper Bt1 MAP in the Ruston soil with dramatically lower MAP on the C plots in 2004 compared to all other years regardless of vegetation management treatment. This effect is due to one measurement that was 83 percent lower than the average of all other comparable measurements. We attribute this unusual measurement and significant interaction to either an artifact of the soil sample or a measurement error.

Typical Beauregard and Ruston soils have average Bt1 bulk densities of 1.6 and 1.7 g/cm^3, respectively (National Cooperative Soil Survey 2013). The subsoils of these two soil types have the potential to restrict pine root system expansion because root growth limitations can be expected as BD exceeds 1.6 g/cm^3 (Kelting and others 1999, Pritchett 1979). In our study, average Bt1 bulk densities on the Beauregard and Ruston soils were 1.5 and 1.6, respectively, by 2009. Comparison between typical and actual BD values, and the temporal decreases in BD that we observed, suggest that the establishment of young longleaf pine plantations moderated the root-growth-limiting character of the subsoil at each site. A similar subsoil response was found 10 years after pines were harvested from six coastal plain soils across Texas, Louisiana, and Mississippi (Scott and others 2007, Sword and Tiarks 2002).

Soil porosities also suggest that the soil became more favorable for root growth between 2004 and 2009. On the Beauregard soil, losses in A and Bt1 horizon MIP that occurred between 2004 and 2009 were correlated with gains in MAP. Although not always significant, a similar trend was observed in the Bt1 horizon of the Ruston soil. It is conceivable that as non-pine vegetation recovered and planted pines grew, roots and their ectomycorrhizal networks were the primary source of these soil porosity changes. Shifts in MIP, therefore, were secondary to these initial changes in the fraction of soil porosity attributed to macropores as reported by Kramer and Boyer (1995).

The understory vegetation associated with the Beauregard soil at site 2 is dominated by grasses while that associated with the Ruston soil at site 1 is primarily composed of forbs and woody shrubs (Haywood 2007). Probable differences in understory vegetation rooting between the two sites in this study provide insight regarding why the two soil types had dissimilar responses to vegetation management treatment. It is likely that soil perturbation in the A horizon by grass roots was greater on the Beauregard soil compared to the Ruston soil because the root systems of grass are generally shallower and more fibrous than those of forbs and woody shrubs (Jackson and others 1996). On the C plots of the Beauregard soil where grasses were not chemically controlled by herbicides during establishment, fibrous grass roots may have maintained relatively higher levels of A horizon perturbation and organic matter enrichment. In this situation, an increase in MAP and a decrease in MIP would be expected as the grass fibrous root network expanded. Curiously, the opposite was observed on the Beauregard C plots with greater MIP and no difference in MAP attributable to grass root system expansion.

An evaluation of A horizon PAWHC provides an explanation for this observation. On the Beauregard soil, greater PAWHC on the C plots compared to the H plots indicates that the fraction of MIP capable of storing plant-available water was greater on the C plots compared to the H plots. This result implies that a larger fraction of large micropores existed in the A horizon on the C plots compared to the H plots. A critical component of this theory is grass and other plant roots that may have increased the fraction of MIP capable of containing plant-available water. Changes in MIP pore size distribution could have occurred directly by root system expansion or indirectly by the breakdown of macropores, established by roots, into large micropores (Kramer and Boyer 1995).

This mechanism of change in MIP and PAWHC, however, does not explain why lower Bt1 PAWHC in the Beauregard soil was greater on the C plots compared to the H plots. For this horizon, we speculate that clay mobilization rather than roots changed the distribution of micropore size within MIP. This was possible because the H plots had less understory vegetation than the C plots throughout the soil sampling period. This condition lessened soil perturbation by understory rooting. With less soil perturbation, relatively large, vertical, continuous soil pores could have been maintained which enabled the vertical movement of soil water containing fine clay particles from the A and upper Bt1 horizons (Bouma and Dekker 1978). The static nature of large soil macropores on the H plots would have allowed the vertical translocation of fine clay particles and their deposition on the surface of structural aggregates and inside soil pores in the lower Bt1 horizon (Buol and other 2011). With deposition of clay in the lower Bt1 horizon on the H plots, the fraction of small micropores would have increased, and the fraction of MIP capable of storing plant-available water would have decreased. To investigate this theory, the long-term monitoring of soil physical properties at this location will continue, and an assessment of lower Bt1 horizon micromorphological responses to vegetation management treatments will be considered if treatment effects on lower Bt1 PAWHC persist.

Tree Physiology and Soil Physical Property Relationships

A detailed synthesis of long-term loblolly pine production at seven locations concluded that water availability was not the main driver of loblolly pine productivity across this species' natural range (Jokela and others 2004). In support, earlier work by Cregg and others (1990) demonstrated that increases in available soil water induced by 50 and 75 percent reductions in basal area had little effect on loblolly pine water relations on a sandy loam soil in Oklahoma. Adaptation to the dry climate of the western edge of the southern pine region was also observed by Blazier and others (2004). In their study, two sources of loblolly pine, one each originating from a wet and dry location, had equivalent rates of gas exchange and stemwood production on a droughty Oklahoma soil. Others have also observed that water availability in the western edge of the southern pine range did not influence loblolly pine gas exchange unless trees were experiencing prolonged drought (Gravatt and others 1997; Tang and others 2003, 2004).

Our results indicate that the marginalization of water availability as a control of pine production in the Southeast may be due to an overriding effect of the soil physical environment on root system growth. We found that relationships between soil physical properties and longleaf pine tree vigor were primarily apparent in the absence of water deficit (i.e., PWP values > -0.6 MPa) (table 3). Patterns of significance in our regression analyses also indicate that maintenance of some level of understory vegetation may alleviate harsh soil physical conditions. On soil types similar to those in our study, therefore, the physical environment of the soil and the amount and distribution of understory vegetation could have a marked effect on pine vigor as measured by photosynthesis rate and predawn water potential.

Two observations suggested that when moisture was adequate on the C plots, the A horizon maintained a level of control on A_{net} by affecting plant-available water storage. These observations

Table 3--Significant partial coefficients of determination (R^2) and probabilities of a greater *F*-value (Pr > *F*) for multiple regressions between two dependent variables (rate of upper-crown net photosynthesis in the afternoon (A_{net}), predawn needle water potential (PWP)), and 14 independent variables (year, depth to the argillic horizon (DAH), and soil bulk density (BD), microporosity (MIP), macroporosity (MAP), and plant-available water holding capacity (PAWHC) of the A and upper and lower Bt1 horizons). When partial R^2 values were significant, correlation coefficients (*r*) are reported. Regressions were conducted by site (sites 1 and 2), water status (dry and moist), and vegetation management treatment for data with DAH within one standard deviation of the mean DAH of each site

Soil type[a]	Water status	Vegetation management treatment	Dependent variable	Observations (no.)	Significant independent variable	Partial R^2	Pr > *F*	*r*
Ruston	Moist	Control	A_{net}	13	none			
			PWP	13	none			
		Herbicide	A_{net}	14	upper Bt1 BD	0.4651	0.0072	0.6820
				14	upper Bt1 MIP	0.2323	0.0143	-0.4820
			PWP	14	lower Bt1 MIP	0.3853	0.0179	0.6207
				14	A BD	0.1992	0.0423	-0.4463
				14	DAH	0.2300	0.0055	-0.4796
	Dry	Control	A_{net}	9	lower Bt1 BD	0.4917	0.0353	0.7012
			PWP	9	none			
		Herbicide	A_{net}	12	year	0.6726	0.0011	-0.8201
			PWP	12	none			
Beauregard	Moist	Control	A_{net}	15	A MIP	0.7506	<0.0001	0.8664
			PWP	15	none			
		Herbicide	A_{net}	16	A PAWHC	0.4181	0.0068	-0.6466
			PWP	16	upper Bt1 BD	0.4633	0.0037	-0.6807
	Dry	Control	A_{net}	1	none			
			PWP	1	none			
		Herbicide		6	none			
			PWP	6	A MIP	0.7708	0.0214	-0.8780

[a]Site 1 is mostly Ruston fine sandy loam with some Malbis fine sandy loam and Gore very fine sandy loam. Site 2 is mostly Beauregard s loam with some Malbis fine sandy loam.

were a positive correlation between A horizon MIP and A_{net} on the C plots (table 3), and greater A horizon PAWHC and MIP on the C plots compared to the H plots (fig. 3). Absence of a significant MIP-A_{net} correlation on the H plots indicates that this relationship may be important when understory vegetation, grass in particular, is competing with pines for water at young ages (Haywood 2007).

In contrast, on the Beauregard H plots we observed negative relationships between A_{net} and A PAWHC and between PWP and upper Bt1 BD (table 3). One interpretation of these negative relationships is that with the translocation of clay and silt from the A horizon to the argillic horizon, A horizon MIP decreased and A horizon MAP increased (Buol and other 2011). These possible soil porosity responses to clay translocation would naturally reduce A horizon PAWHC. With clay illuviation into the upper Bt1, the BD of this soil layer may have grown closer to creating a barrier to root elongation, restricting pine root system expansion and reducing the effective root-zone. If this were the case, PWP would have become more negative due to root-zone restriction and an increase in upper Bt1 BD. Subsequently, an indirect, negative relationship between A_{net} and A horizon PAWHC would develop.

The relationship between physiological function and BD in the upper Bt1 differed between the two sites, and we believe that these findings reflect the distinct textural differences between the Beauregard and Ruston soils. For example, while both soil types contain equivalent amounts of clay in the A, E, and Bt1 horizons, their silt and sand fractions differ. The Ruston soil contains approximately 46 percent less silt and 54 percent more sand than the Beauregard soil (National Cooperative Soil Survey 2013). Also, the E and Bt1 horizons of the Ruston soil are fine sandy loam and clay loam in texture, respectively, whereas comparable horizons of the Beauregard soil are both silt loam in texture (National

Cooperative Soil Survey 2013). As with the Beauregard soil, clay content in the Bt1 horizon of the Ruston soil could have been correlated with the formation of extreme bulk densities that limited the effective root-zone. However, we speculate that the sand content of the Ruston soil maintained the integrity of macropores and allowed the creation of micropores by the formation of clay coatings (i.e., cutans) on the surface of sand particles and clay bridges between sand particles (Buurman and others 1998). Furthermore, this positive effect of sand on soil porosity and, subsequently, root system expansion superseded any negative effects of high BD on root elongation in the Ruston soil. Because a decrease in MIP implies there was an increase in MAP, this concept is supported by the negative relationship between A_{net} and upper Bt1 horizon MIP (table 3). We attribute the absence of a significant positive correlation between pine vigor variables and upper Bt1 MAP to the inherently high variability of MAP.

With the illuviation of clay in the fine sandy loam layer that is below the clay loam layer in the Ruston soil, the formation of cutans and clay bridges could have favored the fraction of MIP capable of storing plant-available water. Positive correlation between PWP and lower Bt1 MIP reflects this phenomenon (table 3). However, absence of a similar significant relationship between PWP and PAWHC in the lower Bt1 MIP suggests that there are aspects of clay illuviation other than its effect on MIP and PAWHC that also affect pine vigor.

SUMMARY

This report summarizes 8 years of soil physical property responses to the manipulation of understory vegetation on two common forest soils in the West Gulf Coastal Plain. This long-term soil monitoring effort will continue, and the mechanisms of soil change will be explored as warranted. We also describe a short-term study of the relationship between pine vigor and the soil physical environment. Results demonstrate temporal changes in porosity fractions and bulk densities that potentially favor pine root system expansion and represent the natural recovery of Beauregard silt loam and Ruston fine sandy loam soils after disturbance. Differences in soil porosity fractions in response to two extremes in vegetation management suggest that temporal changes in soil physical properties were correlated with the reestablishment of understory vegetation after pine seedlings were planted. In the A horizon, it is possible that soil perturbation by grass and other plants indirectly benefited pine vigor by increasing the water-holding capacity of soil micropores. Soil physical property changes in the upper and lower Bt1 horizon may have been due to clay illuviation. Inconsistent relationships between pine vigor variables and subsoil physical properties between the two soils suggest that in addition to clay illuviation, factors such as subsoil sand content play a role in pine vigor. These results provide insight regarding soil variables of probable importance to root system expansion and pine vigor and the sensitivity of pine vigor to differences in understory vegetation and subsoil texture on the West Gulf Coastal Plain.

LITERATURE CITED

Blake, G.R.; Hartge, K.H. 1986. Bulk density. In: Klute, A., ed. Methods of soil analysis part 1, physical and mineralogical methods. 2d ed. Madison, WI: Soil Science Society of America, Inc: 363-375.

Blazier, M.A.; Hennessey, T.C.; Lynch, T.B. [and others]. 2004. Productivity, crown architecture, and gas exchange of North Carolina and Oklahoma/Arkansas loblolly pine families growing on a droughty site in southeastern Oklahoma. Forest Ecology and Management. 194: 83-94.

Buol, S.W.; Southard, R.J.; Graham, R.C. [and others]. 2011. Soil genesis and classification. 6th ed. Hoboken, NJ: Wiley-Blackwell. 560 p.

Bouma, J.; Dekker, L.W. 1978. A case study on infiltration into dry clay soil I. Morphological observations. Geoderma. 20: 27-40.

Buurman, P.; Jongmans, A.G.; PiPujo, M.D. 1998. Clay illuviation and mechanical clay infiltration - is there a difference? Quaternary Journal. 51-52: 66-69.

Cregg, B.M.; Hennessey, T.C.; Dougherty, P.M. 1990. Water relations of loblolly pine trees in southeastern Oklahoma following precommercial thinning. Canadian Journal of Forest Research. 20:1508-1513.

Dawson, T.E. 1993. Hydraulic lift and water use by plants: implications for water balance, performance and plant-plant interactions Oecologia. 95: 565-574.

Gravatt, D.A.; Chambers, J.L.; Barnett, J.P. 1997. Temporal and spatial patterns of net photosynthesis in 12-year-old loblolly pine five growing seasons after thinning. Forest Ecology and Management. 97: 73-83.

Haywood, J.D. 2007. Influence of herbicides and felling, fertilization, and prescribed fire on longleaf pine establishment and growth through six growing seasons. New Forests. 33: 257-279.

Haywood, J.D. 2009. Eight years of seasonal burning and herbicidal brush control influence sapling longleaf pine growth, understory vegetation, and the outcome of an ensuing wildfire. Forest Ecology and Management. 258: 295-305.

Haywood, J.D. 2011. Influence of herbicides and felling, fertilization, and prescribed fire on longleaf pine growth and understory vegetation through ten growing seasons and the outcome of an ensuing wildfire. New Forests. 41: 55-73.

Jackson, R.B.; Canadell, J.; Ehleringer, J.R. [and others]. 1996. A global analysis of root distributions for terrestrial biomes. Oecologia. 108: 389-411.

Johnson, J.D. 1984. A rapid technique for estimating total surface area of pine needles. Forest Science. 30: 913-921.

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.

Kelting, D.L.; Burger, J.A.; Patterson, S.C. [and others]. 1999. Soil quality assessment in domesticated forests - a southern pine example. Forest Ecology and Management. 122: 167-185

Klute, A. 1986. Water retention: laboratory methods. In: Klute, A., ed. Methods of soil analysis part 1, physical and mineralogical methods. 2d ed. Madison, WI: Soil Science Society of America, Inc: 635-660.

Kramer, P.J.; Boyer, J.S. 1995. Water relations of plants and soils. San Diego, CA: Academic Press. 495 p.

National Cooperative Soil Survey. 2013. National cooperative soil characterization database. Lincoln, NE: NRCS NSSC Soil Survey Lab. http://ncsslabdatamart.sc.egov.usda.gov. [Date accessed: September 16, 2013].

Patterson, W.B.; Sword, M.A.; Brooker, S. 2004. Response of soil physical properties to the application and intensity of prescribed fire in a young longleaf pine (*Pinus palustris* Mill.) stand. 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: 74-78.

Pritchett, W.L. 1979. Properties and management of forest soils. New York: John Wiley. 500 p.

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: 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.

Sword, M.A.; Tiarks, A.E. 2002. Local soils information needed to define the root zone in process models on the Gulf Coastal Plain. Gen. Tech. Rep. SRS-58. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 15 p.

Sword Sayer, M.A.; Brissette, J.C.; Barnett, J.P. 2005. Root growth and hydraulic conductivity of southern pine seedlings in response to soil temperature and water availability after planting. New Forests. 30: 253-272.

Sword Sayer, M.A.; Haywood, J.D. 2012. Surface soil root response to season of repeated fire in a young longleaf pine plantation. 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: 342-343.

Tang, Z.; Chambers, J.L.; Sword Sayer, M.A.; Barnett, J.P. 2003. Seasonal photosynthesis and water relations of juvenile loblolly pine relative to stand density and canopy position. Trees Structure and Function. 17: 424-430.

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.

Van Lear, D.H.; Kapeluck, P.R.; Carroll, W.D. 2000. Productivity of loblolly pine as affected by decomposing root systems. Forest Ecology and Management. 138: 435-443.

Warren, J.M.; Brooks, J.R.; Meinzer, F.C.; Eberhardt, J.L. 2008. Hydraulic redistribution of water from *Pinus ponderosa* trees to seedlings: evidence for an ectomycorrhizal pathway. New Phytologist. 178: 382-394.

Warren, J.M.; Meinzer, F.C.; Brooks, J.R. [and others]. 2006. Hydraulic redistribution of soil water in two old-growth coniferous forests: quantifying patterns and controls. New Phytologist. 173: 753-765.

STOCK SIZE AFFECTS EARLY GROWTH OF A LOBLOLLY PINE CLONE

David B. South, Al Lyons, and Russ Pohl[1]

For decades, forest researchers in the South have known that early gains in survival and growth of loblolly pine (*Pinus taeda* L.) can be achieved by planting large-diameter seedlings (South 1993; Wakeley 1949). For *P. radiata*, increasing size of planting stock also increases early growth of both seedlings (Mason and others 1996) and cuttings (South and others 2005). Stock is now produced from somatic embryogenesis (Grossnickle and Pait 2008). Although some landowners have paid $0.40 each for such seedlings, data on the effect of stock size on field-growth of tissue-cultured stock are lacking. Does stock size still make a difference when planting clones?

To address this question, studies were installed in Alabama (cutover site) and Georgia (grassland site) to determine the effects of stock size on early height growth of a tissue-cultured clone (L-3576). The stock was provided by the CellFor Corporation (no longer in business). Loblolly pine transplants (mini-plugs) were grown at the Pearl River Nursery in Mississippi as bareroot stock, and they were sorted into three classes according to root-collar diameter (3 to 4.9 mm, 5 to 6.9 mm, and 8 to 9.9 mm). Trees were planted in block plots, four replications per site, with 100 or 105 trees per plot (1,075 seedlings per ha). Survival after 5 years was greater than 97 percent at both sites (table 1). Trees in grassland plots with the largest diameter class exhibited the greatest height and diameter growth. At year 5, the difference in height between the smallest and largest class ranged from 0.3- to 1.2-m, and basal area was increased by 4- to 23-percent. Data from these trials suggest that early growth performance of one loblolly pine clone can be affected by stock size, but statistical significance depends on site variability. The cutover site was more variable and therefore provided less statistical significance.

In addition to absolute gains, an estimate of the time gain (South and Miller 2007) was made using annual height measurements. By plotting height over time (fig. 1) and using linear equations, a hypothetical number of days required to reach a height of 5 m was calculated, assuming growth rate is constant throughout the year. For the cutover site, the height was reached 3 months sooner for the 6 mm stock (versus 4 mm), and on the grassland site the time was 4 months sooner. Assuming a plantation is harvested when the current annual increment (CAI) is 16 m^3/ha, then 3 months of additional growth would be equivalent to an additional 4 m^3/ha of wood.

In these two trials, the null hypothesis (size had no effect on early growth) was rejected. It may be wise for researchers who compare early growth of various clones to document initial differences in stock size at time of planting.

LITERATURE CITED

Grossnickle, S.C.; Pait, J. 2008. Somatic embryogenesis tissue culture for applying varietal forestry to conifer species. In: Dumroese R.K.; Riley, .L.E, tech. cords. National proceedings: forest and conservation nursery associations - 2007. RMRS-P-57. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 135-139.

Mason, E.G.; South, D.B.; Weizhong, Z. 1996. Performance of *Pinus radiata* in relation to seedling grade, weed control, and soil cultivation in the central North Island of New Zealand. New Zealand Journal of Forestry Science. 26: 173-183.

South, D.B. 1993. Rationale for growing southern pine seedlings at low seedbed densities. New Forests. 7: 63-92.

South, D.B.; Menzies, M.I.; Holden, D.G. 2005. Stock size affects outplanting survival and early growth of fascicle cuttings of *Pinus radiata*. New Forests. 29: 273-288.

South, D.B.; Miller, J.H. 2007. Growth response analysis after early control of woody competition for 14 loblolly pine plantations in the southern US. Forest Ecology and Management. 242: 569-577

Wakeley, P.C. 1949. Physiological grades of southern pine nursery stock. In: Proceedings of the 49th annual meeting of the Society of American Foresters. Bethesda, MD: Society of American Foresters: 311-322.

[1]Emeritus Professor, School of Forestry and Wildlife Sciences, Auburn University, AL 36849; Manager, Hancock Forest Management, Harpersville, AL 35078; and Chief of Reforestation Department, Georgia Forestry Commission, Macon, GA 31202.

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Table 1--Effect of initial stock size on 5th year survival, diameter at breast height (d.b.h.), basal area per tree, and height of loblolly pine clone L-3576. Probabilities of a significant linear contrast are given. HCDN is hypothetical calendar day number when stock reached a height of 5 m in 2010 and was estimated using simple linear equations

Location	Diameter class	Survival	D.b.h.	Basal area	Height	HCDN to reach 5 meters
		%	*cm*	*cm²*	*m*	*days*
Cutover site	4 mm	97.2	7.1	41.7	5.1	339
	6 mm	99.6	8.0	51.4	5.5	253
	9 mm	98.8	8.2	54.8	5.4	254
	probability	0.3106	0.0586	0.0897	0.4774	
Grassland site	4 mm	98.8	10.6	90.6	5.3	267
	6 mm	98.8	11.3	94.9	5.9	139
	9 mm	98.8	11.7	108.0	6.5	52
	probability	0.9870	0.0068	0.0076	0.0066	

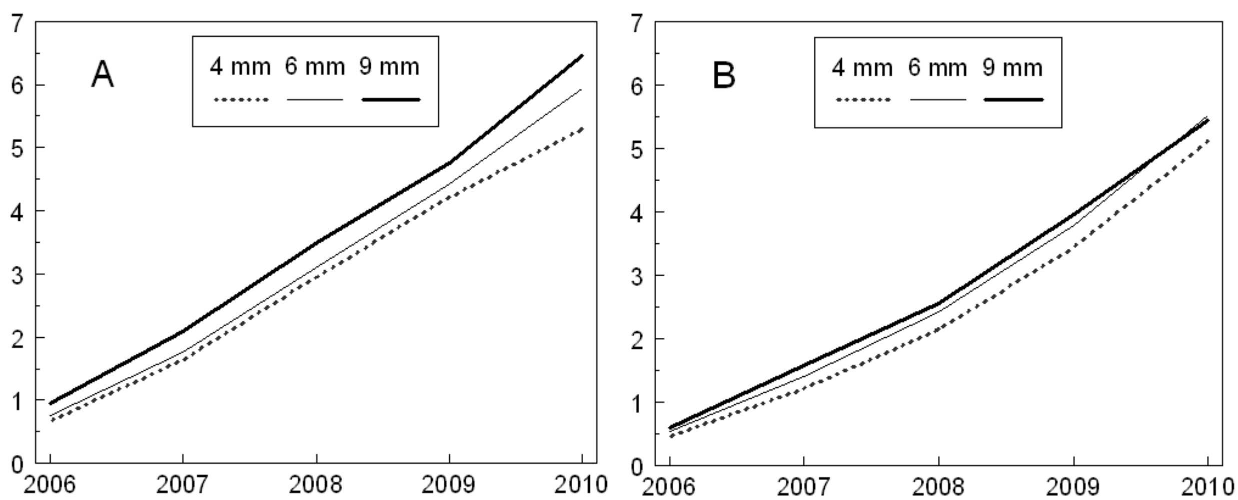

Figure 1--Effect of initial stock size on height (m) of loblolly pine clone L-3576 on a (A) grassland site and (B) cutover site.

YIELD AND FINANCIAL PERFORMANCE ESTIMATES OF FOUR ELITE LOBLOLLY PINE SEED SOURCES PLANTED IN THE WESTERN GULF REGION

Michael A. Blazier and A. Gordon Holley[1]

Eastern seed sources of loblolly pine (*Pinus taeda* L.) have been planted in the Western Gulf region for nearly three decades because they often have higher growth rates than local seed sources. However, productivity gains for eastern families are sometimes offset by poorer survival rates relative to local families. Clonal propagation of loblolly pine seedlings has further increased productivity gain potential of loblolly pine, but survival and yields of eastern varieties grown in the Western Gulf must be determined. In addition, varietal seedlings cost nearly five times that of conventional open-pollinated seedlings. It is essential to determine the effects of this greater establishment cost on plantation financial performance in order to develop economically feasible management strategies for varietal plantations.

Two eastern open-pollinated families (7-56 and 8-103), two eastern varieties (Clone 93 and Clone 9), and one mixture of Western Gulf open-pollinated families (LA OP) where planted at the LSU AgCenter Hill Farm Research Station in northwest Louisiana with the goal to determine survival and growth trends, estimate forest product yields, and estimate financial performance. Trees were planted at a 6 foot by 16 foot spacing in January 2005. Survival, total tree height, and dbh were measured annually since planting in 2005. Age seven (2012) measurements of codominant height, basal area, trees/acre were used as inputs for the FASTLOB growth and yield model to estimate product yields of a 25-year rotation. Modeled stand densities reaching between 35 percent and 55 percent of maximum stand density index dictated simulated stand thinning.

All seed sources exhibited survival rates greater than 80 percent. However, 7-56 showed significantly poorer survival than 8-103, Clone 9, and Clone 93 (fig. 1-A). By age 4 significant height differences emerged among the five seed sources. Clone 93 had the tallest height growth, while 8-103 and LA OP were the shortest of the seed sources (fig. 1B). Clones 9 and 93 had the highest predicted pulpwood and Chip-n-saw yields. The open-pollinated 7-56 had the highest predicted sawtimber yields, although 7-56, 9, and 93 had the highest total yields across all product classes (fig. 1-C). Although all sources achieved a greater than 8 percent return on investment, the open-pollinated sources achieved higher internal rates of returns above the two clonal varieties, which were at least two percent lower (fig 1D).

Although both clone 9 and 93 and 7-56 showed greater productivity than the Western Gulf mixture on this site, the productivity gains were not sufficient to overcome the greater initial cost of the seedlings and planting (Table 1), which led to lower internal rates of return. High survival rates of the clones led to high proportions of pulpwood and chip-and-saw using the simulated stand density index thinning approach. The clonal management scenario had higher cost and led to a greater proportion of lower value forest product yields relative to the open pollinated seedlings. These results indicate the need for different management strategies to capture the higher productivity of the clonal varieties such as increasing the planting spacing to reduce initial cost and increase the proportion of sawtimber yields, thus allowing an earlier final harvest.

[1]Associate Professor, Louisiana State University AgCenter Hill Farm Research Station, Homer, LA, 71040 and Associate Professor, Louisiana Tech University School of Forestry, Ruston, LA, 71272

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Table 1--Cost assumptions for internal rate of return calculations

Year	Activity	Seed Source(s)	Cost ($/acre)
0	Subsoil	All	89.06
0	Hand planting, bareroot	7-56, 8-103, LA OP	22.95
0	Hand planting, container	9, 93	44.10
0	Seedling cost	LA OP	18.00
0	Seedling cost	7-56, 8-103	27.00
0	Seedling cost	9, 93	157.50
1	Two band applications herbicide	All	69.93

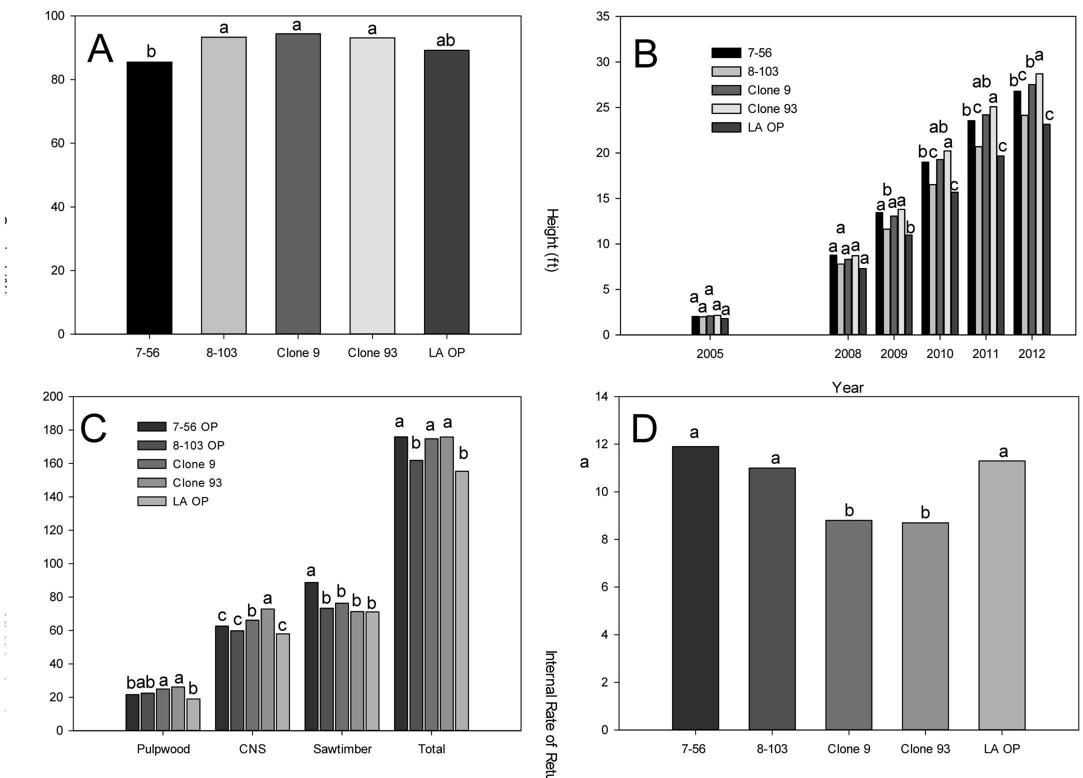

Figure 1. Survival averaged over seven years (A), height (B), predicted yields by product class for a 25-year rotation (C), and predicted internal rates of return (D) for four loblolly pine seed sources at the LSU AgCenter Hill Farm Research Station in northwestern Louisiana. Seed sources 7-56, 8-103, and LA OP are half-sib families; seed sources 9 and 93 are clonal varieties. For each variable, bars headed by different letters differ at $P < 0.05$.

Fire Effects

Moderators:

Brian Oswald
Stephen F. Austin State University
Arthur Temple College of Forestry and Agriculture

and

Morgan Varner
Mississippi State University
College of Forest Resources

PRESCRIBED FIRE EFFECTS IN A LONGLEAF PINE ECOSYSTEM-ARE WINTER FIRES WORKING?

Rebecca J. Barlow, John S. Kush, John C. Gilbert, and Sharon M. Hermann[1]

Abstract--Longleaf pine (*Pinus palustris* Mill.) ecosystems once dominated 60 to 90 million acres and supported one of the most diverse floras in North America. It is well-known that longleaf pine ecosystems must burn frequently to maintain natural structure and function. This vegetation type ranks as one of the most fire-dependent in the country and must burn frequently (multiple times a decade) for natural structure and function to be maintained. Frequent fires maintain relatively low fuel loads, so many burns do not directly affect adult longleaf trees. However all species are immediately affected by each fire that burns through a stand. Because many resident species are perennials that re-sprout after fires, it likely takes multiple burns to change the plant assemblage of the ground layer. There is a need is for better insight into fire effects on small woody stems in the ground layer. In 1984 a long-term study was established on the Escambia Experimental Forest in Brewton, Alabama to study the impact of fire on longleaf pine growth. Spring and winter burns at 2-, 3-, and 5-year return intervals were implemented and have been continued since that time. Hardwood species composition from each of the season of burn and fire frequency treatments will be discussed. Winter burning has not removed what are considered to be fire-intolerant species such as water oak (*Quercus nigra* L.) and sweetgum (*Liquidambar styraciflua* L.), from the landscape. These species will make future fires more difficult to make and eventually make it difficult to regenerate longleaf pine.

INTRODUCTION

Descriptions exist of the southeastern landscape before European settlement (e.g. Bartram 1791) and by all accounts longleaf pine (*Pinus palustris* Mill.) dominated the landscape. At this time, fire was ever-present and the most important ecological process responsible for persistence of longleaf pine forests (Burns and Honkala 1990, Chapman 1932, Croker and Boyer 1975, Wahlenberg 1946, Walker and Wiant 1973). The original longleaf pine ecosystem had a groundcover dominated by perennial grasses and forbs that was maintained by a mosaic of fire, both in time interval and season. Reports at the end of the 1800s suggest that south of the Fall Line, much of the region supported pine-dominated ecosystems (Mohr 1896), most maintained by frequent, low-severity fire. There were accounts of hardwood forests as well, but at such small acreages that they were not reported in timber values (Mohr 1896, Sargent 1884). Fire regimes varied across the landscape depending on topography, soil-type, and biotic factors related to abiotic differences; current estimates suggest burn frequencies of one to five events or more each decade. Open-canopy pine forests served as the landscape matrix with other ecosystems imbedded in them.

In the modern landscape, the acreage of longleaf pine forests has dramatically declined due to conversion to agricultural use, urban development, or plantation forestry. The naturally regenerated stands that remain are often fire suppressed or burned only infrequently. This is especially problematic for longleaf stands (Noss and others 1995). Prescribed fire helps control disease such as brown spot needle blight, eliminates some stems and foliage from woody competition, and promotes seedling establishment and growth by eliminating excessive litter on the forest floor (Chapman 1932, Croker and Boyer 1975, Walker and Wiant 1973). Natural regeneration of longleaf is generally unsuccessful unless seed falls on bare soil; in addition, native ground layer of grasses and forbs declines without fire. Although Outcalt (2000) estimated that slightly more than 80 percent of natural longleaf stands on public property had been burned at least once in the previous 5 years, less than 40 percent of private property stands were burned during the same time period.

Longleaf pine has the potential, if actively managed, to meet modern forestry goals. However, because there was a period of time in U.S. history where fire was not promoted as a management tool, many landowners and land

[1]Associate Extension Professor, Auburn University, Alabama Cooperative Extension System, School of Forestry and Wildlife Sciences, Auburn University, AL 36849; Research Fellow and Research Associate, respectively, Auburn University, School of Forestry and Wildlife Sciences, Auburn University, AL 36849; and Assistant Professor, Auburn University, Department of Biological Sciences, Auburn University, AL 36849.

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managers today are uncomfortable with burning their forestland for fear of killing their crop trees. Longleaf pine is not loblolly (*Pinus taeda* L.) or slash pine (*Pinus elliottii* Engelm.) and evolved under different conditions than the other southern pines. It evolved with fire and regenerated in dense stands that formed in openings created by some disturbance event. Thinning techniques can be used to develop canopy openings to capture regeneration when trees are old enough to produce viable seeds. This in conjunction with an early and active burning regime can promote the development of high-quality sawlogs and understory plant species that are preferred browse and forage for wildlife (Haywood and others 1998, Walker and Wiant 1973). Without fire, hardwood competition will dominate the understory and eliminate or dramatically reduce regeneration opportunities and forage availability in longleaf pine forests.

By considering fire regimes that include season and frequency of burn, modern management goals may be better achieved. This paper examines the results of a long-term study that was established in 1984 on the Escambia Experimental Forest in Brewton, Alabama by Dr. William Boyer to examine 2-, 3- and 5-year fire-return intervals on longleaf pine stands. The objective is to better understand the impact of season and timing of burn on longleaf pine forests, and in particular, the effect it has on hardwood competition in these forests.

MATERIALS AND METHODS

This study was initiated in 1984 on the Escambia Experimental Forest near Brewton, AL to study potential growth losses in frequently burned longleaf pine forests. Specifically, this study examines impact of winter (January/February) and spring (April/May) burns on growth and mortality of longleaf pines. Three study blocks of 9-year-old longleaf pine trees were established on the forest. Efforts were made to locate them in relatively close proximity to each other and on areas of similar site quality. On these blocks, 0.1-acre measurement plots were established, and trees were thinned to 40, fairly uniformly spaced crop trees. Measurements taken at time of study establishment showed no significant differences in species composition or size of trees across all plots.

Prescribed fires were initiated in the winter (January/February) and spring (April/May) of 1985. Flank or strip-head fires were used to minimize crown scorch. Fires were set in periods following soaking rains, when fine fuel moisture was 7 to 10 percent, relative humidity was 35 to 55 percent, and winds were steady. Following that initial burn, stands were burned under similar conditions on 2-, 3-, or 5-year intervals. There was also a no-burn treatment for comparison.

All pine and hardwood trees that were at least 1 inch in diameter at breast height (d.b.h., where breast height = 4.5 feet above the ground level) were measured for diameter to the nearest 1 inch and total height to the nearest 1 foot at the time of study initiation. Similar measurements continued to be taken at 5-year intervals, measuring both trees that were present at the start of the study as well as ingrowth. Treatments by both season and timing of burn were compared at the 95 percent level for differences in d.b.h., height, and basal area through time.

RESULTS

With regard to longleaf pine growth and mortality, this study found no significant differences between burn and no burn treatments until age 19 when basal area was lower on burn plots regardless of time interval. At age 24, height, longleaf pine d.b.h., and basal area were lower on burn plots. By age 29, only height was comparatively lower on burn treatments. There was no significant loss of longleaf pine trees due to mortality at this time. When comparing different seasons and frequency of burns, there were no significant differences among treatments for d.b.h. and basal area through age 29. Height of the overstory longleaf pine trees, however, was significantly lower on 2-year burn compared to 3- and 5-year burn intervals.

Results are varied, however, for hardwood stems on the plots. By age 12, winter 5-year and no-burn plots had significantly higher numbers of hardwood stems and hardwood basal areas compared to other treatments. The winter 5-year and no-burn treatments were not significantly different from each other at this time in terms of hardwood stems and basal area. By age 24, hardwood density on no-burn plots was significantly higher than the winter 5-year burn and all other treatments.

When comparing the diameter distribution of hardwood stems per acre on the unburned sites, the number of stems has continued to increase since the initial measurement in 1984. At the time of the most recent measurement in 2009, there were not only as many 1-inch stems as there were in 1984, but there were more than two times the number of 2- and 3-inch stems. There were also stems in the 4- through 10-inch classes that were not found on the site at the time of study initiation. Fire-intolerant species such as water oak (*Quercus nigra* L.), and sweetgum (*Liquidambar styraciflua* L.) make up the majority of the hardwood stems found on these plots.

There were more hardwood stems present on winter burns overall. For example, winter 5-year burn plots saw increases in the diameter of understory hardwoods into the 2- to 8-inch diameter classes over the last 24 years, but the number of stems was less than that of the unburned treatment. Measurements taken in 2009 show that winter 2- and 3-year burns had more hardwood stems in larger diameter classes than were present at the time of study initiation. However there are fewer 1-inch d.b.h. stems on these treatments than there were in 1985. Oaks and dogwoods (*Cornus* spp.) comprised the majority of stems > 1.5 inches d.b.h. on these winter burn plots, while mostly water oak and sweetgum were found to make up the smaller diameter classes.

Over the same 24-year period, spring burns were found to control almost all hardwood stems that were on site at the beginning of the study regardless of timing. By 2009 neither the spring 2-year nor the spring 3-year burns had any measurable hardwood stems > 1 inch d.b.h. The spring 5-year treatments had fewer than 50 stems per acre of hardwoods in the 1-inch d.b.h. class. Species composition of hardwoods on the spring burn sites were predominantly oaks and dogwoods.

DISCUSSION

Growth losses in longleaf pine this study did not compare similarly to those found in prior studies (Boyer 1987). After 20+ years the longleaf on unburned plots were growing as well or better than the burn treatments. So, one might wonder, “Why bother burning longleaf?”

Based on the results of this study, conducting prescribed fires in the winter on a 3- to 5-year cycle alone is probably not enough to control hardwood competition. Winter fires did not remove hardwood competition as well as growing season burns, even spring burns on a longer rotation. In addition, winter burning did not remove what are considered to be fire-intolerant species such as water oak and sweetgum. As these hardwoods grow, they produce increasing amounts of leaf litter that does not burn as efficiently as pine litter, thus limiting the effectiveness of prescribed fire. This promotes a cycle in which hardwood stems then multiply in the absence of fire forming thick “islands” of hardwood brush. It is more difficult for fire to travel through these “islands” allowing hardwoods to continue to grow in diameter and height, eventually making their way into the overstory.

Once fire has been excluded from a forested site for long periods of time, it likely takes multiple burns over many years to change the plant assemblage of the established ground layer. Although there was no longleaf pine regeneration on these sites as the overstory density was too high, without fire hardwood competition can eliminate or dramatically reduce longleaf pine regeneration opportunities, resulting in the eventual loss of the system. We must understand forest stand dynamics, impacts on native understory, and forest structure to better manage these forests in the future.

CONCLUSION

How best to promote longleaf pine as a major species in the South can be a bit puzzling with the prevailing diversity of management interests. Based on the information presented above, there are options for prescribing fire at varying intervals and seasons to help landowners meet their objectives.

So in answer to the question, “Why burn longleaf?”, as Dr. H.H. Chapman (1932), Yale Professor of Forestry, writes:

> “In the longleaf pine type of the south (and nowhere else in North America to the writer's knowledge) fire at frequent but not necessarily annual intervals is as dependable a factor of site as is climate or soil.”

The longleaf pine ecosystem evolved with fire and is adapted to its presence on the landscape.

Fire is needed to promote and maintain both the trees and the system through time.

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LITERATURE CITED

Bartram, W. 1791. Travels through North and South Carolina, Georgia, East and West Florida [1955 reprint]. New York: Dover Publications. 414 p.

Boyer, W.D. 1987. Volume growth loss: a hidden cost of periodic prescribed burning in longleaf pine? Southern Journal of Applied Forestry. 11(3): 154-157.

Burns, R.M; Honkala, B.H., tech. cords. 1990. Silvics of North America: Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture. 675 p. Vol. 1.

Chapman, H.H. 1932. Is the longleaf pine type a climax? Ecology. 13(4): 328-334.

Croker T.C.; Boyer, W.D. 1975. Regenerating longleaf pine naturally. Res. Pap. SO-105. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 22 p.

Haywood, J.D.; Boyer, W.D.; Harris, F.L. 1998. Plant communities in selected longleaf pine landscapes on the Catahoula Ranger District, Kisatchie National Forest, Louisiana. 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: 86-91.

Mohr, C. 1896. The timber pines of the southern United States. Bull. 13. Washington, DC: U.S. Department of Agriculture Division of Forestry. 176 p.

Noss, R.F.; LaRoe, E.T.; Scott, J.M. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Biol. Rep. 28. Washington, DC: U.S. Department of the Interior Fish and Wildlife Service, National Biological Service. 58 p.

Outcalt, K.W. 2000. Occurrence of fire in longleaf pine stands in the southeastern United States. In: Proceedings, 21st Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station. 21: 178-182.

Sargent, C.S. 1884. Report on the forests of North America (exclusive of Mexico): 10th census report. Washington, DC: U.S. Department of the Interior, Census Office. Vol. 9. 612 p.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth, and management. 1st ed. Washington, DC: Charles Lathrop Pack Forestry Foundation. 429 p. In cooperation with: The U.S. Department of Agriculture Forest Service.

Walker, L.C.; Wiant, H.V., Jr. 1973. Silviculture of longleaf pine. Nacogdoches, TX: Stephen F. Austin State University, School of Forestry. 105 p.

SPROUTING CAPABILITY OF SHORTLEAF PINE SEEDLINGS FOLLOWING CLIPPING AND BURNING: FIRST-YEAR RESULTS

David C. Clabo and Wayne K. Clatterbuck[1]

Abstract--Shortleaf pine (*Pinus echinata* Mill.) is one of the few southern pine species with the ability to sprout after disturbance during the seedling age range, but little is known about sprouting success based on the type of disturbance. This study evaluates sprouting success after controlled burning conditions or manually clipping as compared to untreated controls of planted shortleaf pine 1-0 seedlings approximately 1 month after planting and on subsequent sprout production and growth one growing season following planting. As part of a larger study, randomized plots of 50 seedlings (3 blocks per treatment) were planted on February 25, 2011 at the University of Tennessee Cumberland Forest located in the foothills of the Cumberland Mountains in Morgan County, TN. The burn and clip treatments were conducted in April 2011. Survival, number of sprouts, and height of the tallest sprout were recorded for each seedling in the winter of 2012-2013. The clip treatment and the control had the same survival rate (75.3 percent) and displayed greater survival than the burn treatment. Clipping produced more sprouts and taller sprouts on average compared to the burn treatment, yet the clip treatment sprouts were approximately half the height of the control seedlings. More data on seedling response to these disturbances at older ages will be collected as the study continues. One-year-old planted seedlings do not appear to show high survival rates or produce prolific numbers of sprouts in response to early growing season burns.

INTRODUCTION

The native range of shortleaf pine (*Pinus echinata* Mill.) covers a vast area of approximately 440,000 square miles from eastern Texas north to New Jersey and Pennsylvania (Lawson 1990). The species typically grows on dryer, well-drained sites but can be found in a variety of topographic positions and soils throughout its range. The species can thrive on sites with poor edaphic conditions, primarily due to its extensive taproot, and is capable of forming nearly pure stands (e.g. Ouachita Mountains of Arkansas, and previously the Cumberland Plateau of Tennessee) (Coffey 2012, Lawson 1990, Williams 1998). The species' good growth form (self-pruning ability), disease resistance, fire tolerance, and cold hardiness make it a suitable constituent species for many management objectives (Phelps and Czabator 1978, Guldin 1986). The ability to sprout sets shortleaf pine apart from other southern pine species.

Shortleaf pine sprouts prolifically after the stem is damaged or killed. The species can sprout from the seedling through the pole size ranges, and individual stems may exhibit this sprouting response up to 6 to 8 inches in diameter at breast height (d.b.h.). Other tree species decline in areas with frequent disturbance, whereas shortleaf pine's sprouting ability allows it to continue to occupy an area after repetitive disturbances (Lawson 1990). Sprouting is enabled by a J-shaped basal crook, which contains axillary dormant buds. The crook forms 2 to 3 months after germination, and sprouts typically appear from the root collar directly above the basal crook (Guldin 1986, Lilly and others 2010).

The shortleaf pine resource has been declining. USDA Forest Service Forest Inventory and Analysis data have shown a decrease of stems ≥ 1 inch d.b.h. since the early 1980s (Oswalt 2012). In the middle to second half of the 20th century, factors such as fire suppression, declines in free-range livestock grazing, southern pine beetle outbreaks, and increased urbanization have combined to reduce the prevalence of shortleaf pine across its native range (Birch and others 1986, Coffey 2012). Industry preference for loblolly pine (*P. taeda* L.)

[1]Graduate Research Assistant and Professor, respectively, The University of Tennessee, Department of Forestry, Wildlife and Fisheries, Knoxville, TN 37996-4563.

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and succession of shortleaf pine stands into mixed hardwood stands have contributed to the species' decline as well (Birch and others 1986, Dennington 1992).

Interest in restoring degraded shortleaf pine ecosystems, such as shortleaf pine-bluestem and shortleaf pine-oak savannas, has increased over the last several years, especially in the western portion of the species' range (Elliot and others 2012, Guldin 2007, Guldin and others 2004). Adequate regeneration of shortleaf pine from both artificial and natural means is necessary for restoring these systems. Regeneration from sprouting is typically more advantageous in situations where new age cohorts are desired due to unpredictable seed production and the exacting environmental conditions that are often required to successfully perpetuate shortleaf pine from seed (Guldin 1986, Lawson 1986). Few studies have investigated clipping (Campbell 1985) and fire (Cain and Shelton 2000, Lilly and others 2012) to determine their effects on shortleaf pine sprouting, especially in favorable areas east of the Mississippi River such as the Cumberland Plateau region of Tennessee.

OBJECTIVES

The goals of this study were to: (1) determine survival differences among 1-year-old seedlings that were burned or clipped early in the growing season and untreated controls, (2) compare sprouting numbers among the three treatments, and (3) compare differences in dominant sprout height between burned and clipped seedlings. In addition, this study sought to determine if a relationship existed between the number of sprouts and the height of the tallest sprout on seedlings that survived the treatments.

METHODS

The study was located at the University of Tennessee Forest Resources Research and Education Center's Cumberland Forest Unit in Morgan County, TN. This area of Tennessee is part of Walden Ridge, a subregion of the Cumberland Plateau, and is characterized by broad, rolling ridges and weakly dissected plateau surface (Smalley 1982). A previously maintained 5,796 square foot square field was chosen for planting 1-0 stock bare-root shortleaf pine seedlings on February 25, 2011. The seedlings were purchased from the Tennessee Division of Forestry Nursery at Delano, TN. Fifty seedlings were planted on 1- by 1-foot spacing in 4- by 9-foot plots oriented from north to south. Soils consisted of fine-loamy, siliceous, semiactive, mesic Typic Hapludults on 5 to 12 percent slopes from the Lonewood series with a shortleaf pine site index of 70 feet at base age 50 years (NRCS 2012). Three treatments each within three blocks were designated. The three treatment plots in each block are part of a larger, longer term study. The three treatments analyzed in this study included an early growing season burn conducted on April 14, 2011, early growing season clipping applied on the same date, and an untreated control.

For all three burn plots, the same methodology was used to ensure similar burning conditions across blocks. Proximate white pine (*P. strobus* L.) plantations provided needles that were used as a fuel source for ground fires. Needles were dried approximately 2.5 hours in full sunlight during mid-afternoon on April 14, 2011 and placed within burn plots in approximately equal volumes using 5-gallon buckets to ensure similar burn conditions. Homogeneity among burns (duration and temperature) was determined using a stopwatch and a Kintrex digital infrared thermometer, which determined burn temperatures approximately 70 inches away from the center of each plot. Temperatures were recorded every 15 seconds until complete flame-out.

Seedlings that received the clip treatment were cut approximately 1 to 2 inches above ground level so as not to damage the basal buds and limit sprouting. In addition to clipping the main stem, any other sprouts below the 1-inch threshold were clipped to reduce variation among seedlings.

Survival counts and measurements of seedlings/sprouts were carried out in January

2013, one full growing season after treatments. In order to assist counts and measurements, grasses and weeds were clipped and a 2-ounce-per-gallon solution of glyphosate (Cornerstone Plus®) was applied in September 2012 by sponge-wicking around sprouts and seedlings. Determination of whether a seedling was dead or alive, number of sprouts per seedling, and height of the tallest sprout per seedling were recorded for each treatment. Analysis of variance was used to test for treatment differences with each variable. Data were analyzed as a randomized complete block experimental design using PROC MIXED in SAS 9.3 (SAS Institute 2012). Least squares means were separated using Fisher's protected least significant difference, and a significance level of P = 0.05 was used for all analyses. Data for each variable were transformed as needed using either square root or arc sine square root transformations. Untransformed means and standard errors are reported for each analysis.

RESULTS

No significant differences were found among survival rates (P = 0.067), yet the burn treatment had a much lower numeric survival rate than either the clip or control treatments, which had the same rate of survival (table 1). For the burn plots, temperatures ranged from 512° to 770 °F, and flame-out occurred in 6 minutes on all three plots. The analysis of dominant sprout height indicated significant differences among treatments (P = 0.001). The clip treatment produced dominant sprouts that were approximately 7 inches taller on average than the burn treatment. Compared to the untreated control seedlings, the clip treatment seedlings were approximately 11 inches shorter on average (table 2). The cumulative height of the control seedlings over the 2 years was 48.6 +/- 2.9 inches. Significant differences in sprout numbers among treatments were found (P = 0.003). The clip treatment produced more sprouts on average than the burn treatment. Both the clip and burn treatment seedlings produced more sprouts than the untreated control seedlings (table 3). The correlation between sprout height and sprout number for the burn and clip treatments was significant (P = 0.0001). The relationship was moderately negative (r = -0.433), indicating that as dominant sprout height increased, sprout number decreased.

Table 1—Mean survival percentages and standard errors one full growing season after treatments were applied. Surviving seedlings were 2 years old at time of counting. BM is burning in March; CL is clipping in March; and CO represents untreated controls

Treatment	Mean	Std. error
	--%--	
BM	42.6a	0.078
CL	75.3a	0.078
CO	75.3a	0.078

[a]Treatments with the same letter in the mean column do not differ significantly at P = 0.05.

Table 2—Mean heights and standard errors one full growing season after treatments were applied. Seedlings were 2 years old when measured. Height of the tallest sprout was measured for each sprout clump. BM is burning in March; CL is clipping in March; and CO is the second-year growth of the untreated controls

Treatment	Mean	Std. error
	inches	
BM	19.2a	2.97
CL	26.2b	2.90
CO	37.5c	2.89

[a]Treatments with the same letter in the mean column do not differ significantly at P = 0.05.

Table 3—Mean and standard errors of the number of sprouts (green needles present) per seedling produced one full growing season after treatments were applied. Seedlings were 2 years old when sprouts were counted. BM is burning in March; CL is clipping in March; and CO represents untreated controls

Treatment	Mean	Std. error
	--no.--	
BM	4.8a	0.49
CL	6.2a	0.42
CO	1.3b	0.42

[a]Treatments with the same letter in the mean column do not differ significantly at P = 0.05.

DISCUSSION AND CONCLUSION

The clip and control treatments had similar survival rates (75.3 percent). Little and Somes (1956) reported that clipping near the root collar where the majority of the dormant buds are located could reduce the number of sprouts. Care was taken in this study to clip the seedlings 1 to 2 inches above the ground line or the root collar to ensure that the dormant buds were not damaged. Although the formation of new roots has been observed as the single most important factor in survival of planted bare-root southern pine seedlings, the presence of undamaged dormant buds on the clipped seedlings did not impact survival when compared to the controls (Brissette and Chambers 1992). A clipping study with 4-year-old seedlings in Arkansas revealed that clipping in February at similar heights above ground level had lower survival rates at 48 percent compared to the 1-year-old seedlings in this study (Campbell 1985). In another related study on the Coastal Plain of Arkansas, untreated controls had a 91 percent survival rate that was greater than the survival rate in this study (Cain and Shelton 2000). Differences in survival rate between studies could be attributable to the planting shock associated with lifting, storage, transport and planting processes for seedlings, planting site variations, and soil fertility. The burn treatment had a lower survival rate than the clip or control treatments. The survival rate for the burn treatment was similar to that reported by for late dormant season/early growing season burns on 4- to 6-year-old seedlings (37 percent) (Lilly and others 2012). Variations in litter/soil depth, flame intensity, seedling bark thickness, and seedling age can impact the insulation of the basal, dormant buds and seedling survival after burns (Cain and Shelton 2000, Lilly and others 2012, Little and Somes 1956).

Height differences between the clipped and burned seedlings differed statistically. The short amount of time between planting and treatment application (7 weeks) may have contributed to these height differences. Most seedlings following planting are under soil-water-root and carbohydrate-storage stress as they try to establish a root system in their new environment (Grossnickle 2005, Rietveld 1989). Seedling growth resources were stretched during establishment after outplanting as well as in responding to a disturbance, both of which affected the above-ground stem. Coupled with the absence of thick bark that older seedlings and saplings have to protect them from fire, the burn probably set back height growth in the 1-year old seedlings much more so than clipping alone (Guldin 1986, Fan and others 2012). The burn killed or damaged the basal buds, resulting in less of an ability to sprout and in fewer sprouts that were shorter in height than the other two treatments. The control seedlings were 1.4 times as tall as the clipped seedlings and 1.9 times as tall as the burn seedlings over one full growing season, indicating that the species is capable of fast growth in open conditions on these plateau sites.

No significant differences for sprout number existed among treatments. Burned and clipped seedlings both produced more sprouts than the controls, demonstrating that seedlings are more likely to produce sprouts with disturbance than without it. Numerically, clipping produced more sprouts (6.2 +/- 0.4) than burning (4.8 +/- 0.4). Once again, unlike the buds for seedlings in the clip treatment, burning most likely damaged the basal buds. Also unlike this study, in a similar study focused on the effects of fertilizer and sprouting on older 6-year- old seedlings in New Jersey, the mean number of sprouts produced was significantly different between burning and clipping (Grossmann and Kuser 1988). In a study with older seedlings (Lilly and others 2012), more sprouts on average (8.8 +/- 0.7) occurred with burning than in this study. Young seedlings do not appear to be capable of producing as many sprouts as older seedlings.

The moderately negative correlation between dominant sprout height and sprout number probably was due to limits on resource allocation, especially in recently outplanted seedlings. Factors such as root-water relations, root-soil relations, a lack of fine roots, and weather conditions such as temperature and moisture can affect how well the seedlings respond to disturbance and resultant sprout growth and frequency (Burdett 1990, Rietveld 1989). This relationship may change as seedlings become more established and with increasing age, which would allow them to recover from disturbances more easily.

One-year-old clipped seedlings had better survival rates, attained greater heights, and produced more sprouts than burned seedlings. This study was conducted under open growing conditions. However, seedlings growing in partial shade may perform favorably because shortleaf pine has more shade tolerance in the seedling age range than in mature trees (Shelton 1995). The early results of this study and past research suggests that managers interested in favoring shortleaf pine over other species through burning should wait to implement treatments until the trees are more than 1-year-old (especially for outplanted shortleaf pine) to obtain better survival, sprouting numbers, and heights (Fan and others 2012). More research is needed to determine how well seedlings sprout and grow at different ages and with burns applied during various times throughout the year.

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LITERATURE CITED

Birch, T.W.; Hansen, M.H.; McWilliams, W.H.; Sheffield, R.M. 1986. The shortleaf resource. In: Murphy, P.A., ed. Proceedings, symposium on the shortleaf pine ecosystem. Monticello, AR: University of Arkansas Cooperative Extension Service: 9-24.

Brissette, J.C.; Chambers, J.L. 1992. Root zone environment, root growth, and water relations during seedling establishment. In: Brissette, J.C.; Barnett, J.P., comps. Proceedings of the shortleaf pine regeneration workshop. Gen. Tech. Rep. SO-90. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 67-76.

Burdett, A.N. 1990. Physiological processes in plantation establishment and the development of specifications for forest planting stock. Canadian Journal of Forest Research. 20: 415-427.

Cain, M.D.; Shelton, M.G. 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.

Campbell, T.E. 1985. Sprouting of slash, loblolly, and shortleaf pines following a simulated precommercial thinning. Res. Note SO-320. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 3 p.

Coffey, C. 2012. The history of shortleaf pine on the Cumberland Plateau. 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. Alabama Agricultural Experiment Station Special Rep. No. 11. Auburn, AL: Auburn University, Alabama Cooperative Extension System: 2-6.

Dennington, R.W. 1992. Making natural regeneration methods work with shortleaf pine. . In: Brissette, J.C.; Barnett, J.P., comps. Proceedings of the shortleaf pine regeneration workshop. Gen. Tech. Rep. SO-90. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 167-171.

Elliot, K.J.; Vose, J.M.; Knoepp, J.D.; Clinton, B.D. 2012. Restoration of shortleaf pine (*Pinus echinata)*-hardwood ecosystems severely impacted by the southern pine beetle. Forest Ecology and Management. 274: 181-200.

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.

Grossmann, M.N.; Kuser, J.E. 1988. Rooting primary-leaved sprouts of pitch and shortleaf pine. Northern Journal of Applied Forestry. 5:87, 159.

Grossnickle, S.C. 2005. Importance of root growth in overcoming planting stress. New Forests. 30: 273-294.

Guldin, J.M. 1986. Ecology of shortleaf pine. In: Murphy, P.A., ed. Proceedings, symposium on the shortleaf pine ecosystem. Monticello, AR: University of Arkansas Cooperative Extension Service: 25-40.

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 2003 National Silviculture Workshop. Proceedings RMRS-P-34. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 182-190.

Lawson, E.R. 1986. Natural regeneration of shortleaf pine. In: Murphy, P.A., ed. Proceedings, symposium on the shortleaf pine ecosystem. Monticello, AR: University of Arkansas Cooperative Extension Service: 53-63.

Lawson, E.R. 1990. *Pinus echinata* Mill. Shortleaf pine. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America: Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 316-326. Vol. 1.

Lilly, C.J.; Will, R.E.; Tauer, C.G. [and others]. 2010. Factors influencing shortleaf pine sprouting after fire. In: Shortleaf pine research: past, present, and future. A symposium. Stillwater, OK: Oklahoma State University, Department of Natural Resource Ecology and Management [NREM]: 8-9.

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. Sta. Pap. NE-81. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 14 p.

NRCS [Natural Resources Conservation Service]. 2012. Web soil survey. Washington, DC: U.S. Department of Agriculture National Resources Conservation Service. http://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx . [Date accessed: March 21, 2012].

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. Alabama Agricultural Experiment Station Special Rep. No. 11. Auburn, AL: Auburn University, Alabama Cooperative Extension System: 33-37.

Phelps. W.R.; Czabator, F.L. 1978. Fusiform rust of southern pines. Forest Insect & Disease Leaflet 26. Washington, DC: U.S. Department of Agriculture, Forest Service. http://www.na.fs.fed.us/spfo/pubs/fidls/fusiform/fidl-fusi.htm. [Date accessed: March 16, 2012].

Rietveld, W.J. 1989. Transplanting stress in bareroot conifer seedlings: its development and progression to establishment. Northern Journal of Applied Forestry. 6:99-107.

SAS Institute. 2012. SAS version 9.3. Cary, NC: SAS Institute.

Shelton, M. 1995. Effects of seed production, seedbed condition, and overstory basal area on the establishment of shortleaf pine seedlings in the Ouachita Mountains. Res. Pap. SO-293. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 13 p.

Smalley, G.W. 1982. Classification and evaluation of forest sites on the Mid-Cumberland Plateau. Gen. Tech. Rep. SO-38. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 58 p.

Williams, R. 1998. Effects of fire on shortleaf and loblolly pine reproduction and its potential use in shortleaf/oak/hickory ecosystem restoration. 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: 321-325.

TIMING FIRE TO MINIMIZE DAMAGE IN MANAGING OAK ECOSYSTEMS

Daniel C. Dey and Callie Jo Schweitzer[1]

Abstract--The long history of fire in North America spans millennia and is recognized as an important driver in the widespread and long-term dominance of oak species. Early European settlers intensified the occurrence of fire from about 1850 to 1950, with dates varying by region. This resulted in much forest damage and gained fire a negative reputation. The lack of fire for the past 50 years due to suppression programs is now indicted as a major cause of widespread oak regeneration failures. Alarms are sounding for the continued loss of oak forests. The use of prescribed fire is increasing in forest management and ecosystem restoration. An understanding of fire effects on trees can provide the basis for the silviculture of restoring and sustaining oak ecosystems. We present an overview of fire-tree wounding interactions, highlight important determinants of fire injury and damage, and discuss several practical situations where fire can be used to favor oak while minimizing damage and devaluation of the forest. We also identify stages in stand development, regeneration methods, and management objectives for which fire has the potential of causing substantial damage and alternative practices should be preferred.

INTRODUCTION

A major impetus for forming state and federal forestry agencies was to avert a national timber famine by instituting science-based forest management, halting timber theft, and stopping forest destruction due to frequent and catastrophic wildfires (Keefe 1987, Pyne 1982, Pyne and others 1996, Steen 1976). It was widely recognized that the high levels of decay and cull timber in eastern hardwood forests were due to a history of wildfires (Burns 1955, Gustafson 1944, Hepting 1937, Kaufert 1933). American Indians frequently lit fires for numerous reasons, and these fires burned unsuppressed over large landscapes before European immigration to North America (Guyette and others 2002, 2012a). European settlers initially adopted American Indian burning practices, increasing the frequency of fire and saturating the landscape with fire (Guyette and others 2002). Forest fires are capable of causing wounds at the base of tree boles, and wounds can become quite large with increasing fire intensity. These wounds provide entry points to wood-decaying fungi that can cause substantial loss of wood volume and value over time (Guyette and others 2012b; Hepting 1935, 1941; Hesterberg 1957; Loomis 1974; Stambaugh and Guyette 2008). The cumulative effects of a history of fire persist for decades in forests because trees are long-lived organisms. Therefore, it is not surprising that there is a trend of higher cull percent of live total net volume with increasing historic fire frequency in the eastern United States (fig. 1). The highest levels of live cull in standing timber today occur in the Southern, Great Plains Border, and former Prairie Peninsula regions where fires were historically more frequent. Great Plains Border states commonly experience seasonal drought and cyclical periods (e.g., 21 to 22 years) of severe drought (Stambaugh and Guyette 2004) that promote higher-intensity fires and more severe tree wounding. Also, woods burning in the South and hill country of the Midwest persisted longer than in other regions of the eastern United States due to cultural differences (Pyne 1982).

Foresters have been working hard for the past 75 years or so to suppress wildfires and get private landowners to stop indiscriminate woods burning. Their efforts have been highly successful. For example, about 2,500 human-caused wildfires burn annually in Missouri, but the average fire size is 10 acres due to fire suppression (Westin 1992). Consequently, the percent of cull live timber in Missouri has decreased from about 50 to 18 percent since the 1950s (Burns 1955, Miles 2013). In the past 10 to 30 years, prescribed burning to restore oak/pine (*Quercus* spp./*Pinus* spp.) savannas and woodlands has increased on public lands, especially in the Great Plains Border Region.

[1]Research Forester, USDA Forest Service, Northern Research Station, Columbia, MO 65211; and Research Forester, USDA Forest Service, Southern Research Station, Huntsville, AL 35811.

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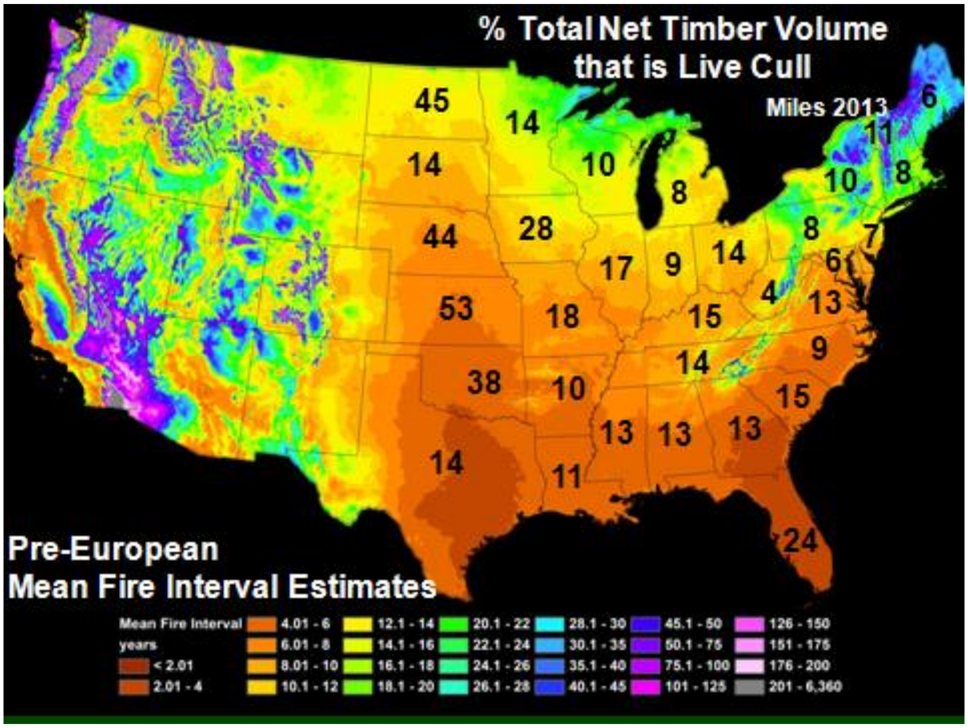

Figure 1--Estimated mean fire interval for low-intensity fires before European settlement circa 1650-1850 (Guyette and others 2012) and percent of total net timber volume that is live cull in modern times according to a national forest inventory (Miles 2013).

For example, the Mark Twain National Forest Plan (MTNF 2005) calls for restoration of woodlands and savannas on 438,000 acres or 29 percent of the MTNF. In addition to these efforts, other federal and state agencies, The Nature Conservancy, National Wild Turkey Federation, and other NGOs are using prescribed burning to restore thousands of acres of woodlands and savannas throughout Missouri.

Efforts to restore native communities on this large-scale followed several decades of debate among resource professionals over the reintroduction of fire, especially in regions where it was a hard-won fight to get people to quit burning their woods. Improvements in timber quality and decreases in the amount of cull in forests following fire suppression were strong testimony to the benefits of keeping fire out of the woods. In this era of ecosystem restoration, using fire to restore native communities puts emphasis on ecological benefits such as increased native plant diversity and improved habitat quality for species that prefer woodlands and savannas. However, age-old concerns about fire damage to trees and forests remain.

The purpose of this paper is to provide a brief background on fire damage and the factors that influence damage in trees in the management of oak-dominated forests. Several management scenarios will be used to explore the appropriateness of fire at key stages in stand regeneration and development. Arthur and others (2012) provided an excellent synthesis of the role of fire in the life cycle of an oak forest with an emphasis on biology and ecology. We used a similar life cycle approach to select the scenarios for discussion. They are common stand conditions and developmental stages that are key break points in sustaining oak forests and woodlands.

TYPES OF FIRE INJURY AND DAMAGE

Tree Mortality

Surface fires in an eastern hardwood forest are capable of killing larger mature trees of any species if the intensity and duration of heating is sufficient to cause death of the cambium and foliage. Ground temperatures in low-intensity fires can be high enough to kill the tree cambium if the duration of heating is sufficiently long (Dey and Hartman 2005, Elliott and Vose 2005, Hutchinson and others 2005). However, thick bark is capable of protecting trees from complete girdling of the stem almost regardless of species (Guyette and Stambaugh 2004, Smith and Sutherland 1999). In mixed oak forests of the Central Hardwood Forest region, high percentages of overstory trees may be scarred on the lower bole from low-intensity fires but relatively small amounts of the overstory (e.g., < 5 percent basal area of trees > 4.5 inches d.b.h. or < 8 percent of stem density of trees > 16 feet tall) are lost to mortality from single or repeated low-intensity fires (Dey and Fan 2009, Hutchinson and others 2005, Regelbrugge and Smith 1994, Smith and Sutherland 2006). Higher fire intensity or longer exposure to fire is needed to kill larger (e.g., > 10 inches d.b.h.) trees, and this may occur locally during low-intensity fires where accumulations of litter, debris, or downed tree tops occur next to the base of individual large trees. Wildfires can cause stand replacement when they burn under severe fire weather, which is more likely to occur in years of severe drought.

Bole Wounding and Wood Decay

Prescribed fires, even of low intensity, can cause wounds to overstory tree stems, though not all trees are wounded, even when their bark may be charred (Smith and Sutherland 1999). Whether a tree is wounded or not depends largely on fire behavior (i.e., temperature, flame length, and duration of heating) at any one location within the burn unit, and bark characteristics of the tree (i.e., thickness, texture and heat conduction properties). Fire scars provide opportunities for wood-decaying fungi to colonize and infect tree stems. Large scars with exposed wood that remain open and moist for long periods provide good environments for fungal colonization and development. However, fire scars are often small and the bark commonly remains intact, covering the injury after low-intensity fires in upland oak forests of the Central Hardwood region (Smith and Sutherland 1999). Loss of volume and value in fire scarred oak trees may be relatively minor in the short-term (< 10 years), but with time, advanced decay can result in substantial value losses (Guyette and others 2012b, Stambaugh and Guyette 2008). Considering that about one-third of the total standing tree board-foot volume is in the butt 8-foot log, fire injury leading to wood decay at the base of a tree has significant potential effects on harvest volume and value. Even where timber production is not the primary management concern such as in woodland and savanna restoration, the longevity of mature overstory trees may be compromised by advance decay in the boles of fire-scarred trees because trees are more susceptible to stem breakage and blowdown during wind and ice storms (Guyette and Stambaugh 2004).

Stem Top-kill

Low-intensity fires are capable of causing death of the entire cambium on smaller diameter trees of any species. The bark of seedlings and saplings is relatively thin and offers less insulating protection to the cambium than mature, large diameter trees for any species (Hare 1965). Complete stem girdling results in the death of the shoot above the point of cambial injury. Many hardwood species are able to produce vegetative sprouts after one episode of top-kill (Dey and Hartman 2005, Regelbrugge and Smith 1994). Whether fire injury by top-kill is a benefit or considered damage depends on management objectives and the stage of stand development (Arthur and others 2012). Top-kill is a positive fire effect when used to favor the development of large and competitive oak reproduction by reducing the density of the mid- and overstory, or to increase the competitive status of oak regeneration by reducing the growth or density of competitors (Brose and others 2013). Repeated fires that cause mortality or top-kill of woody stems are desirable when trying to reduce stem density and forest cover in woodland and savanna restoration. In contrast, repeated top-kill of hardwood sprouts can adversely retard the recruitment of oaks and other desirable reproduction into the overstory, causing years of lost growth and delaying maturity.

DETERMINANTS OF FIRE INJURY AND DAMAGE

Trees can resist being injured by fire or they can minimize the damage following injury by defensive responses that confine damage (e.g., wood decay) to the area of initial injury.

Tree Species

Species-specific growth strategies and morphological characteristics result in different responses among species following fire, with oaks generally better-adapted to persist following burning than many competitors. The susceptibility to cambial death and top-kill by a single fire is nearly equal for seedlings and smaller sapling-sized stems, almost regardless of species. Mortality is high in the smallest of seedlings and new germinants, even in the oaks (Johnson 1974). However, large oak seedlings and saplings are better able to persist with repeated burning than their major competitors (Brose and others 2013). In general, oak species have a distinct advantage over competitors for surviving fire because they preferentially allocate carbohydrates to root growth and have an abundance of dormant buds commonly located in the soil where they are insulated from the heat of a fire (Brose and Van Lear 2004; Iverson and Hutchinson 2002; Iverson and others 2004, 2008; Johnson and others 2009). Nonetheless, oak stems < 4 inches d.b.h. are susceptible to top-kill, but the larger stems have a high capacity to persist by sprouting (Dey 1991), especially when there is adequate light for growth during the fire-free period. However, sprouting ability varies by species and begins to decline beyond a species-specific diameter threshold, which is usually in the pole-sized and small sawtimber size classes (Dey and others 1996, Johnson and others 2009). Lastly, species differences in ability to resist fire injury become more pronounced in the larger diameter size classes, and this has much to do with differences in bark characteristics (see below).

Tree Size

Size influences a tree's ability to sprout after fire-caused top-kill, as do the amount of root carbohydrate reserves and the presence of viable dormant vegetative buds after the fire (Dey and Hartman 2005). Low-intensity fires commonly cause top-kill of hardwood trees < 4 inches d.b.h. and a significant proportion of trees < 8 inches d.b.h. (Dey and Hartman 2005, Green and others 2010, Waldrop and others 1992). Guyette and Stambaugh (2004) found that post oak (*Q. stellata* Wangenh.) trees that were most likely to be scarred and survive a low-intensity dormant season fire were 4 to 8 inches d.b.h.; smaller trees were either top-killed or died. Larger seedlings and saplings of most hardwood species are able to sprout after top-kill caused by a single fire (Dey and Hartman 2005, Iverson and others 2008). For most species, sprouting capacity reaches a maximum with increasing diameter to a threshold size and then declines with further increases in diameter (Dey and others 1996, Johnson and others 2009). When large diameter oak trees in the overstory are girdled by fire, they are completely killed, being unable to produce sprouts. It is in the smaller size classes where oak trees are generally better able to persist after repeated fires than similar sized stems of their competitors (Brose and Van Lear 1998, Dey and Hartman 2005, Kruger and Reich 1997). Red maple (*Acer rubrum* L.) can be a troublesome species that competes with oak. If it is allowed to grow to sapling or pole-size, it becomes a persistent sprouter even after several low-intensity fires in the dormant season (Blankenship and Arthur 2006, Chiang and others 2005).

Bark Characteristics

There are many properties of a tree's bark that influence its ability to insulate the cambium from the heat of a fire: thickness, texture, thermal conductivity, specific heat, and thermal diffusivity. However, it is bark thickness that largely determines the degree of protection of the cambium from lethal temperatures (Vines 1968). As tree diameter increases so does bark thickness, and the degree of insulating protection increases exponentially with small increments in bark thickness (Hare 1965). Guyette and Stambaugh (2004) found that the probability of fire scarring and the percent of bole circumference scarred were significantly and negatively related to tree diameter, bark width, radial growth rate and tree age in post oak (d.b.h. range 4 to 28 inches). Sutherland and Smith (2000) reported that the probability of surviving a fire increases at the sapling size (2 to 4 inches d.b.h.) when the bark starts to achieve sufficient thickness to prevent top-kill, depending on species. Similarly, Guyette and Stambaugh (2004) observed that post oak trees > 4 inches d.b.h. were more likely to survive low-intensity fires without top-kill. There is however a substantial variation in bark thickness, rate of bark growth on the lower bole, and bark texture among species (Harmon 1984, Hengst and Dawson 1994, Sutherland and Smith 2000). Even with thick bark, scarring can occur in areas of bark fissures, creating a pattern of smaller injuries distributed around the circumference of the tree (Guyette and Stambaugh 2004).

In general, upland species have thicker bark than bottomland species for similar sized trees (Sutherland and Smith 2000). Bark thickness in white oak group species (*Quercus* Section *Quercus*) is the greatest followed by the red oak group species (*Quercus* Section *Lobatae*) in the Central Hardwood Forest region. Species with inherently thinner bark include American beech (*Fagus grandifolia* Ehrh.), flowering dogwood

(*Cornus florida* L.), black cherry (*Prunus serotina* Ehrh.), maple (*Acer* spp.), and hickory (*Carya* spp.). The rate of bark thickening during growth is important because faster growth rates allow trees to reach critical thresholds of thickness earlier that are associated with protection of the cambium and survival. Eastern cottonwood (*Populus deltoides* Bart. ex Marsh.) and yellow-poplar (*Liriodendron tulipifera* L.) are both thin-barked, fire-sensitive species when trees are small and young, but they have rapid rates of bark growth and are considered resistant to fire scarring as large mature trees. In contrast, silver maple (*A. saccharinum* L.) has a slow rate of bark growth all its life and is vulnerable to fire injury even when it is a large tree. Species that have smooth bark texture such as water oak are more vulnerable to fire injury to the cambium than are deeply fissured, rough textured species such as chestnut oak (*Q. prinus* L.) and bur oak (*Q. macrocarpa* Michx.).

Defense Against Decay

Diameter growth rate--determines how long a fire scar may provide entry of fungi into the tree's stem, once a wound is opened. In the event of an exposed fire scar, trees with faster rates of diameter growth are able to close the wound sooner, thus minimizing the time the wound face is available for fungal colonization. By sealing the wound, the tree also creates a less favorable environment for wood decay (Sutherland and Smith 2000). High rates of diameter growth more rapidly restore full vascular cambial functioning after fire scarring of the bole (Smith and Sutherland 2006). Growth near the area of injury (wound wood ribs) can be faster than on other portions of the bole (Smith and Sutherland 1999).

Compartmentalization--is a process whereby trees are able to establish a protective boundary surrounding cells injured by fire. The boundary is the result of the formation of tyloses and production of waxes, gums, and resins to form a barrier to further cell desiccation and microbial infection. The ability to compartmentalize injuries varies by species. The birches (*Betula* spp.) are less effective at compartmentalizing stem wounds than maples and oaks (Sutherland and Smith 2000). Oak species, especially those in the white oak group, have an unusual ability to rapidly compartmentalize fire injuries (Smith and Sutherland 1999, Sutherland and Smith 2000). Smith and Sutherland (1999) found that low-intensity dormant season fires produced relatively small scars (scorch height < 40 inches above the ground) that were often concealed by intact bark and were effectively and rapidly compartmentalized in black oak (*Q. velutina* Lam.) and chestnut oak trees (d.b.h. range 4 to 22 inches).

Decay resistance of the heartwood--varies by species and is important to retarding decay that originates from fire scarring. Species of the white oak group, black locust (*Robinia pseudoacacia* L.), catalpa (*Catalpa* spp.), black cherry, cedar (*Juniperus virginiana* L.), and cypress (*Taxodium* spp.) have heartwood that is resistant to very resistant to decay (Forest Products Laboratory 1967). Red oak group species, hickories, maples, sweetgum (*Liquidambar styraciflua* L.), yellow-poplar, birches, eastern cottonwood, and American beech have only slight to no resistance to heartwood decay.

Scar Size and Time Since Wounding

Fungi that infect tree boles through logging or fire scars can cause substantial loss of value and degrade in timber quality over several decades (Hesterberg 1957). Stambaugh and Guyette (2008) found that one third of the volume can be defect in white oak (*Q. alba* L.), black oak, and scarlet oak (*Q. coccinea* Muenchh.) butt logs within 25 years after the trees received a fire scar. The proportion of butt log that was defect after fire scarring increased with increasing size of fire scar (from 155 to 930 square inches) and decreased with increasing size of tree (from about 8 to 22 inches d.b.h.) at time of scarring. Loomis (1974) observed that the potential for volume and value loss from decay following fire scarring increased with diameter of tree at the time of scarring if the decay is allowed to progress for 2 or more decades. The rate and extent of heartwood decay depends on species, which vary in their heartwood resistance to decay (Forest Products Laboratory 1967).

Guyette and others (2012b) reported that both value and volume loss to decay and lumber degrade in black oak, northern red oak (*Q. rubra* L.), and scarlet oak butt logs increased with increasing prescribed fire severity and initial fire scar size as represented by scar height and scar depth. They reported that average scar height was 34 inches (range 6 to 154 inches) and scar depth was 2.6 inches (range 0.1 to 15 inches). Most of the devaluation in the butt log resulted from declines in lumber grade and not from volume loss. However, they found that percent scaled volume loss averaged only 4 percent, and value loss averaged 10 percent after 9 to 14 years since the fire. They concluded that where < 20 percent of the bole circumference was scarred and scar heights were < 20 inches that value loss would be insignificant within 15 years of scarring. Loomis (1974) confirmed that value and volume loss increased with increasing fire scar size (wound width and length), time since wounding, and tree diameter at the time of scarring. Similar evidence of the extent of fire injury was noted by Smith and Sutherland (1999) who measured scorch height on oak boles and found that it was generally < 40 inches after low-intensity prescribed fires in Ohio. They observed that most wounds occurred near the ground and were covered by intact bark, small in size, and rapidly and effectively compartmentalized within 2 years of the fire. Thus, losses due to wood decay can be minimized if fire intensity is low and scarred trees are harvested before decay becomes advanced.

The stage of stand development and tree size at the time of fire scarring may influence the probability that decay will substantially reduce wood volume or value by the time the tree is harvested. Fire scars on small diameter trees that survive the injury are necessarily small in size because they are limited by tree size. Closure of the wound is rapid if the tree is vigorous and free-to-grow; this minimizes the likelihood of fungal infection. Large diameter trees are better protected from fire scarring by their thick bark, and wounds tend to be small and low on the bole in low-intensity fires. These trees are merchantable and may be removed in a timber harvest soon after the fire, before any decay develops. Also, injuries generally occur on the large end of the butt log and therefore they are often outside of the scaling cylinder and do not effect product recovery and value. Fire-scarring of mid-sized trees that will remain in the stand for 30 years or more are most at risk of advanced decay development and significant loss of volume and value by the time they are harvested. Pole-sized and small sawtimber trees can sustain large-sized scars that take time to heal, during which time they are prone to fungal infections, especially on moist scars, which provide more receptive surfaces for fungi. Prolonged moisture in scars is more likely to occur when scars are in contact with the ground or when they are shaped such that they trap water.

In the next section we present several common scenarios in oak forest and woodland management where managers may want to use fire. We also discuss the consequences of burning stands at various times in the life cycle of an oak forest in terms of fire damage to trees and the stand.

SCENARIO 1: MATURE FOREST WITH NO OAK ADVANCE REPRODUCTION

Prescribed fire can be used to prepare the seedbed for a good acorn crop or in advance of artificial regeneration of oak by direct seeding or planting (Dey and others 2008a, 2012). Fire can reduce: (1) the physical barrier to oak seedling establishment created by deep litter, (2) seed of competitors stored in the forest floor, and (3) woody competitor density and structure in the mid- and understory. Litter in Central Hardwood forests accumulates rapidly after burning and in 4 years can return to 75 percent of pre-burn amounts (Stambaugh and others 2006). Midstory release is effective for a number of years but hardwood sprouts from top-killed stems will again begin competing with oak reproduction, although their rate of recovery is reduced by higher levels of overstory stocking (Dey and Hartman 2005, Lockhart and others 2000, Miller and others 2004). Therefore, fires may need to be repeated to manage litter depth for adequate oak seedling establishment and to sustain control over competing vegetation to favor oak seedling establishment and the development of large oak advance reproduction. Because bumper acorn crops occur only periodically and oaks grow slowly under low to moderate understory light levels, it may take 10 to 30 years to develop adequate numbers of large oak advance reproduction using combinations of stand thinning or shelterwood harvesting and prescribed burning. Scarring of merchantable stems or trees that will become merchantable by the time of harvest may lead to substantial loss of volume and value due to decay over 20 to 30 years (Loomis 1989, Stambaugh and Guyette 2008). An alternative method of midstory removal is to use an herbicide application to individual stems, the benefits of which include the avoidance of stem wounding by fire, the prevention of hardwood sprouts from undesirable species, and fewer treatments required for sustained control of competing species. Alternatively, the midstory may be mechanically removed, which avoids fire scarring of residual trees, but it does not prevent sprouting from cut stems.

SCENARIO 2: MATURE FORESTS WITH ABUNDANT SMALL OAK ADVANCE REPRODUCTION

This is a common situation in eastern oak forests, especially following a bumper acorn crop. Small oak advance reproduction (< 1 foot tall and 0.25 inches in basal diameter) have low regeneration potential, and midstory removal or shelterwood harvesting is often recommended to reduce stand density and deliver more light to the forest floor to promote oak seedling growth. Prescribed fire can be a useful tool for controlling competing woody stems that are < 4 inches d.b.h., but it has the potential to cause high mortality in small oak seedlings. Therefore, Brose (2008) and Brose and others (2013) recommend encouraging oak seedling growth with a shelterwood harvest that removes about 50 percent of the initial stand basal area, to about B-level stocking, and burning either just before or several years after final overstory removal. Once oak seedlings have become large (e.g., ≥ 0.75 inches basal diameter), then moderate- to high-intensity fires can increase the relative abundance of competitive oak reproduction (Brose 2011). Waiting as long as possible to conduct the release burn to allow the oak seedlings to grow increases their capacity to sprout vigorously following top-kill from the fire. Basal diameter in oak is an indicator of the size of the root system, which drives sprout growth (Dey and Parker 1997, Knapp and others 2006). For several years after each shelterwood harvest, oak seedlings will benefit from increased light levels. Monitoring the reproduction helps determine the need for and timing of prescribed burning. If the shelterwood is completely removed in 3 to 5 years after the initial cut, then fire scarring is not an issue. Scarring of residual trees that are retained for the long-term for wildlife or aesthetic purposes may reduce their longevity due to advanced decay in the lower bole, which renders trees more susceptible to breakage or blowdown in storms.

SCENARIO 3: STAND INITIATION STAGE AFTER FINAL SHELTERWOOD REMOVAL OR CLEARCUTTING

During the stand initiation stage (Oliver and Larson 1996) following clearcutting or final removal of the shelterwood, prescribed burning is effective in promoting oak dominance over competing woody vegetation, provided the oak advance reproduction is present in sufficient density before harvesting. Periodic fires (e.g., every 3 to 5 years) are useful for increasing the relative abundance of competitive oak seedlings (Brose and others 2013). Moderate to intense fires during early leaf out discriminate more in favor of oak if the oak reproduction is large (Brose 2011). As long as the majority of stems are < 4 inches d.b.h., burning, within typical prescriptions, will cause top-kill throughout the stand of reproduction, which with time will favor oak dominance. There are no long-term deleterious effects of burning at this stage of stand development unless there are larger overstory trees retained for wildlife habitat, aesthetics or other long-term purposes. Scarring of these stems may reduce their life span. At some point, burning must stop for a sufficiently long period to allow seedling sprouts to recruit into the overstory. This may take 10 to 30 years depending on growth rates and source of reproduction. Reproduction from stump sprouts grow initially more rapidly than seedling origin reproduction, reaching 2.3 to 3.1 inches d.b.h. in 10-year-old clearcuts in the Missouri Ozarks (Dey and others 2008b). White oak saplings that are codominant in Missouri clearcuts grow 1.5 inches in diameter per 10 years on sites of average site quality; at this rate it would take 20 years for a small diameter (1 inch d.b.h.) sapling to reach 4 inches d.b.h. and begin to improve its chances of surviving being top-killed by a low-intensity fire (Shifley and Smith 1982). A sufficient fire-free period is crucial to permit recruitment into the overstory.

SCENARIO 4: STEM EXCLUSION STAGE, CROWN CLOSURE

When regenerating stands reach the stem exclusion stage (Oliver and Larson 1996), continued use of prescribed burning indiscriminately causes top-kill and retards stand development. Setting back a stand at this point results in the loss of 20 years of growth. If oak trees still require release to maintain adequate stocking of dominant stems at this stage, it is better to use mechanical or chemical release methods applied as a crop tree or area-based thinning. The risk of fire scars in larger saplings at this stage can result in substantial degrade and volume loss at the time of harvest, especially if the wounds are large enough to remain open to fungal infections for several years.

SCENARIO 5: STANDS MANAGED BY UNEVEN-AGED METHODS

The use of uneven-aged methods, primarily single-tree selection, is not recommended for sustaining oak forests on mesic and hydric sites; however, there is evidence in the xeric forests of the Missouri Ozarks that it may be possible to sustain white oak forests by this method (Johnson and others 2009, Loewenstein 2005). Application of prescribed burning with single-tree selection management is highly likely to cause large amounts of defect in trees by the time they reach sawtimber size. In this silvicultural system, trees are harvested to simultaneously promote regeneration and recruitment into the overstory. Trees of all sizes exist in the stand, and sapling, poles and small sawtimber that sustain fire scars are likely to remain in the stand for decades before being harvested, thus, permitting time for advanced decay to develop. Also, the growth of trees in the mid- and understory is reduced by overstory stocking, and this increases the time it takes for fire scars to heal. Burning in uneven-aged stands also can disrupt the distribution of age classes because seedlings and saplings are susceptible to being top-killed or dying. With repeated burning, the regenerating cohort will be concentrated into a single (or few) age classes. The use of group selection has been advocated for oak regeneration because it provides more light to the regeneration than the single-tree method. However, controlling competing vegetation before and after harvesting is problematic in small random openings located throughout the forest. Without large oak advance reproduction at time of harvest, and control of competing vegetation, typically group openings are dominated by non-oak species (Jenkins and Parker 1998, Weigel and Parker 1997). The use of fire to control competing vegetation in isolated group openings is operationally impractical due to the small size of openings, lack of natural fire breaks around openings, and matrix of uncut or single-tree selection forest that is vulnerable to fire injury and decay.

SCENARIO 6: SAVANNA AND WOODLAND RESTORATION

Savannas and woodlands were once much more abundant across the landscape in the eastern United States, especially in the border region of the tallgrass prairie and eastern deciduous forests. An increasingly common management goal is to restore these ecosystems where forests now prevail. A primary objective is to reduce stand density using prescribed fire to promote development of native grass and forb ground flora typical of these communities (e.g., Nelson 2010). A challenge in restoration is how to reduce the density of larger overstory trees that have developed over the past 50 years or more since the commencement of fire suppression programs. Moderate- to high-intensity fires are needed to reduce overstory density in the larger size classes, which incidentally have the potential to scar the residual overstory trees and reduce their longevity in the overstory. Fire is also less specific about which trees are removed and which remain. An alternative to using fire to reduce stand density is to conduct a timber harvest. This permits recovery of wood products, avoids the problem of fire scarring residual trees, and provides better control over the distribution and composition of the final overstory. Lower intensity fires can be combined with timber harvesting and mechanical/chemical thinning to achieve other ecological objectives and control small hardwood sprouts. When it is time to replace the overstory in woodlands and savannas, a fire-free period is necessary for recruitment. Often there is present large oak advance reproduction because partial overstory density and periodic fire promote oak reproduction.

CONCLUSION

Small diameter trees that survive being burned can only have small wounds because if they had large wounds they would be completely girdled and suffer top-kill or mortality. If they are in a dominant competitive position and are vigorous, they can heal quickly, preventing fungal infection and rapidly compartmentalizing the injured tissue. Large diameter trees are harder to scar by fire due to their thicker bark. Because these larger trees are merchantable, the time to harvest may be nearing, and this limits wood decaying fungi from causing much volume or value loss. Fire scars on the lower end of the butt log are often outside the scaling cylinder and therefore do not affect product recovery or value. It is pole and small sawtimber-sized trees that are at greatest risk of sustaining large scars and remaining in the stand long enough to develop substantial decay. In oak forests and woodlands, prescribed fire is most useful to prepare for and manage regeneration of desirable species. It can be used without causing considerable loss in stand volume or value when incorporated as part of an even-aged silvicultural system. Intermediate-aged stands are at high risk to fire injury and damage; alternatives to fire are preferred for managing stand composition, growth, and quality in these stands. In any case, individual large-diameter trees are at risk of fire damage if the intention is to retain them for the long-term and they are subjected to high-intensity fires.

The use of prescribed fire in restoring and sustaining oak ecosystems does not have to have the same outcome as the history of wildfire, with its resultant forest damage and high amounts of standing live cull volume. Fire was an integral driver of the widespread distribution and dominance of oak, especially on mesic, high-quality sites. There are alternative management practices that can achieve similar outcomes, but sometimes fire is the most effective tool for achieving ecological objectives. Moreover, the silvics of oak species suggest several morphological characteristics that are well adapted to fire. Nonetheless, prescribed fire still has the potential to do damage to the forest and the trees if misapplied or used at the wrong stage of stand development. The extent of damage that develops in forests after fire depends on the silvicultural system and management goals. It is important to time fire use and control its severity by managing fire intensity and applying it judiciously when it is appropriate given stand structure, composition, and desired developmental trajectory. It is imperative to know what the positive and negative consequences are when using fire to sustain oak ecosystems. Fire can provide many ecological benefits; it can also cause much damage and value loss. Wise decisions on fire use derive from knowledge of fire effects on the array of biological, ecological, economic, environmental and social values, goods and services that come from oak ecosystems.

LITERATURE CITED

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(5): 257-266.

Blankenship, B.A.; Arthur, M.A. 2006. Stand structure over nine years in burned and fire-excluded oak stands on the Cumberland Plateau, Kentucky. Forest Ecology and Management. 225: 134-145.

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. 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.; 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.; 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. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 123-127.

Burns, P.Y. 1955. Fire scars and decay in Missouri oaks. Bull. 642. Columbia, MO: University of Missouri, Agriculture Experiment Station, College of Agriculture. 8 p.

Chiang, J.; Arthur, M.A.; Blankenship, B.A. 2005. The effect of prescribed fire on gap fraction in an oak forest on the Cumberland Plateau. Journal of the Torrey Botanical Society. 132: 432- 441.

Dey, D.C. 1991. A comprehensive Ozark regenerator. Columbia, MO: University of Missouri. 238 p. Ph.D. dissertation.

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. Gen. Tech. Rep. NRS-P-46. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 2-20.

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.; Hartman, G. 2005. Returning fire to Ozark Highland forest ecosystems: effects on advance regeneration. Forest Ecology and Management. 217: 37-53.

Dey, D.C.; Jacobs, D.; McNabb, K. [and others]. 2008a. 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.; Jensen, R.G.; Wallendorf, M.J. 2008b. 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. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 26-37.

Dey, D.C.; Johnson, P.S.; Garrett, H.E. 1996. Modeling the regeneration of oak stands in the Missouri Ozark Highlands. Canadian Journal of Forest Research. 26: 573-583.

Dey, D.C.; Parker, W.C. 1997. Morphological indicators of stock quality and field performance of red oak (*Quercus rubra* L.) seedlings underplanted in a central Ontario shelterwood. New Forests. 14: 145-156.

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.

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.

Forest Products Laboratory. 1967. Comparative decay resistance of heartwood of native species. Res. Note FPL-0153. Madison, WI: U.S. Department of Agriculture Forest Service, Forest Products Laboratory. 2 p.

Green, S.R.; Arthur, M.A.; Blankenship, B.A. 2010. Oak and red maple seedling survival and growth following periodic prescribed fire on xeric ridgetops on the Cumberland Plateau. Forest Ecology and Management. 259: 2256-2266.

Gustafson, R.O. 1944. Cull as determined from basal wounds in Kentucky highlands timber. Journal of Forestry. 42(3): 181-184.

Guyette, R.; Marschall, J.; Stambaugh, M.; Stevenson, A. 2012b. Validating time-dependent tree defect models and determining lumber yield changes for prescribed fire injured oak trees. Final report cooperative agreement 327. Jefferson City, MO: Missouri Department of Conservation. 45 p.

Guyette, R.P.; Muzika, R.M.; Dey, D.C. 2002. Dynamics of an anthropogenic fire regime. Ecosystems. 5: 472-486.

Guyette, R.P.; Stambaugh, M.C. 2004. Post-oak fire scars as a function of diameter, growth, and tree age. Forest Ecology and Management. 198: 183-192.

Guyette, R.P.; Stambaugh, M.C.; Dey, D.C.; Muzika, R.M. 2012a. Predicting fire frequency with chemistry and climate. Ecosystems. 15(2): 322-335.

Hare, R.C. 1965.The contribution of bark to fire resistance of southern trees. Journal of Forestry. 63: 248-251.

Harmon, M.E. 1984. Survival of trees after low-intensity surface fires in the Great Smokey Mountains National Park. Ecology. 65: 796-802.

Hengst, G.E.; Dawson, J.O. 1994. Bark properties and fire resistance of selected tree species from the central hardwood region of North America. Canadian Journal of Forest Research. 24: 688-696.

Hepting, G.H. 1935. Decay following fire in young Mississippi Delta hardwoods. Tech. Bull. 494. Washington, DC: U.S. Department of Agriculture. 32 p.

Hepting, G.H. 1937. Decay in merchantable oak, yellow poplar, and basswood in the Appalachian region. Tech. Bull. 570. Washington, DC: U.S. Department of Agriculture. 30 p.

Hepting, G.H. 1941. Prediction of cull following fire in Appalachian oaks. Journal of Agricultural Research. 62(2): 109-120.

Hesterberg, G.A. 1957. Deterioration of sugar maple following logging damage. Sta. Pap. 51. St. Paul, MN: U.S. Department of Agriculture Forest Service, Lake States Forest Experiment Station. 58 p.

Hutchinson, T.F.; Sutherland, E.K.; Yaussy, D.A. 2005. Effects of repeated prescribed fires on the structure, composition, and regeneration of mixed-oak forests in Ohio. Forest Ecology and Management. 218: 210-228.

Iverson, L.R.; Hutchinson, T.F. 2002. Soil temperature and moisture fluctuations during and after prescribed fire in mixed-oak forests, USA. Natural Areas Journal. 22: 296-304.

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 US: 7-year results. Forest Ecology and Management. 255: 3035-3050.

Iverson, L.R.; Yaussy, D.A.; Rebbeck, J. [and others]. 2004. A comparison of thermocouples and temperature paints to monitor spatial and temporal characteristics of landscape-scale prescribed fires. International Journal of Wildland Fire. 13: 311-322.

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. 1974. Survival and growth of northern red oak seedlings following a prescribed burn. Res. 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. 2^{d} ed. New York: CABI Publishing. 580 p.

Kaufert, F.H. 1933. Fire and decay injury in the southern bottomland hardwoods. Journal of Forestry. 31(1): 64-67.

Keefe, J.F. 1987. The first 50 years. Jefferson City, MO: Missouri Department of Conservation. 446 p.

Knapp, B.O.; Wang, G.G.; Van Lear, D.L.; Walker, J.L. 2006. Predicting root biomass of burned and unburned white oak advance reproduction from diameter and height. Southern Journal of Applied Forestry. 30: 40-45.

Kruger, E.L.; Reich, P.B. 1997. Responses of hardwood regeneration to fire in mesic forest openings. I. Post-fire community dynamics. Canadian Journal of Forest Research. 27: 1822-1831.

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.

Loewenstein, E.F. 2005. Conversion of uniform broadleaved stands to an uneven-aged structure. Forest Ecology and Management. 215: 103-112.

Loomis, R.M. 1974. Predicting the losses in sawtimber volume and quality from fires in oak-hickory forests. Res. Pap. NC-104. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 6 p.

Loomis, R.M. 1989. Appraising fire effects. Note 8.03. In: Clark, F.B., tech. ed.; Hutchinson, J.G., ed. Central hardwood notes. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station: 8 p.

Miles, P.D. 2013. Forest Inventory EVALIDator web-application version 1.5.1.04. St. Paul, MN: U.S. Department of Agriculture Forest Service, Northern Research Station. http://apps.fs.fed.us/Evalidator/tmattribute.jsp. [Date accessed: March 14, 2013].

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.

MTNF. 2005. Mark Twain National Forest Plan. Rolla, MO: U.S. Department of Agriculture Forest Service, Mark Twain National Forest: 1-1 to 4-12 with indices.

Nelson, P.W. 2010. The terrestrial natural communities of Missouri. Missouri Natural Areas Committee. 550 p.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. Update edition. New York: John Wiley & Sons. 520 p.

Pyne, S.J. 1982. Fire in America: a cultural history of wildland and rural fire. Princeton, NJ: Princeton University Press. 654 p.

Pyne, S.J.; Andrews, P.L.; Laven, R.D. 1996. Introduction to wildland fire. 2^{d} ed. New York: John Wiley & Sons. 769 p.

Regelbrugge, J.C.; Smith, D.W. 1994. Postfire tree mortality in relation to wildfire severity in mixed oak forests in the Blue Ridge of Virginia. Northern Journal of Applied Forestry. 11: 90-97.

Shifley, S.R.; Smith, W.B. 1982. Diameter growth, survival, and volume estimates for Missouri trees. Res. Note NC-292. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 7 p.

Smith, K.T.; Sutherland, E.K. 1999. Fire-scar formation and compartmentalization in oak. Canadian Journal of Forest Research. 29: 166-171.

Smith, K.T.; Sutherland, E.K. 2006. Resistance of eastern hardwood stems to fire injury and damage. In: Dickinson, M.B., ed. Fire in eastern oak forests: delivering science to land managers. Proceedings of a conference. Gen. Tech. Rep. NRS-P-1. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 210-217.

Stambaugh, M.; Guyette, R. 2008. Prescribed fire effects on the wood quality of three common oaks in the Ozark region, *Q. coccinea, Q. velutina, Q. alba*. Jefferson City, MO: Missouri Department of Conservation. 23 p.

Stambaugh, M.C.; Guyette, R.P. 2004. Long-term growth and climate response of shortleaf pine at the Missouri Ozark forest ecosystem project. 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: 448-458.

Stambaugh, M.C.; Guyette, R.P.; Grabner, K.W.; Kolaks, J. 2006. Understanding Ozark forest litter variability through a synthesis of accumulation rates and fire events. In: Andrews, P.L.; Butler, B.W., comps. Fuels management - how to measure success: conference proceedings. Proceedings RMRS-P-41. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 321-332.

Steen, H.K. 1976. The U.S. Forest Service: a history. Seattle: University of Washington Press. 356 p.

Sutherland, E.K.; Smith, K.T. 2000. Resistance is not futile: the response of hardwoods to fire-caused wounding. In: Yaussy, D.A., comp. Proceedings of the workshop on fire, people, and the central hardwood landscape. Gen. Tech. Rep. NE-274. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station: 111-115.

Vines, R.G. 1968. Heat transfer through bark, and the resistance of trees to fire. Australian Journal of Botany. 16: 499-514.

Waldrop, T.A.; White, D.L.; Jones, S.M. 1992. Fire regimes for pine-grassland communities in the southeastern United States. Forest Ecology and Management. 47: 195-210.

Weigel, D.R.; Parker, G.R. 1997. Tree regeneration response to the group selection method in southern Indiana. Northern Journal of Applied Forestry. 14: 90-94.

Westin, S. 1992. Wildfire in Missouri. Jefferson City, MO: Missouri Department of Conservation. 161 p.

PRESCRIBED FIRE AND BRUSH REMOVAL AFFECT VEGETATION, FUEL LOADS, AND ABUNDANCE OF SELECTED BEETLE POPULATIONS IN PINE STANDS

James D. Haywood, Tessa A. Bauman, Richard A. Goyer, and Gerald J. Lenhard[1]

Abstract--Three forest sites were selected in Louisiana in early 2001. On each site, three treatments were applied: (1) Check: no further management; (2) PF: prescribed fire was applied in May 2001 and June 2003; and (3) PF-MPC: between the two prescribed fires, midstory and understory woody vegetation was masticated with mechanical equipment in July 2002. Management did not change overstory composition, and basal area per ha increased on all treatments through February 2005. Percentage of understory arborescent cover also increased, although less so on PF and PF-MPC treatments than Checks. Herbaceous plant cover decreased on all treatments through August 2004, and increased shading and crowding were the likely reasons. Both PF and PF-MPC reduced 10-hour time-lag dead fuels but did not affect 1-hour or 100-hour fuels. Abundance of pine bark beetles, *Dendroctonus terebrans*, *Hylastes salebrosus*, *H. tenuis*, and *Ips* spp., and root-feeding weevils, *Hylobius pales* and *Pachylobius picivorous*, increased after each prescribed fire but not after mastication.

INTRODUCTION

Before European settlement, longleaf pine (*Pinus palustris* Mill.) forests occupied as much as 38 million ha across the southeastern United States and were the most extensive ecosystem in North America (Brockway and others 2005, Landers and others 1995). Across this extent, longleaf pine depended upon natural and anthropogenic fires for existence, because without fire, other pines and hardwoods would eventually succeed them (Brockway and others 2005, Haywood and others 2001, Wahlenberg 1946). Once longleaf pine forests were depleted for their many desirable commercial attributes, they failed to naturally recover because the species does not successfully invade open lands; livestock, especially feral hogs (*Sus scrofa*), destroyed established seedlings, and the natural fire regime was disrupted (Barnett 1999, Komarek 1983, Landers and others 1995). Fire protection implemented during the 1920s allowed invasive hardwoods and other southern pines to replace longleaf pine within its natural range (Barnett 1999). By 1996, only 1.2 million ha of longleaf pine forest remained (Outcalt and Sheffield 1996).

Currently, several state and federal agencies, non-governmental organizations, and private individuals are restoring longleaf pine to portions of its former range (Brockway and others 2005). The desired future condition is park-like longleaf pine grasslands with few midstory hardwoods except in riparian and other moist areas, and a rich and diverse ground cover of herbaceous and low woody plants that can be maintained by frequent surface fires; thus, prescribed fire is an essential element in these efforts (Barnett 1999, Brockway and others 2005, Haywood and others 2001, Landers and others 1995).

Forest managers, however, are faced with multiple challenges in planning and executing prescribed fires to minimize injury to the desired trees while removing unwanted vegetation. While fire may promote longleaf regeneration and facilitate restoration efforts, it may also weaken trees, making them susceptible to attacks by insects and pathogens (Hanula and others 2002, Sullivan and others 2003). Of prime concern is the increased attraction caused by resin flow and the reduced resistance to bark beetles such as the southern pine beetle (*Dendroctonus frontalis* Zimmermann), black turpentine beetle (*D. terebrans* Olivier), and *Ips* species. Root-feeding weevils are also attracted to fire-stressed stands (Bauman 2002, Hanula and others 2002, Sullivan and others 2003). Longleaf pine trees tolerate fire and have high resin flow rates that make them more resistant to bark beetle attack than other southern pines. Still, the effects of fire on the interaction of insect pests and longleaf pine health are largely unknown. This lack of knowledge is a major

[1]Supervisory Research Forester, USDA Forest Service, Southern Research Station, Pineville, LA 71360; Graduate Research Assistant, Louisiana State University Agricultural Center, Department of Entomology, Baton Rouge, LA 70803; Professor (retired), Louisiana State University Agricultural Center, School of Renewable Natural Resources, Baton Rouge, LA 70803; and Research Assistant (retired), Louisiana State University Agricultural Center, Department of Entomology, Baton Rouge, LA 70803.

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concern given the high fuel loads and mixed composition of many upland stands currently undergoing fire-aided restoration efforts.

Because prescribed burning is challenging, mechanical or chemical methods have been proposed as supplemental means for reducing woody vegetation (Brockway and others 2005, Drewa and others 2002, Haywood and Harris 1999, Haywood and others 2001, Provencher and others 2001). In fact, woody plant control may be required where managers must restore certain plant communities quickly (Brockway and others 2005, Haywood 2000). To study the relationship between fire, fire surrogates, and insects, two management schemes involving fire and mechanical treatment were evaluated in three upland pine forests in Louisiana to determine if management influenced fuel conditions, vegetation, and selected beetle populations.

METHODS

Study Sites

The three study sites lie within the boundaries of the Kisatchie National Forest in central Louisiana at an average elevation of 45 m above sea level. All three sites are suitable for restoration of loamy dry-mesic upland longleaf pine forests (Turner and others 1999), which is the long-term management objective.

Study Site 1 is a mixed loblolly pine (*P. taeda* L.) and longleaf pine stand on a well-drained and moderately permeable Ruston fine sandy loam (fine-loamy, siliceous, semiactive, thermic, Typic Paleudults) (Kerr and others 1980). Site 2 is a loblolly pine stand on a Ruston fine sandy loam with Malbis fine sandy loam (fine-loamy, siliceous, subactive, thermic, Plinthic Paleudults) interspersed. Site 3 is a mixed loblolly and longleaf pine stand on Smithdale fine sandy loam (fine-loamy, siliceous, subactive, thermic, Typic Hapludults) and Kolin silt loam (fine-silty, siliceous, active, thermic, Glossaquic Paleudalfs). The Ruston and Smithdale soils form well-drained, moderately permeable uplands. The Malbis is moderately well drained and moderately to slowly permeable, and the Kolin is moderately well drained and slowly permeable. The Malbis, Ruston, and Smithdale soils have few limitations for growing pine trees, while soil drainage may limit growth on the Kolin soil. The site index (base age 50) at Sites 1 and 2 is 24 m for longleaf pine and 21 m at Site 3 (Kerr and others 1980).

Study Establishment and Measurements

Each site or block is about 10 ha and was divided into three treatment areas: (1) Check: management was discontinued; (2) PF: prescribed fire was applied to the plots in May 2001 and June 2003; and (3) PF-MPC: between prescribed fires, the understory and midstory vegetation was masticated to within 5 cm of the ground with a machine-mounted horizontal-shaft drum shredder (Woodgator®) as a mechanical plant control (MPC) treatment in July 2002.

Prescribed burning was by strip headfiring the treatment areas after baseline backfires were established. Rates of spread were estimated as best as possible from outside the burned area. Overall, the prescribed fires in 2001 averaged 342 kJ/s/m (table 1), which was higher than the 173 kJ/s/m intensity recommended by Deeming and others (1977) as a maximum for prescribed fires. However, the 2001 intensities were comparable to fire intensities reported for prescribed burning in longleaf pine-grassland fuels (Haywood 2002). Fire intensities were lower in 2003 than in 2001 but still above the 173 kJ/s/m intensity recommended by Deeming and others (1977). In addition, fire intensity between the PF and PF-MPC treatments were comparable in both years.

Table 1—Fire intensities by treatment and time of burning

Treatments	May 2001	June 2003
	------------*kJ/s/m*------------	
PF[a]	345	258
PF-MPC[a]	338	192

[a]PF = prescribed fire; MPC = midstory and understory woody vegetation, masticated in July 2002

For the MPC treatment, it was argued that mastication of the woody plants had to precede the second prescribed fire by about a year. The woody debris would have enough time to deteriorate so that the prescribed fire in June 2003 would not be too severe and injure the overstory pine trees. However, the next prescribed fire could not be delayed longer than a year after treatment or the understory brush would regain its pre-masticated stature.
Within each treatment area, a 0.10-ha plot was established for making fuel and vegetation measurements. We inventoried and took the diameter at breast height (d.b.h.) of overstory trees (stems > 10 cm d.b.h.) in April 2001 and

total height and d.b.h. of the overstory trees in February 2005. The 2005 measurements were used to calculate a volume per tree with Baldwin and Feduccia's (1987) formula for loblolly pine, Baldwin and Saucier's (1983) formula for longleaf pine, and Clark and others' (1985) formula for mixed hardwoods.

Four 10-m^2 subplots were randomly selected and established within each main plot. In August 2004, all understory trees and shrubs > 1.4-m tall with a d.b.h. < 2.5 cm were inventoried on the 10-m^2 subplots, and d.b.h. was measured. Initially, we attempted to study midstory vegetation changes using the 10-m^2 subplots; however, the midstory trees were too scattered and the mastication treatment eliminated the midstory on the PC-MPC plots, nullifying this plan.

A 1-m^2 subplot was randomly nested within each 10-m^2 subplot; therein, cover of nine understory vegetation classes were estimated to the nearest percent. The nine classes were trees, shrubs, blackberry (*Rubus* spp.), ferns, forbs, grasses, grass-likes, legumes, and woody vines. Cover estimates were made in July 2001, June 2003, July 2003, and August 2004.

Fuel load was measured on four randomly established 2- by 5-m fuel-sample plots within each 0.10-ha main plot. The fuel-sample plots were distinct from the subplots used to inventory vegetation. Each fuel-sample plot was divided into 10 1-m^2 subplots. On each sample date, a 1-m^2 subplot was randomly selected for sampling fuel load without replacement. Fuel samples were collected before and 6 weeks after each prescribed fire on all three treatments. The sampled fuels were separated into five fuel classes considered available for burning based on Deeming and others (1977) fire-danger-rating system. The five fuel classes were as follows: (1) living foliage of all trees, shrubs, vines, grasses, and forbs within 2 m of the ground; (2) living blackberry canes, woody stems, and vines no more than 6 mm in diameter within 2 m of the ground; (3) 1-hour time-lag dead fuels (surface litter and duff to a 0 to 6 mm depth and small roundwood and stubble no more than 6 mm in diameter); (4) 10-hour time-lag dead fuels (litter from a 7 to 25 mm depth and roundwood and stubble between 7 and 25 mm in diameter); and (5) 100-hour time-lag dead fuels (litter from a 26 mm to 100 mm depth and roundwood between 26 and 75 mm in diameter). In addition, in the post-burn samples, two additional fuel classes were collected--(6) needlecast after the prescribed fire; and (7) regrowth of vegetation--to keep these classes from biasing how much fuel was consumed in the fires. The sampled fuels were oven dried at 80 °C for 72 hours in a forced-air oven and weighed. The differences in pre-burn and post-burn fuel samples (not including needlecast or regrowth immediately after the prescribed fire) were used with a best estimate of rate of spread to calculate Byram's fire intensity as described by Haywood (1995).

Insects were collected over the entire treatment area and not just in the 0.1-ha main plot. Flight interception and pitfall traps were used to sample insects after Klepzig and others (1991). Flight interception traps were placed at 0.5 m above ground, and pitfall traps were placed in the ground with openings drilled at intervals around the circumference to allow crawling insects to enter. The traps were baited with ethyl alcohol (95 percent) and Klean Strip® turpentine, general attractants for pine-infesting insects (Chénier and Philogène 1989, Hunt and Raffa 1989, Phillips and others 1988, Schroeder and Lindelöw 1989), and placed in pairs of two-dram vials that were secured to the interior of the traps. The baited vials were refilled at each collection period. A 5- to 8-cm long and 1- to 2-cm diameter segment of loblolly pine stem was placed in the bottom of the pitfall trap as a substrate for insect attack/breeding. Ten pitfall and ten flight interception traps were deployed yearly in the check and PF-MPC treatments and on the PF treatment in 2001 and 2003. The traps were spaced 30 to 50 m apart and near pine trees. Initially, insects of interest included pine bark beetles, ambrosia beetles, and root-feeding weevils, which were collected, identified to species (Wood 1982), and counted weekly. Voucher specimens were stored with the USDA Forest Service, Forest Insect Research Unit at Pineville, LA. Trapping was conducted 3 to 4 weeks pre-treatment and 6 to 8 weeks post-treatment for the prescribed fires in 2001 and 2003 and for the mastication treatment in 2002.

It was ascertained from related, concurrent studies at nearby sites that six beetle taxa regularly occurred in longleaf pine stands after prescribed burning (Bauman 2002). These were pine bark beetles, *Dendroctonus terebrans* Olivier, *Hylastes salebrosus* Eichoff, *H. tenuis* Eichoff, and *Ips* spp. (Coleoptera: Scolytidae) and the root-feeding weevils, *Hylobius pales*

Herbst and *Pachylobius picivorous* Germar (Coleoptera: Curculionidae). These beetles do not cause major damage in healthy longleaf pine stands and mostly reproduce in the lower boles and roots of damaged or stressed pine trees. However, they are attracted to burned stands, which benefit these beetles by permitting location and exploitation of host trees made susceptible by fire injury. Data from other insects collected, mostly ambrosia beetles (Coleoptera: Scolytidae and Platypodidae), were not included in the analyses because of their unknown association with longleaf pine and their irregular distribution.

Data Analysis

The statistician reviewing this paper suspected that there were site-by-treatment interactions, which nullified the block effect and would leave no degrees of freedom for Error Mean Squares in the analyses. Therefore, we do not present statistical results, although we will discuss trends in the data based on prior, albeit inappropriate, analyses. In those earlier analyses, the overstory stocking and basal area measurements in April 2001 and February 2005 were compared as was volume per ha in 2005. Number of stems and basal area per ha and average d.b.h. of understory trees and shrubs > 1.4-m tall in August 2004 were compared. Percent cover of understory vegetation were compared on the following dates, post burn in June 2001, preburn in June 2003, post burn in July 2003, and post burn in August 2004. Besides percent cover of all understory vegetation, we compared the cover of arborescent vegetation (trees, shrubs, and blackberry) and the other taxa (ferns, forbs, grasses, grass-likes, legumes, and woody vines).

The ovendried mass of live stems and foliage both followed the same pattern of treatment response throughout the study. Therefore, the two classes of live fuels were combined. Fuel loads (kg/ha) in May and June 2001, June and July 2003, and August 2004 were compared for live fuels, 1-hour time-lag dead fuels, 10-hour time-lag dead fuels, and 100-hour time-lag dead fuels. The beetle counts were combined for all taxa. Means were based on pooled weekly trap catches over the 8-week sampling period from 10 sampling points per study site and treatment. For 2001 and 2003, beetle counts on all three treatments were compared. For all 3 years, beetle counts on the check and PF-MPC treatments were likewise compared.

RESULTS

Overstory Vegetation

The number of overstory trees per ha did not change much from April 2001 to February 2005, although the average number of trees decreased from 205 to 185 trees per ha over this period (table 2). Average basal area per ha greatly increased during this period from 19.8 to 20.9 m^2/ha across all treatments. In 2005, total volume for all species averaged 267 m^3/ha across all three treatments but was similar among treatments.

Loblolly and longleaf pine trees dominated the overstory, comprising 81 percent of the trees, 94 percent of the basal area, and 95 percent of the total volume per ha in February 2005. Loblolly pine was the more common pine species comprising 87 percent of the pine trees, 88 percent of the pine basal area, and 89 percent of the total pine volume per ha. The minor overstory species were flowering dogwood (*Cornus florida* L.), sweetgum (*Liquidambar styraciflua* L.), blackgum (*Nyssa sylvatica* Marsh.), black cherry (*Prunus serotina* Ehrh.), southern red oak (*Quercus falcata* Michx.), blackjack oak (*Q. marilandica* Muenchh.), and post oak (*Q. stellata* Wangenh.).

Understory Vegetation

The number of understory trees and shrubs ≥ 1.4-m tall with a d.b.h. < 2.5 cm ranged from

Table 2—Number of overstory trees, basal area, and volume per hectare

	----------April 2001----------		----------------February 2005---------------		
Treatments	Stocking	Basal area	Stocking	Basal area	Volume
	trees/ha	*m^2/ha*	*trees/ha*	*m^2/ha*	*m^3/ha*
Check	203	20.0	200	21.1	269.0
PF[a]	183	19.8	173	21.4	274.1
PF-MPC[a]	230	19.6	183	20.2	256.4

[a]PF= prescribed fire; MPC = midstory and understory woody vegetation, masticated in July 2002.

2,833 to 14,500 stems/ha on the PF-MPC and check plots, respectively, 40 months after the first prescribed fire (table 3). D.b.h. and basal area per ha of the understory plants were likewise affected by treatment with checks having the greatest d.b.h. and basal area and PF-MPC having the least d.b.h. and basal area per ha. Prescribed burning also affected stocking and plant stature but not as much as the PF-MPC treatment.

After the first prescribed fire, percent cover of all of the understory vegetation was not much affected by burning, although cover on the PF and PF-MPC treatments averaged 54 percent and was 87 percent on the checks (fig. 1, top graph). However, after the mastication treatment and before the second prescribed fire, vegetation cover varied among treatments: 111, 91, and 63 percent on the check, PF, and PF-MPC treatments, respectively, because mastication of the woody material and vehicle trafficking destroyed a part of the understory plant community on the PF-MPC plots. Following the second prescribed fire, both of the PF and PF-MPC treatments had less understory cover than the checks. In August 2004, the understory vegetation on the PF and PF-MPC treatments had not fully recovered, and there were still treatment differences at the last measurement.

The mastication treatment kept understory arborescent vegetation suppressed until the second prescribed fire (fig. 1, middle graph), in which the second fire reduced woody cover on both the PF and PF-MPC treatments to a similar percentage. The overall pattern of responses for understory arborescent vegetation was similar to the pattern for all understory vegetation, because woody plants dominated in the understory.

Table 3--Number of stems, d.b.h., and basal area per hectare of understory trees and shrubs > 1.4 m tall with a d.b.h. < 2.5 cm in August 2004, which was 40 months after the first prescribed fire

Treatments	Stocking	D.b.h.	Basal area
	stems/ha	*mm/stem*	*m^2/ha*
Check	14,500	11	1.58
PF[a]	5,417	6	0.21
PF-MPC[a]	2,833	2	0.03

[a]PF = prescribed fire; MPC = midstory and understory woody vegetation, masticated in July 2002.

Average percent cover of ferns, forbs, grasses, grass-likes, legumes, and woody vines was not affected by treatment until after the second prescribed fire, when there was less cover on the PF-MPC plots (11 percent) than on check and PF plots (an average of 24 percent) (fig. 1, bottom graph). The mastication treatment doubtlessly changed the structure of the fuel bed although it did not increase the amounts of 1-hour or 10-hour time-lag dead fuels (fig. 2). The difference in fuel bed structure may have influenced how the fire spread, and therefore, there was an immediate, adverse effect on the cover of ferns, forbs, grasses, grass-likes, legumes, and woody vines after the second prescribed fire (fig. 1). By August 2004, however, the percent cover among treatments was less on the checks (15 percent) than on the PF and PF-MPC treatments (22 percent).

Fuel Loads

Before the first prescribed fire, the mass of live stems and foliage averaged 1486 kg/ha across

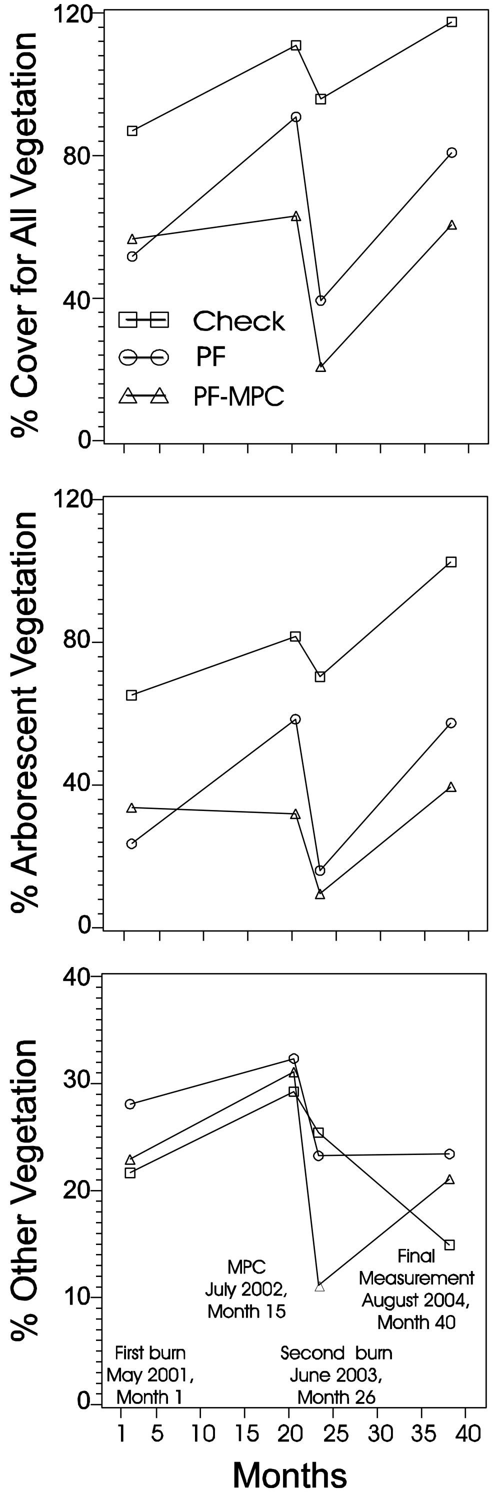

Figure 1—Percent cover of all understory vegetation (top); arborescent vegetation only (middle); and ferns, forbs, grasses, grass-likes, legumes, and woody vines (bottom) that were measured post burn in June 2001, preburn in June 2003, post burn in July 2003, and post burn in August 2004: PF-prescribed fire was applied in May 2001 and June 2003, and MPC-midstory and understory woody vegetation was masticated in July 2002.

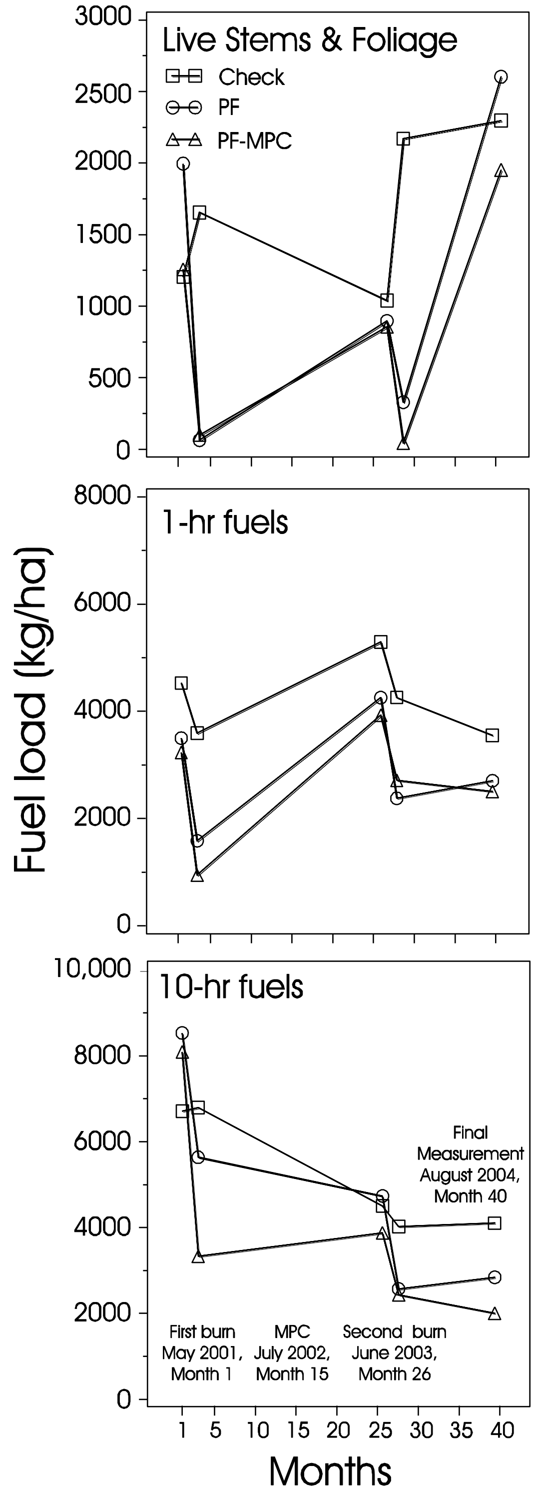

Figure 2—Fuel loads (kg/ha) for the live stems and foliage (top), 1-hr time-lag dead fuels (middle), and 10-hr time-lag dead fuels (bottom) that were measured preburn in May 2001, post burn in June 2001, preburn in June 2003, post burn in July 2003, and post burn in August 2004: PF-prescribed fire was applied in May 2001 and June 2003, and MPC-midstory and understory woody vegetation was masticated in July 2002.

all three treatments (fig. 2, top graph). The first prescribed fire reduced the amount of live stems and foliage, while they continued developing on the checks. Before the second prescribed fire, the mass of stems and foliage was again similar on all three treatments averaging 664 kg/ha, and the post-burn pattern of response was the same as after the first prescribed fire. On the checks, the mass of live stems and foliage ranged from 1204 kg/ha in May 2001 to 2298 kg/ha in August 2004 (fig. 2). The mass of stems and foliage on the PF and PF-MPC treatments averaged 2278 kg/ha in August 2004.

The preburn mass of 1-hour time-lag dead fuels averaged 3726 kg/ha in May 2001 and 4482 kg/ha in June 2003 across all three treatments (fig. 2, middle graph). Prescribed burning reduced the mass of 1-hour fuels, but there was also a decrease in 1-hour fuels on the checks in both years. In August 2004, 1-hour fuels averaged 2876 kg/ha across all three treatments.

In May 2001, the PF and PF-MPC treatments had more preburn mass of 10-hour time-lag dead fuels (an average of 8404 kg/ha) than the checks (6776 kg/ha) (fig. 2, bottom graph). However, prescribed burning reduced the amount of these fuels on the PF and PF-MPC plots. In June 2003, the preburn mass of 10-hour fuels was similar on all three treatments and averaged 4396 kg/ha. Prescribed burning again reduced the mass of 10-hour fuels, while the change in mass of 10-hour fuels on the checks was minor. In August 2004, there was more mass of 10-hour fuels on the checks (4121 kg/ha) than on the PF and PF-MPC treatments (an average of 2411 kg/ha). The mass of 100-hour time-lag fuels was unaffected by time or treatment, and the 100-hour fuels averaged 1983 kg/ha in May 2001 and 1672 kg/ha in August 2004.

Beetle Abundance

Both prescribed fires resulted in significant increases in abundance of beetles (fig. 3, top graph). The overall number of beetles appeared to be greater in 2001 than in 2003. The mastication treatment in 2002 on the PF-MPC plots did not directly influence beetle abundance perhaps because only minor damage to the overstory pine trees occurred, and therefore no chemical attractants were released from the trees to attract beetles (fig. 3, bottom graph).

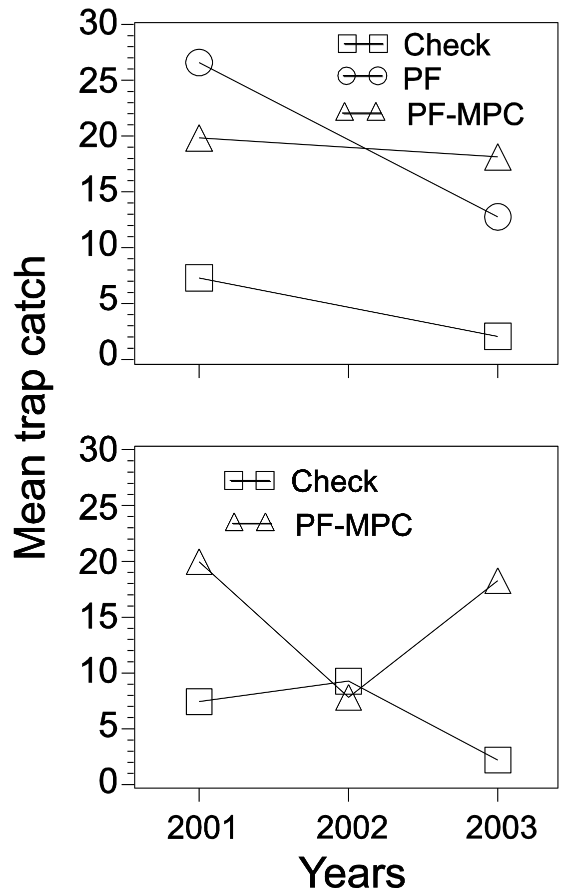

Figure 3—Abundance of six beetle taxa among all three treatments in 2001 and 2003 (top) and between the check and PF-MPC treatments in 2001, 2002, and 2003 (bottom): PF-prescribed fire was applied in May 2001 and June 2003, and MPC-midstory and understory woody vegetation was masticated in July 2002.

DISCUSSION

Prescribed burning, with or without the mastication treatment, did not change overstory composition; rather, basal area increased on all three treatments. The only pine tree mortality noted was associated with lightning strikes.

However, increasing fire intensities have been correlated with greater mortality within older longleaf pine stands where pine litter and duff are the primary available fuels (Sullivan and others 2003). For example, conditions favorable for fires that move too slowly at the base of trees where sloughed bark and needles accumulate can cause basal scarring, cambial death, root damage, and tree mortality (Wade and Johansen 1986). Presumably, because the stands in this study have a history of prescribed fire, litter and duff did not accumulate in large amounts, and longleaf pine mortality did not result.

There are no comparable studies to utilize in making valid comparisons of the number of beetles in our study responding to these treatments. However, Bauman (2002) and Sullivan and others (2003) found that prescribed burning resulted in peak trap captures 2 to 3 weeks after burning, which was similar to our results. We found consistently higher captures of bark beetles and root-feeding weevils after prescribed burning although the insect collections were conducted during the growing season and after the peak activity for *Hylastes* beetles (Bauman 2002). Further, their and our data indicate that the local population increases were the result of immigration (attractiveness) and not to increased breeding.
Fire intensity is positively correlated with post-fire insect abundance (Sullivan and others 2003), and the insects of interest are attracted to fire-injured pine trees where they may be able to reproduce in the lower boles and roots. Fire intensities were above the maximum intensity for a “safe” winter backfire (Deeming and others 1977) but about half the intensity expected when burning heavy grass rough (Haywood 2002) (table 1). *Hylastes* beetles and root-feeding weevils might be vectors for stain fungi (Eckhardt and others 2004, Hanula and others 2002, Klepzig and others 1991), and increasing degrees of crown thinning and foliage discoloration after prescribed burning have been associated with greater prevalence of stain fungi in longleaf pine (Otrosina and others 1999) and red pine (*P. resinosa* Soland.) (Klepzig and others 1991). A decline in tree vigor was not seen on our study sites, possibly because the pine trees were not seriously injured by the fires and were healthy enough to withstand insect attack. Regardless, insect abundance increased in recently burned stands, and this is of particular concern to land managers when and where the southern pine beetle is active.

Prescribed burning is often used as a way to open forest stands, remove litter, and create conditions where herbaceous productivity can increase (Grelen and Epps 1967, Haywood and others 2001). However, in this study, the percent cover of non-arborescent understory vegetation on the checks decreased from 22 percent to 15 percent from 2001 to 2004 and decreased from 26 percent to 22 percent over the same period on the PF and PF-MPC treatments (fig. 1). Both understory arborescent vegetation and overstory basal area adversely affect herbaceous plant productivity (Haywood and Harris 1999, Wolters 1982). Understory arborescent cover increased by 17 percentage points on the PF and PF-MPC treatments and 31 percentage points on the checks from 2001 to 2004. Therefore, the increased shading from a developing overstory and understory reduced herbaceous plant cover regardless of treatment. A possible solution would be to apply herbicides to reduce woody plant cover to levels that can be maintained by prescribed fire (Brockway and others 2005, Drewa and others 2002, Provencher and others 2001), and possibly favor herbaceous plant development (Grelen and Epps 1967, Haywood and others 2001).

The mass of live fuels increased by 91 percent on the checks from May 2001 to August 2004 but only increased by 40 percent on the PF and PF-MPC treatments (fig. 2). This trend was similar to the one for percent cover of understory arborescent vegetation (fig. 1). Fire initially affected live available fuels, but prescribed burning has not arrested plant growth.

None of the dead fuels accumulated on any of the treatments (fig. 2). Mass of 1-hour fuels decreased by 23 percent from 2001 to 2004 across all three treatments; the 10-hour fuels decreased by 71 percent on the PF and PF-MPC treatments and by 39 percent on the checks; and the 100-hour time-lag fuels decreased by 16 percent across all three treatments. On the checks, the decrease in fuel loads was either due to decomposition or sampling error, although the same personnel collected the fuel samples throughout the study. Rather, since all three available dead-fuel classes are decreasing on all three treatments, the decrease in surface fuels may be truly occurring, but whether this is a short-term or long-term trend is not certain. Nevertheless, the two prescribed fires initially reduced available fuel load by 49 percent on average (fig. 2), and this reduction temporarily lowered the risk of catastrophic wildfire.

The mastication treatment cost $337/ha U.S. while prescribed burning was $30/ha U.S. in 2002 (Personal communication. 2002. Daniel McDonald, Fire Management Officer, Calcasieu Ranger District, Kisatchie National Forest, 9912 Highway 28 West, Boyce, LA 71409). Given the cost differential and the limited effect of mastication, its extensive use seems unwarranted except in specific circumstances such as where midstory removal is needed beneath red cockaded woodpecker (*Picoides borealis*) colonies or at the wildland-urban interface.

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LITERATURE CITED

Baldwin, V.C., Jr.; Feduccia, D.P. 1987. Loblolly pine growth and yield prediction for managed West Gulf plantations. Res. Pap. SO-236. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 27 p.

Baldwin, V.C., Jr.; Saucier, J.R. 1983. Aboveground weight and volume of unthinned, planted longleaf pine on West Gulf forest sites. Res. Pap. SO-191. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 25 p.

Barnett, J.P. 1999. Longleaf pine ecosystem restoration: the role of fire. Journal of Sustainable Forestry. 9(1/2): 89-96.

Bauman, T.A. 2002. Interaction of fire and insects in the restoration and management of longleaf pine. Baton Rouge, LA: Louisiana State University. 72 p. M.S. thesis.
Brockway, D.G.; Outcalt, K.W.; Tomczak, D.J.; Johnson, E.E. 2005. Restoring longleaf pine forest ecosystems in the southern U.S. In: Stanturf, J.A.; Madsen, P., eds. Restoration of boreal and temperate forests. Boca Raton, FL: CRC Press: 501-519.

Chénier, J.V.R.; Philogène, B.J.R. 1989. Field responses of certain forest Coleoptera to conifer monoterpenes and ethanol. Journal of Chemical Ecology. 15: 1729-1746.

Clark, A., III; Phillips, D.R.; Frederick, D.J. 1985. Weight, volume, and physical properties of major hardwood species in the Gulf and Atlantic coastal plains. Res. Pap. SE-250. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 66 p.

Deeming, J.E.; Burgan, R.E.; Cohen, J.D.1977. The national fire-danger rating system - 1978. Gen. Tech. Rep. INT-39. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 63 p.

Drewa, P.B.; Platt, W.J.; Moser, E.B. 2002. Fire effects on resprouting of shrubs in headwaters of southeastern longleaf pine savannas. Ecology. 83: 755-767.

Eckhardt, L.G.; Goyer, R.A..; Klepzig, K.D.; Jones, J.P. 2004. Interactions of *Hylastes* species (Coleoptera: Scolytidae) with *Leptographium* species associated with loblolly pine decline. Journal of Economic Entomology. 97(2): 468-474.

Grelen, H.E.; Epps, E.A., Jr. 1967. Herbage responses to fire and litter removal on southern bluestem range. Journal of Range Management. 20(6) :403-404.

Hanula, J.L.; Meeker, J.R.; Miller, E.R.; Barnard, E.L. 2002. Association of wildfire with tree health and numbers of pine bark beetles, reproduction weevils and their associates in Florida. Forest Ecology and Management. 170: 233-247.

Haywood, J.D. 1995. Prescribed burning and hexazinone herbicide as release treatments in a sapling hardwood-loblolly pine stand. New Forests. 10(1): 39-53.

Haywood, J.D. 2000. Mulch and hexazinone herbicide shorten the time longleaf pine seedlings are in the grass stage and increase height growth. New Forests. 19(3): 279-290.

Haywood, J.D. 2002. Delayed prescribed burning in a seedling and sapling longleaf pine plantation in Louisiana. 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: 103-108.

Haywood, J.D.; Harris, F.L. 1999. Description of vegetation in several periodically burned longleaf pine forests on the Kisatchie National Forest. In: Haywood, J.D., ed. Proceedings of the 10th biennial southern silvicultural research conference. Gen Tech. Rep. SRS-30. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 217-222

Haywood, J.D.; Harris, F.L.; Grelen, H.E.; Pearson, H.A. 2001. Vegetative response to 37 years of seasonal burning on a Louisiana longleaf pine site. Southern Journal of Applied Forestry. 25(3): 122-130.

Hunt, W.A.; Raffa, K.F. 1989. Attraction of *Hylobius radicis* and *Pachylobius picivorus* (Coleoptera: Curculionidae) to ethanol and turpentine in pitfall traps. Environmental Entomology. 18(3): 351-355.

Kerr, A., Jr.; Griffis, B.J.; Powell, J.W. [and others]. 1980. Soil survey of Rapides Parish, Louisiana. Washington, DC: U.S. Department of Agriculture Soil Conservation Service and Forest Service, and Louisiana State University, Cooperative Extension Service. 87 p + 111 maps.

Klepzig, K.D.; Raffa, K.F.; Smalley, E.B. 1991. Association of an insect-fungal complex with red pine decline in Wisconsin. Forest Science. 37(4): 1119-1139.

Komarek, E.V. 1983. Fire as an anthropogenic factor in vegetation ecology. In: Holzner, W.; Werger, M.J.; Ikusima, I., eds. Man's impact on vegetation. Geobotany 5 [series]. New York: Springer Berlin Heidelberg: 77-82.

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(11): 39-44.

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.; Sheffield, R.M. 1996. The longleaf pine forest: trends and current conditions. Res. Bull. SRS-9. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 23 p.

Phillips, T.W.; Wilkening, A.J.; Atkinson, T.H. [and others]. 1988. Synergism of turpentine and ethanol as attractants for certain pine-infesting beetles (Coleoptera). Environmental Entomology. 17: 456-462.

Provencher, L.; Herring, B.J.; Gordon, D.R. [and others]. 2001. Longleaf pine and oak responses to hardwood reduction techniques in fire-suppressed sandhills in northwest Florida. Forest Ecology and Management. 148: 63-77.

Schroeder, L.M.; Lindelöw, A. 1989. Attraction of scolytids and associated beetles by different absolute amounts and proportions of à-pinene and ethanol. Journal of Chemical Ecology. 15: 807-818.

Sullivan, B.T.; Fettig, C.J.; Otrosina, W.J. [and others]. 2003. Association between severity of prescribed burns and subsequent activity of conifer-infesting beetles in stands of longleaf pine. Forest Ecology and Management. 185: 327-340.

Turner, R.L.; Van Kley, J.E.; Smith, L.S.; Evans, R.E. 1999. Ecological classification system for the national forests and adjacent areas of the West Gulf Coastal Plain. Nacogdoches, TX: The Nature Conservancy. 305 p.

Wade, D.D.; Johansen, R.W. 1986. Effects of fire on southern pine: observations and recommendations. Gen. Tech. Rep. SE-41. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 14 p.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth, and management. 1st ed. Washington, DC: Charles Lathrop Pack Forestry Foundation. 429 p. In cooperation with: The U.S. Department of Agriculture Forest Service.

Wolters, G.L. 1982. Longleaf and slash pine decreases herbage production and alters herbage composition. Journal of Range Management. 35(6): 761-763.

Wood, S.L. 1982. The bark and ambrosia beetles of North and Central America (Coleoptera: Scolytidae), a taxonomic monograph. Provo, UT: Great Basin naturalist memoirs, Brigham Young University. 1359 p.

BURNING FOR CONSERVATION VALUES: SHOULD THE GOAL BE TO MIMIC A NATURAL FIRE REGIME?

Sharon M. Hermann, John S. Kush, John C. Gilbert, and Rebecca J. Barlow[1]

Abstract--Managers are often asked to include conservation values in forest management plans. In the upland coastal plain of the southeastern United States, fire is an important natural process and a vital land management tool. Many native ecosystems are dependent on frequent burns. It is often suggested that mimicking a natural fire regime is the best way to improve and maintain conservation values in many forest types. Unfortunately, fire return interval has been the primary component of a fire regime historically considered, with seasonality of fire generally playing a lesser role. Here, we review what constitutes a fire regime and present data from two long-term burn treatments based in naturally regenerated loblolly-shortleaf pine (*Pinus taeda* L. - *P. echinata* Mill.) and longleaf pine (*P. palustris* Mill.). The information is used to: (1) consider how fire return interval and/or season of burn influence stand structure, and (2) determine if applying one or both of these components of a natural fire regime is likely to meet desired outcomes for conservation concerns. Data from the long-term studies indicate that limiting consideration to frequency is unlikely to produce desired results. In addition, the combination of natural frequency and season of burn may not always be successful. A more productive goal is to mimic long-term outcomes of natural fire regimes. In the modern landscape this will likely require innovative uses of prescribed fire and, at times, supplemental treatments to meet the needs of conservation concerns in upland coastal plain pine forests.

INTRODUCTION

The importance of fire in restoring and maintaining numerous native ecosystems is well recognized. When a significant management goal is to promote conservation values, a common recommendation is to implement burn plans that mimic natural fire regimes. While this appears to make intuitive sense there is increasing evidence that promoting this goal may be unproductive and/or impossible in the modern landscape. This issue coincides with increasing anthropogenic pressures on many natural landscapes and so enhancing even more our need to improve management to promote conservation concerns. An expanding wildland-urban interface coupled with growing concerns for the ecological quality of remaining natural areas means prescribed fire must be used in the most productive ways possible. At the national level, efforts have increased to assess whether the effects of prescribed burns are adequate replacement for natural fires (c.f. Nesmith and others 2011). Increasing concerns for conservation and management of fire-maintained ecosystems in the U.S. southeastern coastal plain suggest a need to consider where the regional goal for prescribed burning in conservation areas will benefit from attempting to mimic a natural fire regime.

In this paper we consider whether a goal of mimicking natural fire regimes is likely to be effective in promoting conservation values in the coastal plain of the southeastern United States. We provide a short overview of relevant data from two long-term fire studies in the lower coastal plain: (1) Tall Timbers Research Station in north Florida, where the Stoddard fire plots support loblolly-shortleaf pine in the overstory; and (2) Escambia Experimental Forest in south Alabama, where fire plots are dominated by an overstory of longleaf pine. Although targets for sampling differed between the two case studies, the overall results provide a useful basis for comparison. We discuss selected results from each case study and consider what this information reveals about the utility of the goal of mimicking a natural fire regime to promote conservation values.

Prescribed burning is an especially significant management tool in longleaf pine (*Pinus palustris* Mill.) and loblolly-shortleaf mixed pine (*P. taeda* L.-*P. echinata* Mill.) forests; without frequent application of fire these ecosystems experience hardwood encroachment and eventually lose the open canopy and unobstructed midstory required by specialist species of these forests such as the red-cockaded woodpecker [*Picoides borealis* (Vieillot)] and gopher tortoise [*Gopherus polyphemus* (Daudin)]. Van Lear and others (2005) noted that most federally listed southeastern vertebrate species were

[1]Assistant Professor, Auburn University, Department Biological Sciences, Auburn University, AL 36849; Research Fellow and Research Associate, respectively, Auburn University, School of Forestry and Wildlife Sciences, Auburn University, AL 36849; and Associate Extension Professor, Auburn University, Alabama Cooperative Extension System, School of Forestry and Wildlife Sciences, Auburn University, AL 36849.

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associated with longleaf pine ecosystems, and Hermann and others (2007) determined that this relationship was likely related, in large part, to the species need for habitat structure provided by these native forests. Almost 225 years ago Bartram (1791) described the coastal plain as "… mostly a forest of the great long-leaved pine, the earth covered with grass, interspersed with an infinite variety of herbaceous plants …". It is this habitat structure and not the specific species of tree that is required by many of the vertebrate species of conservation concern, and it is widely accepted that to maintain open habitat structure there must be frequent burns. Because restoring and maintaining habitat structure are significant goals of land management and conservation efforts, it is important to understand the effects of fire on this critical forest element. In longleaf and loblolly-shortleaf pine forests, encroachment by hardwood species is mitigated by fire. Fire also indirectly influences canopy openness. Although there is some debate on the desired minimum level of openness, a general rule of thumb currently applied is 50 percent or greater for upland pine habitats (e.g. Florida Fish and Wildlife Conservation Commission 2007).

FIRE REGIMES

Before delving into details of specific fire regimes, it is useful to define the topic and consider various components that are often listed as descriptors or factors. Unless fires are very uncommon in an ecosystem (e.g. once in many decades to centuries), consideration of the effects of a single fire may not reveal much about the likely ecological trajectory of the habitat. In other words, it is often not only productive but often necessary to consider the effects of a series of burns over time. A simple definition of a fire regime is the range of factors that describe multiple burns over a designated time period. We base our list of factors that make up a fire regime, in part, on the glossary of the Fire Effects Information System (2013); these factors include:

- frequency (mean number of fires/time period) or return interval (mean time between fires)
- season or month of burn
- day of burn conditions
- ignition pattern
- area and/or placement in the landscape
- intensity (heat/unit time)
- severity (impact on the ecosystem)
- synergism (interactions with other disturbances such as drought or windstorm)

Early in the history of prescribed fire, return interval (frequency) was the primary concern in burning for its ability to reduce fuels and enhance conservation values. In recent years, seasonality has been added as an additional target especially when conservation values are involved. General considerations of natural fire regimes, especially on large blocks of public lands, are often limited to frequency or return interval while plans for specific sites may include seasonality. One of our case studies is based on only differing burn frequencies while the other combines frequency and season of burn (in this case spring). It is widely understood that burns in the growing season were more likely on the landscape prior to European settlement than were dormant season (winter) burns.

ESTIMATES OF NATURAL FIRE RETURN INTERVALS (FREQUENCIES)

There have been a number of efforts to map coarse-scale general estimates of fire return intervals across the United States. We discuss three of them in relation to our two case studies.

1. An often-cited map used by many agency programs is found in Brown (2000). This map provides estimates of fire return intervals and general fire types related to pre-European settlement conditions based on Kuchler's Potential Natural Vegetation Types. It is, by design, coarse-scaled and almost all of the coastal plain falls under the category of "understory fires 0 to 10 years".
2. Frost (1998) created a map based on historical documentation and topography. It provides a more ecological and detailed classification system compared to Brown (2000). Frost (1998) suggests a fire return of 1 to 3 years for the lower coastal plain and 4 to 6 years for the upper coastal plain.
3. Guyette and others (2012) base fire return intervals primarily on plant chemistry of natural vegetation and climate. In this map, a mean fire return interval estimated at < 2 years is indicated for portions of the lower coastal plain of peninsular Florida, extending north into central Georgia. A mean fire return interval of 2 to 4 years

is estimated for almost all of the remainder of the coastal plain.

Although differences in fire return interval are small among the three maps, when the fire return interval is short, a shift of even one or two more burns every decade results in significantly more fire on the ground over a few decades. This is important to remember when considering the results of the two case studies. In addition, it is important to note that none of the three fire maps pays direct attention to seasonality of burn. Guyette and others (2012) indirectly address the issue when they employ proxies for mean maximum temperature, and Frost (1998) discusses the fire regime factor. However, it is generally recognized that season of burn is an important aspect of natural fire regimes; this may be especially true in the regimes that are based on frequent burns.

TWO CASE STUDIES

Tall Timbers Research Station (Stoddard Fire Plots): Fire Return Interval Treatments

Background--Tall Timbers Research Station is located in Leon County, FL. The long-term Stoddard fire plots were established in naturally regenerated loblolly-shortleaf pine with mature old-field ground cover following agricultural use 50 years or more before the study began. Although wiregrass (*Aristida beyrichiana* Trin. & Rupr.) is found on nearby properties, past land use may have eliminated this species from the fuel base of this site. Glitzenstein and others (2012) provide detailed information on the plots and the site. In this study, the only fire regime factor that was varied was fire return interval. All burns occurred in the dormant season (late February to early March). Plans for this project included burns later in the spring, but after the first set of burns it was decided to eliminate that treatment. The fire crew reported that it was difficult to burn effectively during spring because once "greened up" occurred, old field herbaceous plants did not readily burn in the higher humidity conditions of the season.

We targeted six of the treatments: fires every 1, 2, 3, 5, 9, and 20 years. We do not consider the influence of time since last burn. Data presented here were collected 35 years after the beginning of the study and do not include fires that were conducted following recent modification of the plots (Glitzenstein and others 2012). We document effects of fire frequency on three aspects of habitat structure: mean percent canopy cover, mean shrub height, and mean grass height. See Glitzenstein and others (2012) for information on fire frequency and plant species composition on the Stoddard fire plots.

Methods--Each fire frequency treatment was administered to three replicate plots (0.5 acre each). Measurements of percent canopy cover were made using a hemispherical densitometer approximately every 3 feet along a 60-foot diagonal transect in the center of the plot. Height of shrubs and grasses was assessed using the same diagonal transect but as a 3-foot wide belt placed on one side of each transect.

Results

Habitat structure--Figure 1a illustrates the estimated mean percent canopy cover averaged over three plots/fire frequency. Burns applied every 3 years or less frequently resulted in 75 percent or greater canopy cover after 35 years. Burns every 2 years maintained canopy cover at approximately 50 percent, generally considered to be along the borderline acceptable conditions for conservation lands. Annual fire maintained an open canopy with approximately 25 percent cover. Heights of hardwood stems averaged over 4-feet tall for burns applied less often than every 1 or 2 years (fig.1b) and was significantly different from more frequently burned plots. Conversely, the mean maximum height of grasses generally declined with increasing time between fires (fig. 1b).

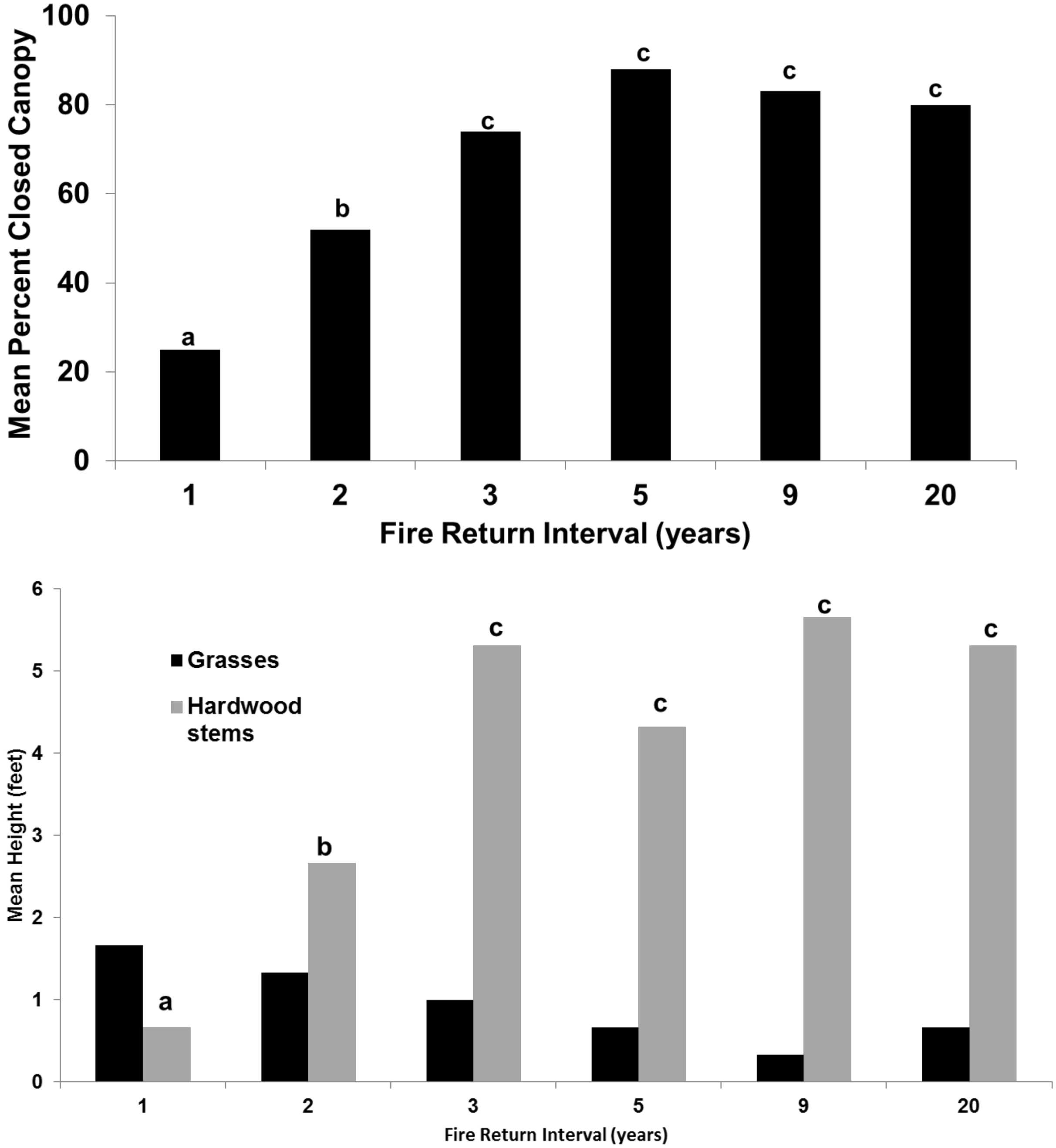

Figure 1--Relationship between fire return interval and measures of vegetation structure after 35 years of burn treatments on the Stoddard fire plots at Tall Timbers Research Station, Leon Co, FL. Fire return intervals were 1, 2, 3, 5, 9, and 20 years. Different small letters indicate significant difference among treatments ($p \leq 0.05$). (A) Percent canopy cover. (B) Maximum height of grasses and hardwood stems.

Natural fire return intervals--The location of the Tall Timbers Stoddard Fire Plots falls within: (a) the 0 to 10 year fire return interval proposed by Brown (2000); (b) the 1 to 3 year fire return interval indicated by Frost (1998); and (c) on the border of the < 2 years and the 2 to 4 year return interval proposed by Guyette and others (2012). Results of 35 years of fire frequency treatments indicate that a fire frequency of every 3 years or less does not maintain desired habitat structure. This is despite fire every 3 years falling within estimates of the natural fire regime proposed by

Brown (2000) and Frost (1998). The maintenance of desired habitat structure does appear to fit with one of two shortest fire return intervals indicated by Guyette and others (2012). Unfortunately, burning every 2 years does not appear to result in ideal canopy cover.

Escambia Experimental Forest: Fire Return Combined with Season of Burn Treatments

Background--Escambia Experimental Forest is a USDA Forest Service research site located in Escambia County, AL. The long-term fire plots were established in naturally regenerated longleaf pine with native ground cover on land that has never been in agriculture. There is no record of wiregrass ever occurring on the site. In this study, both fire return interval and season of burn were considered. Fire return intervals were limited to 2-, 3-, and 5-year treatments combined with two seasons of burn: winter (W) and spring (S). This resulted in six treatments (W2, W3, W5, S2, S3, and S5). In the current paper, we do not consider time since burn; data present in the current paper were collected 25 years after the beginning of the study. We evaluate effects of fire frequency coupled with month of burn on two habitat variables: number of hardwood stems per acre and basal area of hardwood stems per acre. Both number and size of hardwood stems may be important in determining habitat value to species of special conservation concern.

Methods--Each fire-frequency-by- season treatment was represented by three replicate plots (0.5 acre each). Number of hardwoods stems ≥ 1 inch d.b.h. (diameter at breast height) was tallied, stem diameters were measured, and basal area was calculated. No hardwood stem exceeded 10 inches d.b.h.

Results

Hardwood encroachment--After 25 years, density of hardwood stems was significantly different among treatments. Winter burns were associated with more hardwood stems (fig. 2a) compared to spring burns, and a 5-year fire return interval also resulted in more hardwood stems than did shorter fire return intervals. In addition, plots with higher density of hardwood stems also supported a higher basal area (fig. 2b). Although we have not yet assessed canopy cover, we predict that winter-burned plots will also have greater cover than spring-burned plots.

Comparison to natural fire return intervals--
The location of the Escambia Experimental Forest fire plots falls within: (a) the 0 to 10 year fire return interval proposed by Brown (2000); (b) the 1 to 3 year fire return interval indicated by Frost (1998); and (3) the 2 to 4 year return interval proposed by Guyette and others (2012). Two of the fire frequencies applied to the Escambia plots fall within the range of all three estimates of natural fire frequencies (every 2 or 3 years); only the 5-year burn cycle is viewed as unnatural. Burning every 2 or 3 years results in fewer hardwood stems and lower basal area compared to burning every 5 years. However, all winter burn plots support more and larger hardwoods than are desired. Only spring burns are effective in controlling encroachment over the 25-year period.

DISCUSSION

Although the two long-term fire studies differ in land use history and dominant canopy species, the general result is the same. Frequent fires (1-year to, at most, 3-year return intervals) are required to maintain ecologically desirable forest structure. Growing season burns may enhance effects produced by high frequency but only if the type of ground layer vegetation permits effective fire during that time of the year and only if local situations permit burning in the growing season on a consistent basis. It should also be noted that prolonged periods of annual fires will also not produce desired results because of lack of regeneration of most plant species. Adaptive management is critical.

If Tall Timbers Research Station plots are representative of the coastal plain that has been altered by agriculture, then the desired fire return interval for such sites may be as narrow as burns every 1 to 2 years. This is the same frequency that is often applied on many quail-hunting properties across the region and is shorter than the 3-year frequency that appears in many agency burn plans. Masters and others (2007) noted that “Less than a 3-year interval is recommended if hardwoods are problematic …” Although most individual burns do not result in drastic changes, effects of slightly less frequent

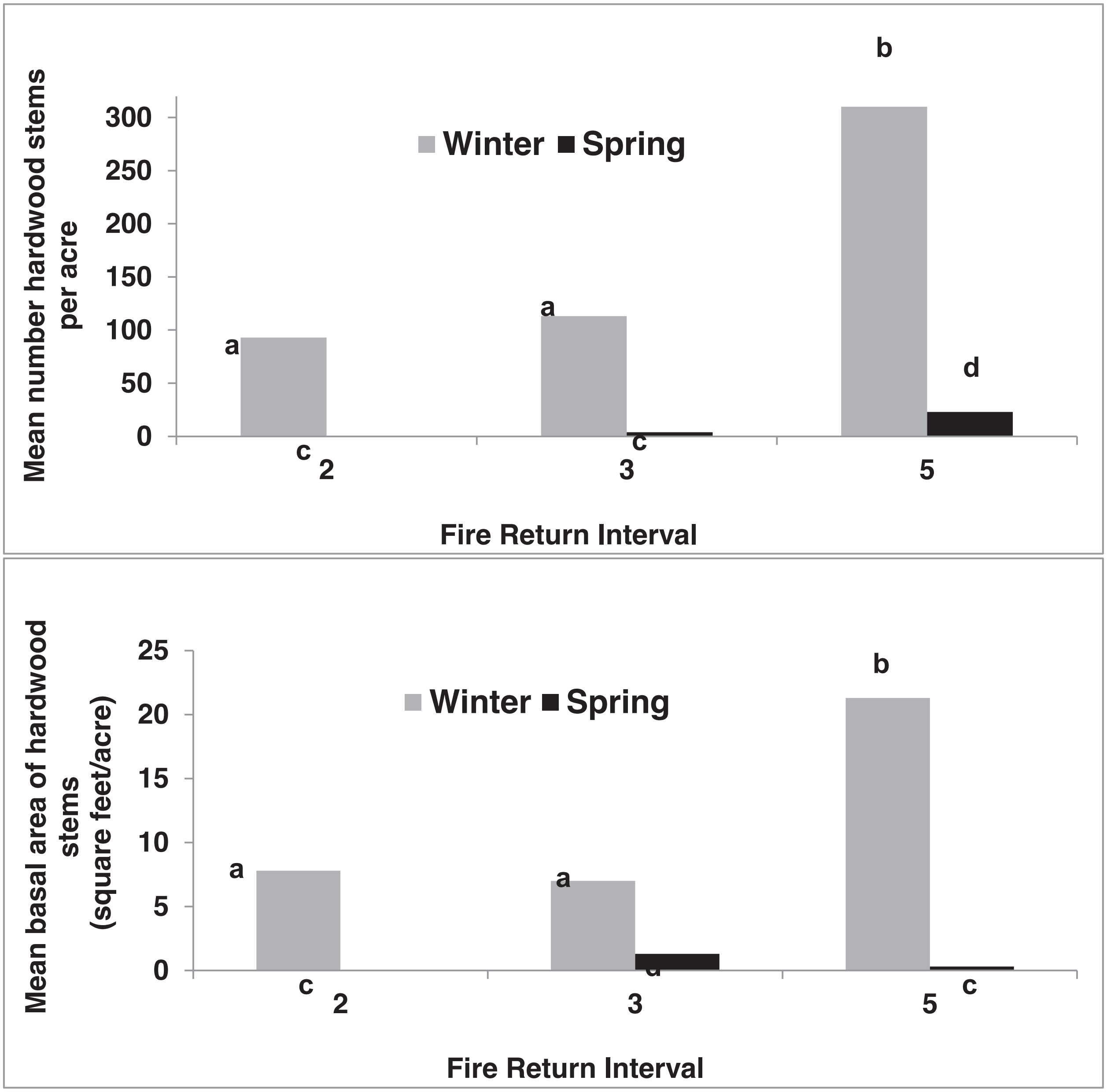

Figure 2--Relationship between fire return interval couple with season of burn treatments and hardwood stems after 25 years of treatments on plots at Escambia Experimental Forest, Escambia Co, AL. Fire return intervals were 2, 3, and 5 years; seasons were winter (W) and spring (S). Treatments were W2, S2, W3, S3, W5, and S5. Different small letters indicate significant difference among treatments ($p \leq 0.05$). (A) Estimated density (number of stems per acre) of hardwood stems ≥ 1 inch d.b.h. (B) Estimated basal area (square inches) of hardwood stems ≥ 1 inch d.b.h.

Fires are likely compounded over time. This is, in part, because "… fire season appears to have less of an influence on hardwood encroachment where the understory is dominated by bluestems or is the result of old-field succession" (Masters and others 2007). The Tall Timber plots also provide insight into the frequency of fire that may be required to maintain canopy openness at 50 percent or greater; only two treatments (annual and biennial fires) met those criteria after 35 years of treatment.

Escambia Experimental Forest fire plots integrate both frequency and season of burn in the long-term study of fire effects. Additionally, this site is less disturbed than the Tall Timbers site. It should also be remembered that different metrics assess fire effects at the two sites. At this point in time, it is not possible to determine if any one of the measurements is more important than the others in understanding the varying results. However, it appears as if the combination of two factors of a natural fire regime (frequency and season) may be related to maintaining a desired forest structure over long periods of time.

If adding growing-season burns to the tool box of prescribed fire enhances desired outcomes, then why is it not always used? There are many logistical and economic reasons including: (1) smoke management and air quality issues, (2) concerns for wildlife, (3) staffing conflicts related to wildfire season, and (4) the number of available legal burn days during the growing season. Of course there are more factors in a fire regime in addition to frequency and season but unfortunately adding more realistic day-of-burn conditions or more natural fuel types is not always possible. In fact, the day-of-burn conditions that would result in the best hardwood control may not be appropriate burn days because of the associated extreme weather (high temperature, low relative humidity, etc.). In addition, although a natural fire regime may be able to maintain a site in a desired ecological condition, it may not be adequate to recover a site once hardwoods have encroached.

The goal of applying a natural fire regime is laudable but perhaps not realistic in the modern landscape. However, understanding what such a goal would entail provides us with important ways to assess the outcomes of burn plans. Schmidt (1996) asked "Can we restore the fire process?" and "What awaits us if we don't?" He concluded that the goal should not be to replicate the past but to consider the future. In other words, the goal should not be to apply a natural fire regime but rather to obtain desired results of either: (1) maintaining current high-quality habitat conditions, or (2) nudging the current conditions to an improved state. Improved understanding of natural fire regimes can provide valuable information for meeting those goals and can guide adaptive management.

By necessity large public lands may rely on general canned burned plans. Sometimes this means that all units under fire management will be treated in a similar fashion. This, coupled with applying a natural frequency of fire in an unnatural season, may mean that conditions at ecologically valuable sites will degrade slowly over time. Hiers and others (2003) provide a useful and pragmatic approach to prioritizing sites for prescribed burning. Such an approach, coupled with a better understanding of natural fire regimes, could improve management and conservation efforts on many sites. Important regional efforts, such as those to restore longleaf pine, will ultimately fail to benefit conservation efforts if long-term effects of prescribed fire are not improved. When planted trees are large enough to contribute to the desired habitat structure, fire must be applied in ways to maintain critical ecological values or significant ecological benefits expected from the effort will not materialize.

Given all of the challenges imposed by a modern landscape, is the future for prescribed fire as a primary tool for ecological management hopeless? No! However, adaptive management must be implemented to enhance desired fire effects of prescribed burning programs. Additional tools may need to be added to the fire management tool box, including mechanical and chemical treatments. We do not promote using these treatments as on-going surrogates for fire but rather as a way to periodically boost the effectiveness of fire. Again, adaptive management is critical.

Even if fire ecologists can adequately describe a natural fire regime for the southeastern coastal plain, the goal should not be to implement it. Fuels loads and vegetation composition probably differ from natural conditions at many, if not most, sites. In addition, conditions of the surrounding landscape, coupled with social concerns, often limit growing-season burns. If only one factor of a natural fire regime, such as frequency, is consistently applied, then the outcome expected of a natural fire regime cannot be obtained over a long time period. However, if fire practitioners focus on obtaining outcomes of a natural fire regime rather than the process itself, then use of adaptive management will promote desired ecological values.

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LITERATURE CITED

Bartram, W. 1791. Travels through North and South Carolina, Georgia, East and West Florida [1955 reprint]. New York: Dover Publications. 414 p.

Brown, J.K. 2000. Introduction and fire regimes. In: Brown, J.K.; Smith, J.K., eds. Wildland fire in ecosystems: effects of fire on flora . Vol. 2. Gen. Tech. Rep. RMRS-GTR-42. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 1-8. Chapter 1.

Fire effects information system. 2013. Glossary. http://www.fs.fed.us/database/feis/glossary. [Date accessed: June 2013].

Florida Fish and Wildlife Conservation Commission. 2007. Gopher tortoise management plan. Tallahassee, FL: Florida Fish and Wildlife Conservation Commission. 143 p.

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.

Glitzenstein, J.S.; Streng, D.R.; Masters, R.E. [and others]. 2012. Fire-frequency effects on vegetation in north Florida pinelands: another look at the long-term Stoddard fire research plots at Tall Timbers Research Station. Forest Ecology and Management. 64: 197-209.

Guyette, R.P.; Stambaugh, M.C.; Dey, D.C.; Muzika,R. 2012. Predicting fire frequency with chemistry and climate. Ecosystems. 15: 322–335

Hermann, S.M.; Kush, J.S.; Gjerstad, D. 2007. Emerging issues in forest health and their significance for rare species of the southeastern United States. In: HydroGeoLogic, eds. Proceedings from the Southeast Region threatened, endangered, and at-risk species workshop. Reston, VA: Department of Defense Legacy Resource Management Program: E3-E22.

Hiers, J.K.; Laine, S.C.; Bachant, J.J. [and others]. 2003. Simple spatial modeling tool for prioritizing prescribed burning activity at the landscape scale. Conservation Biology. 17(6): 1571-1578.

Masters, R.E.; Robertson, K.M.; Palmer, W.E. [and others]. 2007. Red Hills forest stewardship guide - 2007. Misc. Series No. 12. Tallahassee, FL: Tall Timbers Research Station. 78 p.

Nesmith, J.C.B.; Caprio, A.C.; Pfaff, A.H. [and others]. 2011. A comparison of effects from prescribed fires and wildfires managed for resource objectives in Sequoia and Kings Canyon National Parks. Forest Ecology and Management. 261(7): 1275-1282.

Schmidt, R. G. 1996. Can we restore the fire process? What awaits us if we don't? In: Hardy, C.C.; Arno, S.F., eds. The use of fire in forest restoration. Gen. Tech. Rep. INT-GTR-341. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station: 85-86.

Van Lear, D.H.; Carroll, W.D.; Kapeluck, P.R.; Johnson, R. 2005. History and restoration of the longleaf pine–grassland ecosystem: implications for species at risk. Forest Ecology and Management. 211: 150–165.

MYCORRHIZAE PROMOTE FIRE ADAPTATION IN OAK-HICKORY FORESTS IN EASTERN USA

Aaron D. Stottlemyer, G. Geoff Wang, Thomas A. Waldrop, and Christina E. Wells[1]

Prescribed fire is commonly used in silvicultural programs designed to promote oak (*Quercus* spp.) and hickory (*Carya* spp.) regeneration in eastern deciduous forests (Brose and others 2008). Thick bark, hypogeal germination, large root systems, repeated-prolific sprouting, and the ability to compartmentalize scars are well-known characteristics that enable oaks and hickories to tolerate fire (Burns and Honkala 1990) unlike certain mesophytic competitors including maples (*Acer* spp.), black cherry (*Prunus serotina* Ehrh.), and yellow-poplar (*Liriodendron tulipifera* L.). Relationships between mycorrhizas and fire adaptation in eastern ecosystems, however, are poorly understood.

Mycorrhizas are symbiotic associations between soil fungi and host plants. When plant roots become colonized by mycorrhizal fungi, their ability to absorb nutrients (phosphorous and nitrogen) and water is greatly enhanced (Smith and Read 1997). Mycorrhizal colonization also lengthens the life of roots, protects against pathogens, and improves early survival and growth (Perry and others 1987). Oaks and hickories are ectomycorrhizal (ECM) while many of their competitors (e.g., maples, black cherry, and yellow-poplar) are vesicular-arbuscular mycorrhizal (VAM). Objectives of this study were to examine the impacts of prescribed burning and the influence of the pre-treatment plant community on ECM and VAM colonization of new seedlings.

Peak temperature and heating duration were measured during site-preparation prescribed burns in South Carolina's upper Piedmont. Greenhouse assays of intact soil cores (Brundrett and others 1996) collected after the burns revealed that the amount of ECM colonization of loblolly pine (*Pinus taeda* L., an ECM host) seedlings was positively correlated with both metrics of fire behavior. Conversely, VAM colonization of corn (*Zea mays* L., a VAM host) seedlings was negatively correlated with peak temperature and heating duration. In addition, best subset regression models revealed that post-burn mycorrhizal colonization was associated with the genera, life stage, and growth habits of plants in the pre-treatment vegetation assemblage.

Results of this study suggest that mycorrhizae are important factors in determining the tolerance (or sensitivity) of forest plants to fire. Moreover, the mycorrhizal colonization of new seedlings after fire depends on the composition and structure of the plant community existing prior to treatment. Finally, with the potential of mycorrhizae to influence regeneration dynamics in eastern deciduous forests, additional work is needed to determine whether traditional silvicultural practices sustain mycorrhizae in this region.

LITERATURE CITED

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. 90 p.

Brundrett, M.C.; Bougher, N.; Dell, B. [and others]. 1996. Working with mycorrhizas in forestry and agriculture. Monogr. 32. Canberra, Australia: Australian Centre for International Agricultural Research [ACIAR]. 374 p.

Burns, R.M; Honkala, B.H., tech. cords. 1990. Silvics of North America: Hardwoods. Agric. Handb. 654.

[1]Assistant Professor, Penn State University, Wildlife Technology Department, DuBois, PA 15801; Professor, Clemson University, Agricultural, Forest, and Environmental Sciences, Clemson, SC 29634; Research Forester, USDA Forest Service, Southern Research Station, Clemson, SC 29634; and Associate Professor, Clemson University, Agricultural, Forest, and Environmental Sciences, Clemson, SC 29634.

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Washington, DC: U.S. Department of Agriculture. 877 p. Vol. 2.

Perry, D.A.; Molina, R.; Amaranthus, M.P. 1987. Mycorrhizae, mycorrhizospheres, and reforestation: current knowledge and research needs. Canadian Journal of Forest Research. 17: 929-940.

Smith, S.E.; Read, D.J. 1997. Mycorrhizal symbiosis. 2[d] ed. London: Academic Press. 605 p.

CURRENT-YEAR FLUSH AND NEEDLE DEVELOPMENT IN LONGLEAF PINE SAPLINGS AFTER A DORMANT SEASON PRESCRIBED FIRE

Shi-Jean S. Sung, James D. Haywood, and Mary Anne S. Sayer[1]

Abstract--A longleaf pine (*Pinus palustris* Mill.) field performance study was established in central Louisiana in 2004. The study has received three prescribed burns (February 2006, May 2009, and February 2012) since establishment. In late April 2012, 35 saplings were selected and classified based on ocular estimates of needle mass scorch percentages. Mean needle scorch percentages for the lightly (LS), moderately (MS), and severely scorched (SS) saplings were 17, 50, and 99 percent, respectively. The prescribed fire did not change the temporal development of all flushes and their needles. However, the first three flushes of the SS saplings were significantly shorter than those of the LS saplings. The fourth and fifth flushes did not differ in lengths among scorch classes. Compared to flushes developed in 2011, which was a severe drought year, flush growth in the SS saplings was less in the fire year. Needles from the first SS flushes extended faster than those from the LS and MS saplings in May. Final needle lengths for the first and fourth flushes were not different among scorch classes whereas needles of the second and third SS flushes were shorter than those of less-scorched saplings. The physiological parameters, such as photosynthesis, stomatal conductance, and chlorophyll content, generally did not vary among scorch classes in August and September. However, these parameters were greater for the first SS flush needles than those of less-scorched saplings. The possibility of residual impact from prescribed fire on sapling growth in a subsequent year is discussed.

INTRODUCTION

The loss of 97 percent of the pre-European settlement longleaf pine (*Pinus palustris* Mill.) ecosystem acreage in the South was attributed to extensive harvesting of longleaf pine for timber and naval store products, conversion of lands supporting longleaf pine to croplands, pasture, or other fast growing southern pine species, and exclusion of fire from the landscape (Brockway and Outcalt 1998, Landers and others 1995, Outcalt 2000). For the last three decades, many public, industrial, and private land managers and owners have been actively restoring longleaf pine ecosystems in the southern United States (Barnett 2002, Boyer 1989, Landers and others 1995). Much of the original longleaf pine range is within 240 km of the Atlantic or Gulf coasts, a region frequented by tropical wind storms and hurricanes (Landers and others 1995). Longleaf pine trees suffered less wind damage than loblolly pine (*Pinus taeda* L.) during Hurricanes Hugo in South Carolina (Gresham and others 1991) and Katrina in Mississippi (Johnsen and others 2009). Due to longleaf pine's ability to withstand storm damage, many forest practitioners decided to restore this species to the hurricane-prone Atlantic and Gulf coast states. Another reason to restore longleaf pine is to arrest the decline of many associated plants and animals that depend on these highly biologically diverse ecosystems (Brockway and others 1998, Mitchell and others 2006). Because of the efforts to restore longleaf pine ecosystems, the Range-wide Conservation Plan for Longleaf Pine was compiled and proposed by America's Longleaf (2009). This plan has a goal of increasing longleaf acreage from 1.4 million to 3.2 million ha by 2024 (America's Longleaf 2009). To be successful in this conservation endeavor, prescribed fire regimes need to be implemented in natural and plantation longleaf pine forests.

Most longleaf pine seedlings experience delayed height growth and have been known to remain in the grass stage for up to 9 years (Wahlenberg 1946). The grass-stage seedlings are vulnerable to brown-spot needle blight fungal infection (caused by *Mycosphaerella dearnessii* Barr), vegetation competition, and smothering by dead grass and litter (Wahlenberg 1946). On the one hand, prescribed fire has been used as a management tool in longleaf pine forests to reduce brown-spot needle blight-related mortality and to relieve longleaf from vegetative competition (Grelen 1983, Haywood 2007). On the other hand, prescribed fire has negatively affected longleaf pine productivity (Boyer 1987, Boyer and Miller 1994, Haywood 2009, Haywood and Grelen 2000). Ford and others (2010) observed that adult longleaf pine tree growth was reduced during fire years, but tree

[1]Research Plant Physiologist, Supervisory Research Forester, and Research Plant Physiologist, respectively, USDA Forest Service, Southern Research Station, Pineville, LA 71360.

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growth during years between fire events did not differ significantly among fire-return intervals.

Prescribed fire regimes routinely implemented in longleaf pine forests on the Kisatchie National Forest in Louisiana have 2- to 3-year fire-return intervals. The burns are alternated between dormant and growing seasons with the first burn occurring 13 to 15 months after outplanting. In November 2004, 28-week-old longleaf pine seedlings grown in six kinds of containers were outplanted at the Kisatchie National Forest. The trial site has been prescribe burned three times since then. The objective of this study was to follow the dynamics of flush and needle development of these longleaf pine saplings with different levels of needle mass scorch after a dormant season prescribed burn in February 2012.

MATERIALS AND METHODS

Seedling Culture and Field Establishment in the Original Study

Details of seedling culture and field establishment protocols were reported by Sword Sayer and others (2009). Briefly, longleaf pine seeds were sown in containers of three cavity sizes (54, 93, and 170 mL) with two cavity types (with and without copper oxychloride lining) in April 2004. In early November 2004, 27-week-old container-grown longleaf pine seedlings were lifted and outplanted on the same day. The field experiment site is located on the Palustris Experimental Forest within the Kisatchie National Forest in Rapides Parish of central Louisiana (31°11' N, -92°41' W). The soil is a moderately well-drained, gently sloping Beauregard silt loam (fine silty, siliceous, superactive, thermic, Plinthaquic Paludults). Mima mounds of Malbis fine sandy loam (fine loamy, siliceous, subactive, thermic, Plinthic Paleudults) are scattered across the study area. The study was a 3 by 2 randomized complete block factorial design with four replications. Twenty-four (3 cavity sizes x 2 cavity types x 4 blocks) treatment plots of 0.0576 ha (24- by 24-m) each were established. Seedlings were planted at 2- by 2-m spacing. All plots were prescribed burned in February 2006, May 2009, and February 2012.

Fire Impact Study

Two months after the latest prescribed burn, a total of 35 longleaf saplings were selected based on the ocular estimates of needle mass scorch in 10 percent increments. For accessibility, only saplings shorter than 280 cm were considered, and the original treatments were disregarded for the current study. The lightly scorched (LS) class had 10 saplings with 10 to 30 percent of needle mass scorch. The moderately scorched (MS) and severely scorched (SS) classes had, respectively, 12 saplings with 40 to 70 percent and 13 saplings with 90 to 100 percent needle scorch. Means and ranges of height and breast height diameter at the end of 2011 for these saplings are presented in table 1. Individual flush development in the leader shoot of each sapling was monitored biweekly from May through September. The last measurements were made in mid-November. One fascicle was selected from the middle section of each flush and tagged with a paper clip to track needle development over time.

Table 1--Number of saplings, mean and range of 2011 height and breast height diameter of longleaf pine saplings selected for each needle mass scorch class. Only saplings shorter than 280 cm were considered

Scorched class	Sapling number	2011 height (range)	2011 breast height diameter (range)
		-------*cm*-------	----*mm*----
Lightly[a]	10	233 (183-271)	32 (28-46)
Moderately	12	220 (177-253)	32 (20-51)
Severely	13	227 (180-262)	35 (28-41)

[a] Classification was based on the percentages of needle mass scorched by a February 2012 prescribed fire: 10 to 30, 40 to 70, and 90 to 100 percent for lightly, moderately, and severely scorched classes, respectively.

In mid-August, four saplings were randomly selected from each scorch class for photosynthesis and stomatal conductance measurements with a LiCor 6400 portable, open-system infrared gas analyzer (LiCor, Lincoln, NE). Measurements were made between 9:00 and 11:00 am and again between 1:00 and 4:00 pm on the same day on fascicles from the third flushes of 2011 and the first three or four flushes of current year. Photosynthetic active radiation was set between 1400 and 1600 $\mu E\ m^{-2}\ sec^{-1}$ with a red-blue light source, and the CO_2 level for the reference chamber was 400 ppm. The middle section of one three-needle fascicle was enclosed in the measuring chamber (2- by 3-cm) within 20 seconds of detachment from the flush. After measurements, the same fascicle was stored on ice and transported to the laboratory. Needle surface area was measured with a displacement method described 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 as described in detail by Sung and others (2010). The same saplings were measured again in mid-September following the same procedures. Number of needle fascicles in each flush of the leader shoot in these saplings was counted in mid-November.

Statistical Analysis

Using Proc GLM in SAS (2004), a one-way analysis of variance was conducted for each variable of interest against three levels of needle scorching. Means were compared using the Duncan's Multiple-Range Test at significance level 0.05. The assumptions for homogeneous variance and normality were checked using Levene's test and the Kolmogorov-Smirnov D statistic, respectively.

RESULTS

By the time the first set of measurements were made on May 1, 2012 (Julian Day, JD 122), the current-year first flushes had completed elongation (fig. 1a); the second flushes for the LS, MS, and SS saplings averaged, respectively, 10.6, 9.2, and 3.6 cm (fig. 1b); and the third flushes were about to begin elongation (fig. 1c). Monitoring the development of the third and subsequent flushes began when bud swelling was visible. The bud swelling dates for the third, fourth, and fifth flushes did not vary among scorch classes (fig. 1c, 1d, and 1e). More than half of the saplings started the elongation of the third, fourth, and fifth flushes by JD 135, 174, and 202, respectively. The linear growth rate (slope) of the third SS flushes was less than those of the LS and MS saplings (fig. 1c). Levels of needle mass scorch affected final lengths of the first three flushes with the SS flushes being shorter than the corresponding LS flushes (fig. 1a, 1b, and 1c). The first and second flushes of the MS saplings were also shorter than those of the LS saplings. Levels of needle scorch did not affect the development or final lengths of the fourth and fifth flushes (fig. 1d and 1e).

Within each scorch class, the first flush was the longest, and the subsequent four flushes were not different from each other except that in the SS saplings the second flushes were shorter than their subsequent flushes (results of statistical analysis not shown). When compared to flush growth in 2011, which had a prolonged and severe drought, the current-year first-three flushes of the SS saplings were shorter whereas the current-year first flushes of the LS saplings were longer (fig. 1f). The fourth and fifth flushes developed in 2012 were longer than those in 2011 except the fourth MS flushes.

The initial development of the first-flush needles was not monitored. The first-flush needles of the SS saplings were longer than those of the LS and MS saplings in May (JD 122, 135, and 146), although the final needle lengths were not different among scorch classes (fig. 2a). Days of year for the appearance of needle fascicles in the second and subsequent flushes were not affected by the levels of needle scorch (fig. 2b through 2e). More than half of saplings had needle fascicles emerged on JD 122, 160, 202, and 228 from the second, third, fourth, and fifth flushes, respectively. Similar to the third-flush development (fig. 1c), rates of the linear elongation by the second and third-flush needles in the SS saplings were less than those of LS and MS saplings (fig. 2b and 2c). Levels of needle scorch did not affect the fourth or the fifth-flush needle development (fig. 2d and 2e).

Mean chlorophyll a and b contents in the first-flush needles of the SS saplings sampled in August and September ranged between 21.8 and 23.8 $\mu g/cm^2$ and were greater than those (18.0 to 19.6 $\mu g/cm^2$) of the less-scorched saplings. Chlorophyll contents for the subsequent flush needles did not differ among scorch classes and ranged between 16.4 and 22.0 $\mu g/cm^2$. With one exception, the photosynthetic parameters measured were not different among scorch classes; means of all 12 saplings measured were presented in figure 3.

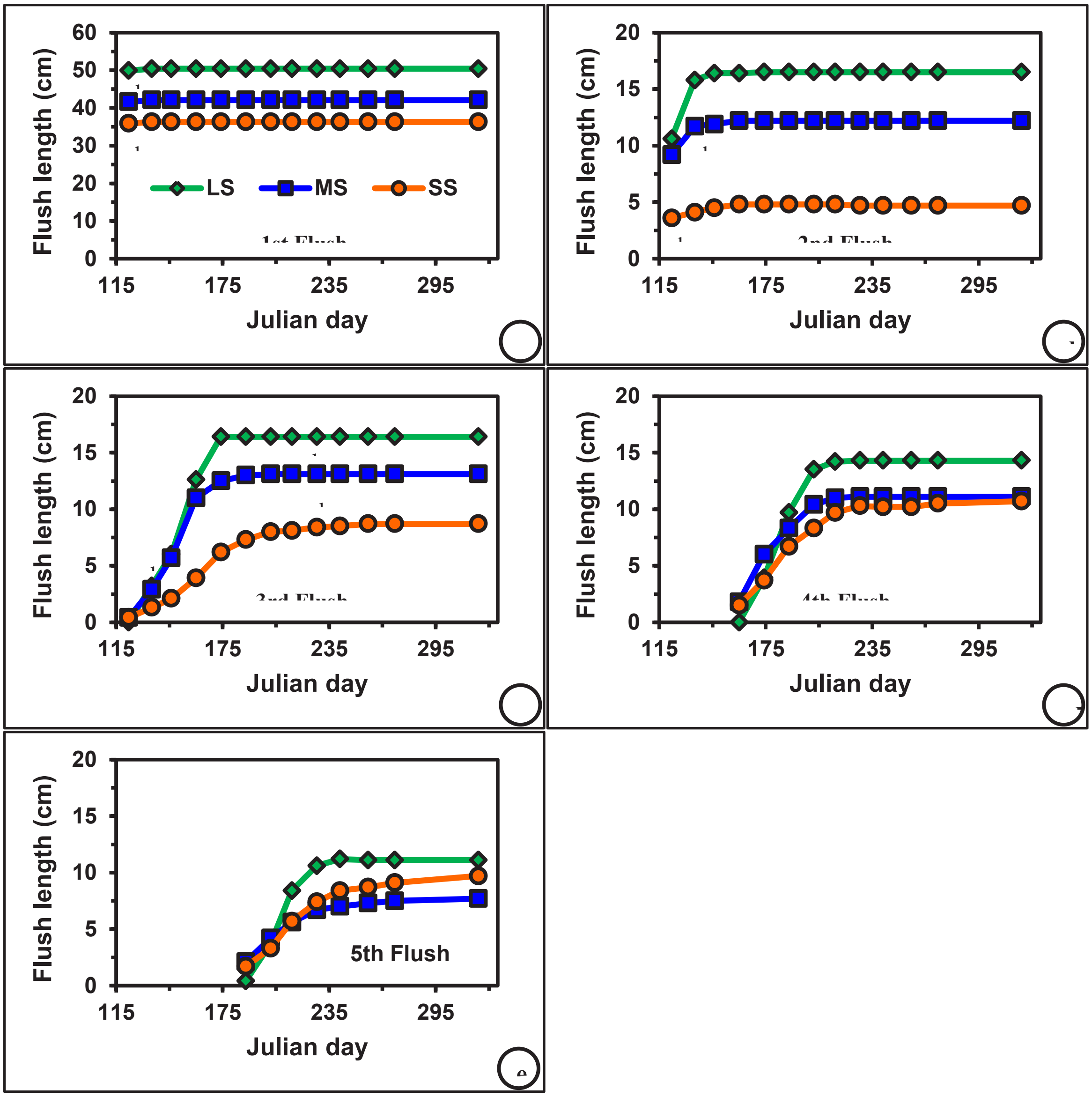

Figure 1--Temporal development patterns of current-year leader shoot flushes in longleaf pine saplings outplanted in November 2004 in central Louisiana with lightly (LS), moderately (MS), and severely (SS) needle mass scorched by a prescribed fire in February 2012. Within each flush, means with the same letter were not different at significance level of 0.05 using Duncan's Multiple-Range Test. For clarity, the same statistical test results as previous date were omitted. (a) First flush; (b) second flush; (c) third flush; (d) fourth flush; (e) fifth flush; (f) differences in final flush lengths between current year (2012) and previous year (2011).

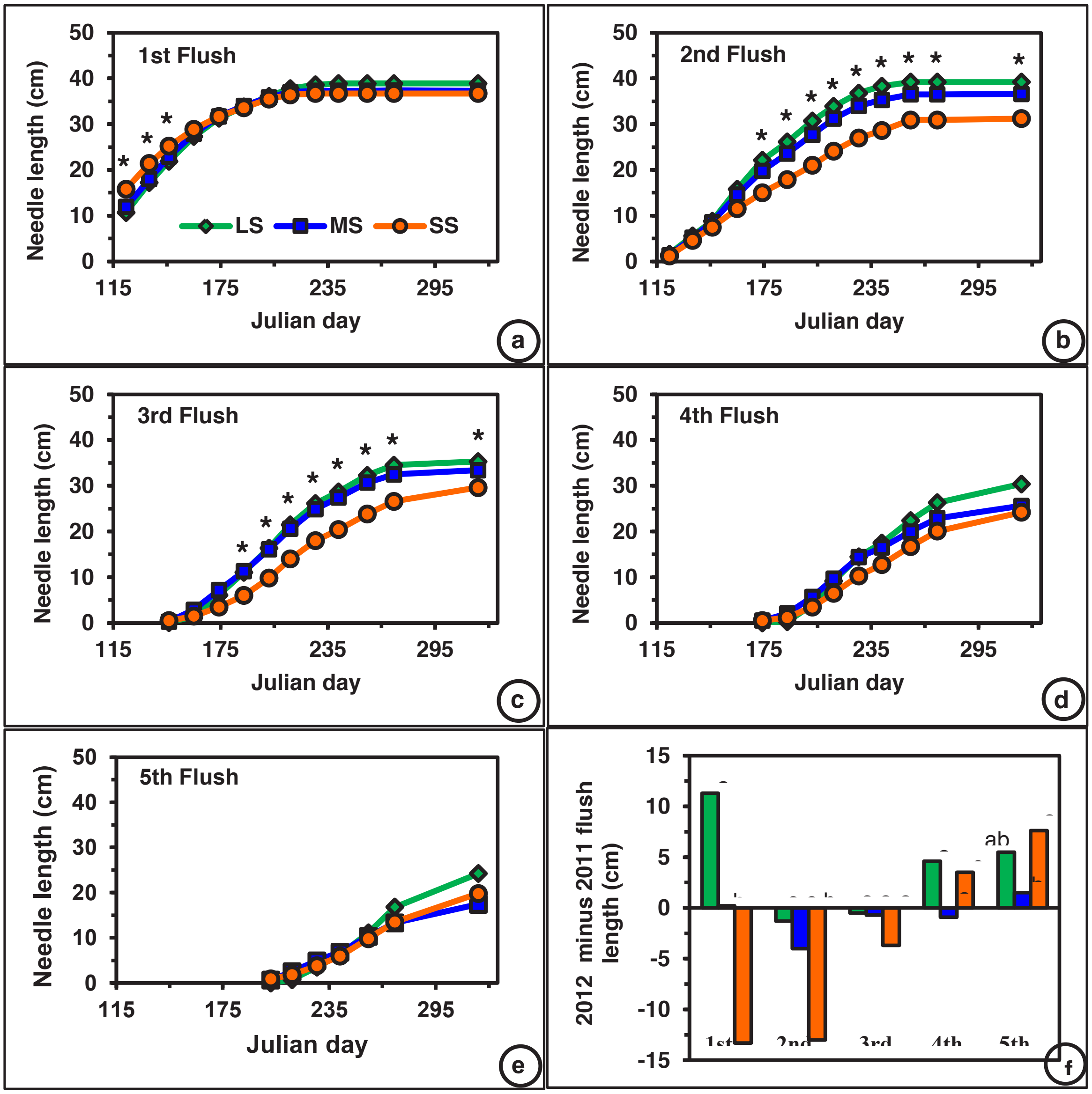

Figure 2--Temporal development patterns of needles in longleaf pine saplings outplanted in central Louisiana in November 2004 with lightly (LS), moderately (MS), and severely (SS) needle mass scorched by a prescribed fire in February 2012. Within each flush, an asterisk (*) indicates significant difference between SS saplings and the other saplings at the 0.05 level using Duncan's Multiple-Range Test; (a) first flush; (b) second flush; (c) third flush; (d) fourth flush; and (e) fifth flush.

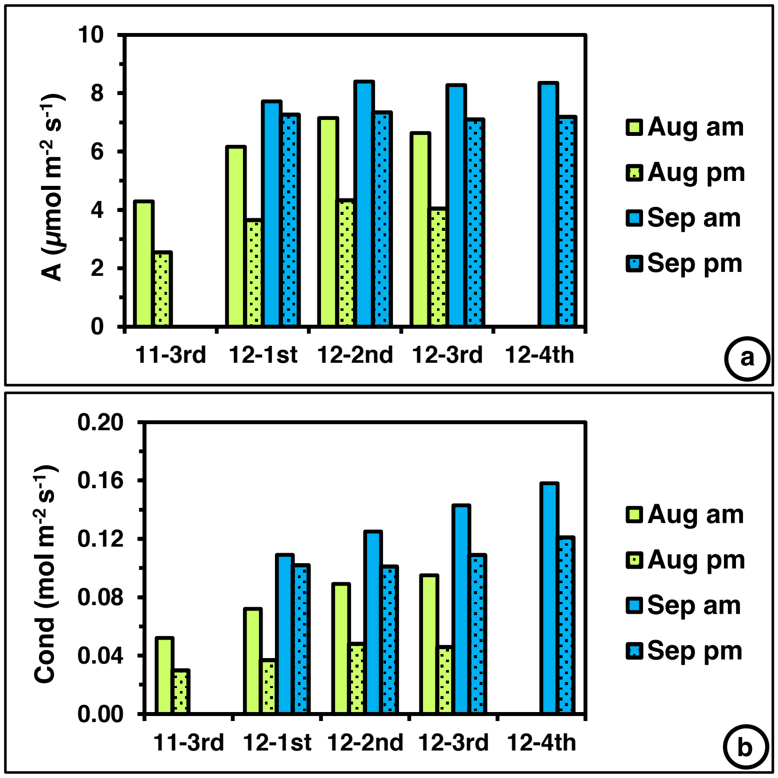

Figure 3—Means from all scorch class longleaf pine saplings for (a) photosynthetic rate and (b) stomatal conductance of needles from the third flushes of previous year (2011) and the first four current-year (2012) flushes measured in August and September. The study was prescribed burned in February 2012.

The exception is that the first SS flush needles had greater photosynthetic rate and stomatal conductance for the September morning measurements than needles of less severely scorched saplings (data not shown). In August, photosynthetic rates and stomatal conductance were lower in the afternoon than in the morning whereas no differences between morning and afternoon measurements existed in September (fig. 3). Based on the number of needle fascicles in each flush (including previous year's third flush in the LS and MS saplings), the needle surface area, and photosynthetic rate from August and September measurements, estimates of total needle surface area for the leader shoots and the amount of daily (8 hours) photosynthesis were derived (fig. 4). Compared to the LS and MS saplings, the SS saplings were more negatively impacted in their capacity to produce photosynthate by the prescribed fire (fig. 4b).

DISCUSSION

The dormant season prescribed burn did not change the temporal development patterns of flushes and needles (figs. 1 and 2) when compared to the 2007 and 2008 growth patterns reported for the same study (Sung and others 2013). Unlike the mature longleaf pine trees where there was a 30-day delay between flush elongation cessation and needle elongation (Sheffield and others 2003), needles of these young longleaf pine saplings started elongation before flush elongation was complete (figs. 1 and 2). Furthermore, the levels of needle scorch had no effects on flush or needle development patterns with two exceptions. First, both the third flushes and needles had slower linear elongation rates in the SS saplings than the other less-scorched saplings (figs. 1c and 2c). Second, the second flushes were the shortest among all flushes within the SS saplings. It was generally accepted that longleaf pine remains very susceptible to heat-related injury until the

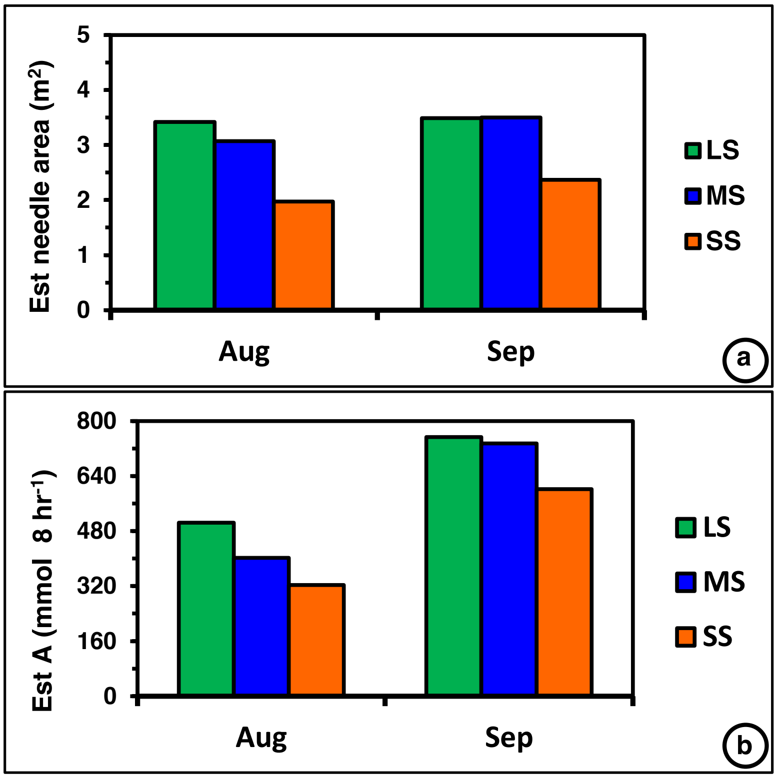

Figure 4--Estimated sums of (a) total needle surface area and (b) daily (8-hour) photosynthesis in leader shoots of longleaf pine saplings with lightly (LS), moderately (MS), and severely (SS) needle mass scorched from a February prescribed fire.

seedlings are above 1.8 m in height (Bruce 1951). However, the negative impacts of the prescribed fire on the SS saplings were readily evident with the much shorter lengths of the first three current-year flushes and of the second and third-flush needles compared to those of less severely scorched saplings. Haywood and Grelen (2000) and Haywood (2009) found that March-, May-, and July-burns reduced longleaf sapling growth with March-burn having the most negative impact. The negative impact of prescribed fire on longleaf sapling growth is also shown when growth of the first three flushes in the year with a severe drought was greater than that of the SS saplings in the subsequent fire-year that had average rainfall (fig. 1f).

Studies in loblolly pine (Chung and Barnes 1980), red pine (*P. resinosa* Ait.) (Gordon and Larson 1968), and Scots pine (*P. sylvestris* L.) (Ericsson 1978) showed that pine needles start to export photosynthate when they are almost mature. With all previous years' needles scorched by the dormant season prescribed fire, carbon needed for the growth of the first and second flushes and their needles in the SS saplings has to originate from stored reserves in stems and roots before the current year first-flush needles become a carbon source. By JD 202, the majority of first-flush needles in the SS saplings had completed elongation and were able to export photosynthate to meet the carbon demands of sinks such as needles of the third flushes and the fourth and fifth flushes and their needles as well as stem cambial tissues and fine roots (Chung and Barnes 1980, Gordon and Larson 1968, Sword Sayer and Haywood 2006, van den Driessche 1987). By JD 257, the fully extended second-flush needles joined the first-flush needles to supply carbon for growth. In less severely scorched saplings, carbon for the first flush and initial needle growth can come from current photosynthate produced by previous year's needles and from stored reserves (Dickson 1991, Kuhns and Gjerstad 1991). Shorter first flushes in the SS saplings indicate that these saplings either exhausted their reserves or stopped mobilizing more reserves before their first flushes were fully extended. Had the first flush development been monitored earlier than JD 122, we may have learned more about the rate differences of flush elongation based on possible carbon sources present (current photosynthate or reserves). Longer first-flush needles in the SS saplings observed in May resulted either from earlier appearance of needles or greater elongation rates. No conclusion can be drawn about carbon source and growth rate based on needle development.

O'Brien and others (2010) reported a negative relationship between crown scorching levels and chlorophyll contents in needles that flushed after a wildfire in a long-unburned longleaf pine stand. The current study agreed with their finding for the first-flush needles but not the needles in subsequent flushes. The estimates of daily photosynthesis in SS saplings were lower than those of LS and MS saplings. This may result in lower amounts of reserves being stored in the stems or roots of SS saplings that have used some or most of their reserves earlier in the season. Three years after a prescribed burn, the annual diameter growth of 22-year-old loblolly pine trees with complete crown scorch was still less than trees without crown scorch (Lilieholm and Hu 1987). It would be interesting to follow the development patterns of the first two flushes and their needles in the year following the prescribed fire to detect any carry-over effects of a dormant season prescribed burn as reported by Ford and others (2010).

CONCLUSIONS

Variations in needle mass scorch levels existed as a result of dormant season prescribed fire. The negative impact from fire on longleaf sapling growth persisted for the first three flushes and their needles in saplings with all their previous year's needles scorched. Although the fourth and fifth flushes of the severely scorched saplings were similar in lengths to those of less severely scorched saplings, the amounts of stored reserves in these saplings may be reduced at the end of the fire year which in turn could affect flush growth the year after fire.

LITERATURE CITED

America's Longleaf. 2009. Range-wide conservation plan for longleaf pine. http://www.americaslongleaf.org/resources/conservation-plan. [Date accessed: July 15, 2013].

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.

Boyer, W.D. 1987. Volume growth loss: a hidden cost of periodic prescribed burning in longleaf pine? Southern Journal of Applied Forestry. 11: 154-157.

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: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 165-168.

Boyer, W.D.; Miller, J.H. 1994. Effect of burning and brush treatments on nutrient and soil physical properties in young longleaf pine stands. Forest Ecology and Management. 70: 311-318.

Brockway, D.G.; Outcalt, K.W. 1998. Gap-phase regeneration in longleaf pine wiregrass ecosystems. Forest Ecology and Management. 106: 125-139.

Brockway, D.G.; Outcalt, K.W.; Wilkins, R.N. 1998. Restoring longleaf pine wiregrass ecosystems: plant cover, diversity and biomass following low-rate hexazinone application on Florida sandhills. Forest Ecology and Management. 103: 159-175.

Bruce, D. 1951. Fire, site, and longleaf height growth. Journal of Forestry. 49: 25-27.

Chung, H-H.; Barnes, R.L. 1980. Photosynthate allocation in *Pinus taeda*. III. Photosynthate economy: its production, consumption and balance in shoots during the growing season. Canadian Journal of Forest Research. 10: 348-356.

Dickson, R.E. 1991. Assimilate distribution and storage. In: Raghavendra, A.A., ed. Physiology of trees. New York: John Wiley: 51-85.

Ericsson, A. 1978. Seasonal changes in translocation ^{14}C from different age classes of needles on 20-year-old Scots pine trees (*Pinus sylvestris*). Physiologium Plantarum. 43: 351-358.

Ford, C.R..; Minor, E.S.; Fox, G.A. 2010. Long-term effects of fire and fire-return interval on population structure and growth of longleaf pine (*Pinus palustris*). Canadian Journal of Forest Research. 40: 1410-1420.

Gordon, J.C.; Larson, P.R. 1968. Seasonal course of photosynthesis, respiration, and distribution of ^{14}C in young *Pinus resinosa* trees as related to wood formation. Plant Physiology. 43: 1617-1624.

Grelen, H.E. 1983. May burning favors survival and early height growth of longleaf pine seedlings. Southern Journal of Applied Forestry. 7: 16-20.

Gresham, C.A.; Williams, T.M.; Lipscomb, D.J. 1991. Hurricane Hugo wind damage to southeastern U.S. coastal forest tree species. Biotropica. 223: 420-426.

Haywood, J.D. 2007. Influence of herbicides and felling, fertilization, and prescribed fire on longleaf pine establishment and growth through six growing seasons. New Forests. 33: 257-279.

Haywood, J.D. 2009. Eight years of seasonal burning and herbicidal brush control influence sapling longleaf pine growth, understory vegetation, and the outcome of an ensuing wildfire. Forest Ecology and Management. 258: 295-305.

Haywood, J.D.; Grelen, H.E. 2000. Twenty years of prescribed burning influence the development of direct-seeded longleaf pine on a wet pine site in Louisiana. Southern Journal of Applied Forestry. 24: 86-92.

Johnsen, K.H.; Butnor, J.R.; Kush, J.S. [and others]. 2009. Hurricane Katrina winds damaged longleaf pine less than loblolly pine. Southern Journal of Applied Forestry. 33: 178-181.

Johnson, J.D. 1984. A rapid technique for estimating total surface of pine needles. Forest Science. 30: 913-921.

Kuhns, M.R.; Gjerstad, D.H. 1991. Distribution of ^{14}C-labeled photosynthate in loblolly pine (*Pinus taeda*) seedlings as affected by season and time after exposure. Tree Physiology. 8: 259-271.

Landers, J.L.; VanLear, D.H.; Boyer, W.D. 1995. The longleaf pine forests of the southeast: requiem or renaissance? Journal of Forestry. 93: 39-44.

Lilieholm, R.J.; Hu, S.C. 1987. Effect of crown scorch on mortality and diameter growth of 19-year-old loblolly pine. Southern Journal of Applied Forestry. 11: 209-211.

Mitchell, R.J.; Hiers, J.K.; O'Brien, J.J. [and others]. 2006. Silviculture that sustains: the nexus between silviculture, frequent prescribed fire, and conservation of biodiversity in longleaf pine forests of the southern United States. Canadian Journal of Forest Research. 36: 2724-2736.

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(2): 1-12.

Outcalt, K.W. 2000. The longleaf pine ecosystem of the south. Native Plants Journal. 1(1): 42-53.

SAS Institute Inc. 2004. SAS/STAT® 9.1 User's Guide. Cary, NC: SAS Institute Inc. 5121 pp.

Sheffield, M.C.; Gagnon, J.L.; Jack, S.B. [and others]. 2003. Phenological patterns of mature longleaf pine (*Pinus palustris* Miller) under two different soil moisture regimes. Forest Ecology and Management. 179: 157-167.

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.; Zarnoch, S.J.; Haywood, J.D. [and others]. 2013. Developmental dynamics of longleaf pine seedling flushes and needles. 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: 149-155.

Sword Sayer, M.A.; Haywood, J.D. 2006. Fine root production and carbohydrate concentrations of mature longleaf pine (*Pinus palustris* P. Mill.) as affected by season of prescribed fire and drought. Trees. 20: 165-175.

Sword Sayer, M.A.; Haywood, J.D.; Sung, S-J.S. 2009. Cavity size and copper root pruning affect production and establishment of container-grown longleaf pine seedlings. Forest Science. 55: 377-389.

van den Driessche, R. 1987. Importance of current photosynthate to new root growth in planted conifer seedlings. Canadian Journal of Forest Research. 17: 776-782.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth, and management. 1st ed. Washington, DC: Charles Lathrop Pack Forestry Foundation. 429 p. In cooperation with: The U.S. Department of Agriculture Forest Service.

FLAMMABILITY OF LITTER FROM SOUTHEASTERN TREES: A PRELIMINARY ASSESSMENT

J. Morgan Varner, Jeffrey M. Kane, Erin M. Banwell, and Jesse K. Kreye[1]

Abstract--The southeastern United States possesses a great diversity of woody species and an equally impressive history of wildland fires. Species are known to vary in their flammability, but little is known about southeastern species. We used published data and our own collections to perform standard litter flammability tests on a diverse suite of 25 native overstory trees from the region. Flame heights, duration of flaming and smoldering combustion, and fuel consumption were measured for each species. Species spanned a wide spectrum of flammability, from highly flammable to fire-impeding. The southeast has several flammable species, including several *Pinus* and *Quercus* that rank among the most flammable species ever measured. In addition to these species, several species burned with poor flammability, notably eastern hemlock [*Tsuga canadensis* (L.) Carrière], Ocala sand pine [*P. clausa* var. *clausa* (Chapm. ex Engelm.) Vasey ex Sarg.], and eastern white pine (*P. strobus* L.), dampening local fire intensity and enabling these fire-intolerant species to survive in fire-prone landscapes.

INTRODUCTION

The southeastern United States encompasses a wide variety of fire-prone ecosystems. Within this diversity of ecosystems, individual species have traits that allow them to withstand heating (e.g., thick bark), recover damaged above-ground stems (e.g., via resprouting), or link reproduction or establishment to the season or frequency of fires. Another aspect of resilience to fire is litter flammability, whereby plants cast litter that elevates local flammability (the so-called “kill thy neighbor” strategy; Bond and Midgley 1995) or that diminishes local flammability (so-called “mesophiers”; Nowacki and Abrams 2008). Highly flammable species have litter that burns with great intensity, subjecting nearby plants to high temperatures and injury while diminishing local competition. Species with low flammability diminish local fire intensity, allowing them to escape injury or dampen intensity or overall fire spread. Differential flammability has been found across many plant communities in North America (de Magalhães and Schwilk 2012, Engber and Varner 2012, Fonda 2001, Fonda and others 1998), Australia (Scarff and Westoby 2006), and Europe (Curt and others 2011). In the southeastern United States, flammability research to date has focused on a narrow suite of pines (*Pinus*, four species) and oaks (*Quercus*, eight species). This low number of species is not reflective of the high tree diversity and frequent-fire history of the region.

The focus of flammability in forests is senesced litter, the primary carrier of fire in these ecosystems. Litter flammability can be evaluated in a number of ways that quantifies its ignitability (resistance or delay to ignition), fire intensity (energy release rate, as measured by flame characteristics), sustainability (the duration of flaming and smoldering phases of combustion), consumability (the proportion of the fuel consumed), or combinations of these four factors (Fonda 2001).

Understanding interspecific flammability has implications for land management in fire-prone southeastern forests. Prescribed fires are widely used in many southeastern ecosystems (Wade and Lunsford 1989) but have variable effects. Some of these differences may be due to variation in fuel flammability and its subsequent effect on fire behavior (Kane and others 2008, Wenk and others 2011). Distinguishing the flammability of species in southeastern forests can assist managers in targeting species for removal to better allow for the reintroduction of fire to these ecosystems (Kane and others 2008).

We sought to evaluate litter flammability of 25 native tree species common to many southeastern ecosystems. The specific objectives of this study were: (1) to evaluate and compare the flammability of the species, and (2)

[1]Assistant Professor, Mississippi State University, College of Forest Resources, Department of Forestry, Mississippi State, MS 39762; Assistant Professor and Research Assistant, respectively, Humboldt State University, College of Natural Resources, Forestry and Wildland Resources, Arcata, CA 95521; and Post-doctoral Research Scientist, Mississippi State University, College of Forest Resources, Department of Forestry, Mississippi State, MS 39762.

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to evaluate and compare the flammability of oaks and pines separately.

MATERIALS AND METHODS

Study Species and Collections

We collected leaf litter from 25 native species from several locations throughout the southern United States for this study and for those published in previous work (Fonda 2001, Fonda and Varner 2004, Kane and others 2008; table 1). For each species, we collected approximately 20 g (oven-dry weight) of recently fallen litter from beneath individual trees, with eight trees sampled per species. Collected litter was placed in paper bags, transported to the laboratory, sorted (removing non-leaf litter), and stored under laboratory conditions prior to burning experiments.

Laboratory Burning Experiments

Our analysis used data from published values from two laboratories, the Western Washington University combustion facility (Fonda 2001, Fonda and Varner 2004) and the Humboldt State University Wildland Fire Lab (Kane and others 2008). Both sites used identical methods in similar facilities, consisting of a 1- by 1-m steel platform mounted beneath an adjustable fume hood. In the Humboldt State University Wildland Fire Lab, we burned 13 additional species as part of this study (table 1). For all burning experiments, a 15 g oven-dry litter sample was spread across a 25-by 25-cm lattice of eight xylene-soaked cotton strings placed on the steel platform. Between five and eight replicates were burned for all 25 species. The string ends were ignited with a standard lighter, spreading to the interior litter. Maximum flame height was estimated by two trained observers (to the nearest cm) against a ruler mounted behind the platform. The duration of flaming and smoldering combustion was measured for all burns (to the nearest second). All residues following burning were sorted to remove unburned string and then weighed to calculate fuel consumption (percent).

Data Analysis

Burning experiment data were compared using principal components analysis (PCA) with standardized values (mean = 0, standard deviation = 1) for the each of the 25 species' four measured flammability variables: flame height (cm); flaming duration (seconds); smoldering duration (seconds); and fuel consumption (percent). When outliers were detected, we repeated the analyses after excluding those species. We only considered axes in PCA that resulted in cumulatively > 75 percent of the variation explained. Factors that were most related to the generated PCA axes were reported. We ran PCA for all 25 species and separately for the conifer (14 species) and oak (11 species) datasets.

RESULTS AND DISCUSSION

All Species

The southeastern tree species burned here are comparable to species burned using similar methods elsewhere in North America. The first PCA generated a two-axis solution that comprised 80.3 percent of the variation in the four variable dataset but detected three outlier species. These three species - eastern hemlock [*Tsuga canadensis* (L.) Carrière], Ocala sand pine [*P. clausa* var. *clausa* (Chapm. ex Engelm.) Vasey ex Sarg.], and eastern white pine (*P. strobus* L.) - were the least-flammable species evaluated and excluded from subsequent analyses. The analysis of the remaining 22 species resulted in a two-axis solution that explained 79.0 percent of the dataset, with no outliers. The first axis of this PCA, explaining 51.3 percent, was positively related to flame height and fuel consumption. Species at the high extreme of the first axis included longleaf pine (*P. palustris* Mill.), loblolly pine (*P. taeda* L.), pond pine (*P. serotina* Michx.), shortleaf pine (*P. echinata* Mill.), white oak (*Q. alba* L.), and turkey oak (*Q. laevis* Walt.), while those at the low extreme included sand live oak (*Q. geminata* Small), live oak (*Q. virginiana* Mill.), laurel oak (*Q. hemisphaerica* Bartr. ex Willd.), and Virginia pine (*P. virginiana* Mill.). The second axis explained an additional 27.6 percent and was positively related to flaming and smoldering durations. The extremes of this axis included turkey oak and southern red oak (*Q. falcata* Michx.) on the protracted burning extreme and pitch pine (*P. rigida* Mill.), longleaf pine, and

Table 1--List of southeastern tree species burned in laboratory flammability experiments. Code name refers to the abbreviated species name in all figures

Species	Code name	Collection location	Data source[a]
Pinus clausa var *clausa*	Ocala sand	Ocala NF, FL	1
Pinus clausa var *immuginata*	Choct sand	Eglin Air Force Base, FL	2
Pinus echinata	shortleaf	Sterrett, AL	2
Pinus elliottii var *densa*	S Fl slash	Archbold Biol. Station, FL	1
Pinus elliottii var *elliottii*	slash	Ichauway, GA	2
Pinus glabra	spruce	Tuskegee NF, AL	2
Pinus palustris	longleaf	Ichauway, GA	2
Pinus pungens	Table Mtn	Pisgah NF, NC	2
Pinus rigida	pitch	Pisgah NF, NC	2
Pinus serotina	pond	Ocala NF, FL	3
Pinus strobus	E white	Pisgah NF, NC	2
Pinus taeda	loblolly	Ichauway, GA	2
Pinus virginiana	Virginia	Sterrett, AL	2
Quercus alba	white	Sterrett, AL	2
Quercus falcata	So red	Ichauway, GA	4
Quercus geminata	sand live	Ichauway, GA	2
Quercus hemisphaerica	laurel	Ichauway, GA	4
Quercus incana	bluejack	Ichauway, GA	4
Quercus laevis	turkey	Ichauway, GA	4
Quercus margaretta	sand post	Ichauway, GA	4
Quercus marilandica	blackjack	Sterrett, AL	2
Quercus nigra	water	Ichauway, GA	4
Quercus stellata	post	Ichauway, GA	4
Quercus virginiana	live	Ichauway, GA	4
Tsuga canadensis	E hemlock	Pisgah NF, NC	2

[a]1 = Fonda (2001); 2 = collected, unpublished data; 3 = Fonda and Varner (2004); 4 = Kane and others (2008).

loblolly pine on the brief extreme. Oaks and pines generally overlapped, each having examples of highly flammable species and those with diminished flammability (fig. 1).

Conifer Litter Flammability

The most flammable pines in this study matched or exceeded published values for pines in the western U.S. (Fonda 2001, Fonda and others 1998). As in the overall PCA, eastern hemlock was an outlier in the conifer-only PCA. The remaining 13 conifers resulted in a two-axis PCA explaining 93.4 percent of the dataset (fig. 2). The first axis of this PCA explained 57.5 percent and was most related to flame height, fuel consumption, and flaming duration. Species at the positive extreme of this axis included longleaf, loblolly, pond, shortleaf, spruce (*P. glabra* Walt.), and pitch pines, while those at the negative extreme included Ocala sand, eastern white, and Virginia pines. The second axis explained an additional 36.0 percent and was negatively related to flaming and smoldering durations. The extremes of this axis included Virginia, Choctawhatchee sand [*P. clausa* var *immuginata* (Chapm. ex Engelm.) Vasey ex Sarg.], and Table Mountain (*P. pungens* Lamb.) pines on the protracted burning extreme and Ocala sand, eastern white, pitch, longleaf, and loblolly pine on the brief extreme. Eastern hemlock burned with lower intensity than any species in North America to-date.

Oak Litter Flammability

The most- and least-flammable oaks tracked values observed for California oaks (Engber and Varner 2012). No outliers were detected in the oak-only PCA, and the two-axis solution explained 95.1 percent of the dataset (fig. 3). The first axis of this PCA explained 69.1 percent and was negatively related to flame height, fuel consumption, and flaming duration. Species at the low-flammability extreme of this axis included live and sand live oaks, while those at the high-flammability end included turkey, southern red, and white oak. The second axis

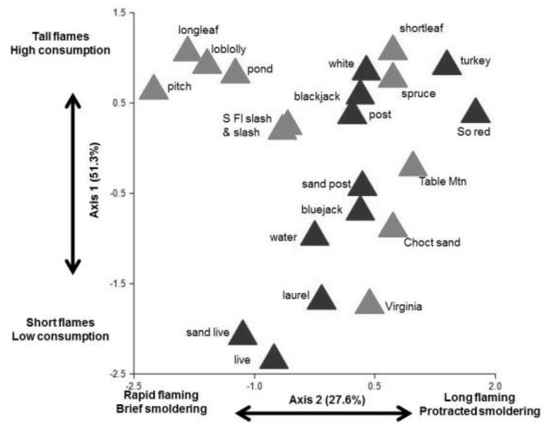

Figure 1--Principal components analysis of 22 southeastern tree species across litter flammability axes. Oaks are denoted with dark gray triangles and their common names. Pines are denoted with light gray triangles and their corresponding common names.

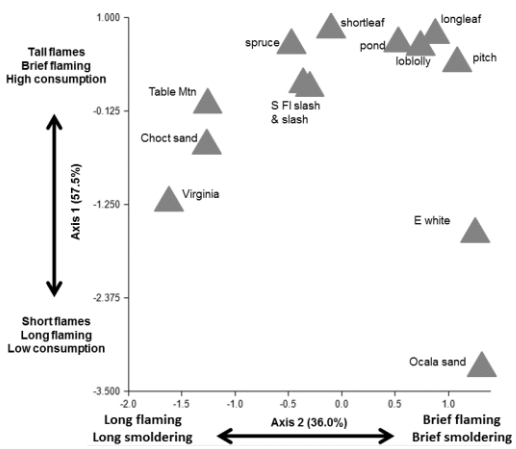

Figure 2--Principal components analysis of 13 southeastern pines across litter flammability axes.

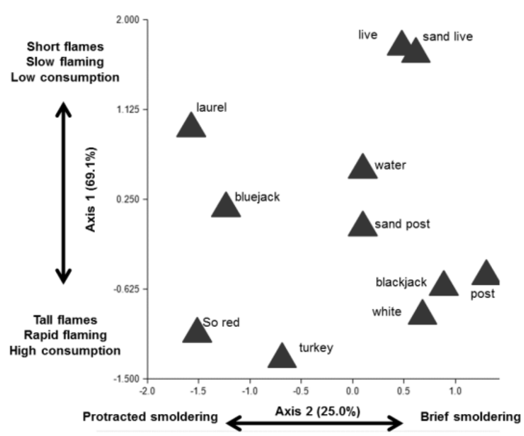

Figure 3--Principal components analysis of 13 southeastern oaks across litter flammability gradients.

explained an additional 25.0 percent and was negatively related to smoldering duration. The brief smoldering end of this axis included post oak (*Q. stellata* Wangenh.), blackjack oak (*Q. marilandica* Münchh.), and white oak, and the protracted smoldering extreme included southern red, laurel, and bluejack oak (*Q. incana* Bartr.).

Fire Management Implications

These results reveal that many southeastern trees cast highly flammable litter capable of facilitating fire spread and several that impede fire. Those highly flammable species facilitate the maintenance of fire-stable ecosystems, killing invading species that dampen flammability and decrease ecosystem integrity (Nowacki and Abrams 2006). Those species with low flammability are capable of diminishing fire intensity, enabling their persistence and the survival and persistence of other fire-impeding species that can degrade fire-prone plant communities. Future flammability work that scales these results to field fire behavior (e.g., Wenk and others 2011) will provide a stronger understanding of the consequences of fire or its exclusion on southeastern ecosystems.

CONCLUSIONS

Litter from the tree species burned in this study revealed that many southeastern species, both pines and several oaks, are highly flammable. These flammable species fuel fires that maintain open pine-oak forests and woodlands. Several species also burned with low intensity, perhaps enabling these fire-sensitive species to persist in these landscapes.

ACKNOWLEDGMENTS

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LITERATURE CITED

Bond, W.J.; Midgley, J.J. 1995. Kill thy neighbour: an individualistic argument for the evolution of flammability. Oikos. 73: 79-85.

Curt, T.; Schaffhauser, A.; Borgniet, L. [and others]. 2011. Litter flammability in oak woodlands and shrublands of southeastern France. Forest Ecology and Management. 261: 2214-2222.

de Magalhães, R.M.Q.; Schwilk, D.W. 2012. Leaf traits and litter flammability: evidence for non- additive mixture effects in a temperate forest. Journal of Ecology. 100: 1153-1163.

Engber, E.A.; Varner, J.M. 2012. Patterns of flammability of the California oaks: the role of leaf traits. Canadian Journal of Forest Research. 42: 1965-1975.

Fonda, R.W. 2001. Burning characteristics of needles from eight pine species. Forest Science. 47: 390-396.

Fonda, R.W.; Belanger, L.A.; Burley, L.L. 1998. Burning characteristics of western conifer needles. Northwest Science. 72: 1-9.

Fonda, R.; Varner, J.M. 2004. Burning characteristics of cones from eight pine species. Northwest Science. 78: 322-333.

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.

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the eastern United States. BioScience. 58: 123-138.

Scarff, F.R.; Westoby, M. 2006. Leaf litter flammability in some semi- arid Australian woodlands. Functional Ecology. 20: 745-752.

Wade, D.D.; Lunsford, J.D. 1989. A guide for prescribed fire in southern forests. Tech. Pub. R8-TP-11. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region. 56 p.

Wenk, E.S.; Wang, G.G.; Walker, J.L. 2011. Within-stand variation in understorey vegetation affects fire behaviour in longleaf pine xeric sandhills. International Journal of Wildland Fire. 20: 866-875.

Eco-physiology

Moderators:

Chris Maier
USDA Forest Service
Southern Research Station

and

Rodney Will
Oklahoma State University
Natural Resource Ecology and Management

ESTABLISHMENT PATTERNS OF WATER-ELM AT CATAHOULA LAKE, LOUISIANA

Karen S. Doerr, Sanjeev Joshi, and Richard F. Keim[1]

Abstract--At Catahoula Lake in central Louisiana, an internationally important lake for water fowl, hydrologic alterations to the surrounding rivers and the lake itself have led to an expansion of water-elm (*Planera aquatic* J.F. Gmel.) into the lake bed. In this study, we used dendrochronology and aerial photography to quantify the expansion of water-elm in the lake and identify patterns of expansion. Our data suggest woody vegetation encroached into Catahoula Lake by about 10.4 km^2 (30 percent of the lake area) between 1940 and 2007. Encroachment has been concentrated in the northeast near the connection to Dry River, in the southwest at the input of Little River, and to a lesser extent south of the Diversion Canal on the eastern side of the lake. Woody vegetation is encroaching on the lake in three patterns. The first is a pattern of continuous encroachment, which occurred in 50 percent of our transects. The second pattern is a long-term stable pattern (no encroachment; younger and older trees intermingling), which occurred in 25 percent of our transects. The third pattern is complex with no discernable trend and is complicated by attempts to manage the woody expansion. The reasons for expansion are not well understood, and recently, increased rates have been manifest in multiple modes of establishment.

INTRODUCTION

Catahoula Lake is a backswamp lake in the floodplain of the Black River that is seasonally inundated and palustrine. Lake levels fluctuate about 6 m each year, going almost completely dry beginning in July to about November (USGS 2002). This annual variation promotes growth of wetland plants like sprangletop [*Leptochloa fusca* (L.) Kunth ssp. *fascicularis* (Lam.) N. Snow], Walter's millet [*Echinochloa walteri* (Pursh) A. Heller], and yellow nutsedge (*Cyperus esculentus* L.) which are sources of carbohydrates for wintering waterfowl and migratory birds using the lake as a stop along their route (Bruser 1995). The variability of Catahoula Lake water levels has been long known. Tedford (2009) found that lake water levels have been fluctuating at least since 4,000 years BP.

Hydrologic alterations to Catahoula Lake and the surrounding rivers have been occurring since at least the early 1920s. Catahoula Lake receives inputs from the headwaters of the Little River and backwater flooding from the Mississippi, Ouachita, Black, and Red rivers and drains through a diversion canal that runs from near French Fork to the Black River (Fig. 1) (Tedford 2009). Before the canal was built, the lake drained through the French Fork of Little River. A series of locks and dams were built first in 1926 and again in 1972 on the Black River to allow for navigation. These adjustments raised the level of the Black River and would have flooded Catahoula Lake permanently (Sessums 1954). The sole drainage through French Fork of the Little River was no longer adequate. To allow the natural, seasonal drawdown, the Corps of Engineers built a diversion canal which runs from near French Fork to the Black River. Today water is managed through opening and closing the water control structure on the diversion canal (Saucier 1998).

There has been great concern over the last few decades that Catahoula Lake is experiencing a shift from wetland vegetation toward woody plants on the lake bed which outcompete the herbaceous vegetation. We do not have complete information about historic management of the lake, but historic aerial imagery suggests woody expansion has been occurring since 1952. Today managers mow, burn, and herbicide to eliminate young water-elm (*Planera aquatic* J.F. Gmel.) and swamp privet [*Foresteria acuminata* (Michx.) Poir.] on the bed of the lake. These techniques are both costly and time consuming. Our goal is to quantify the amount and rate of woody expansion into the lake and better understand patterns of

[1]Graduate Research Assistant, Louisiana State University Agricultural Center, School of Renewable Natural Resources, Baton Rouge, LA 70803; Graduate Research Assistant, University of South Carolina, Department of Geography, Columbia, SC 29208; and Associate Professor, Louisiana State University Agricultural Center, School of Renewable Natural Resources, Baton Rouge, LA 70803.

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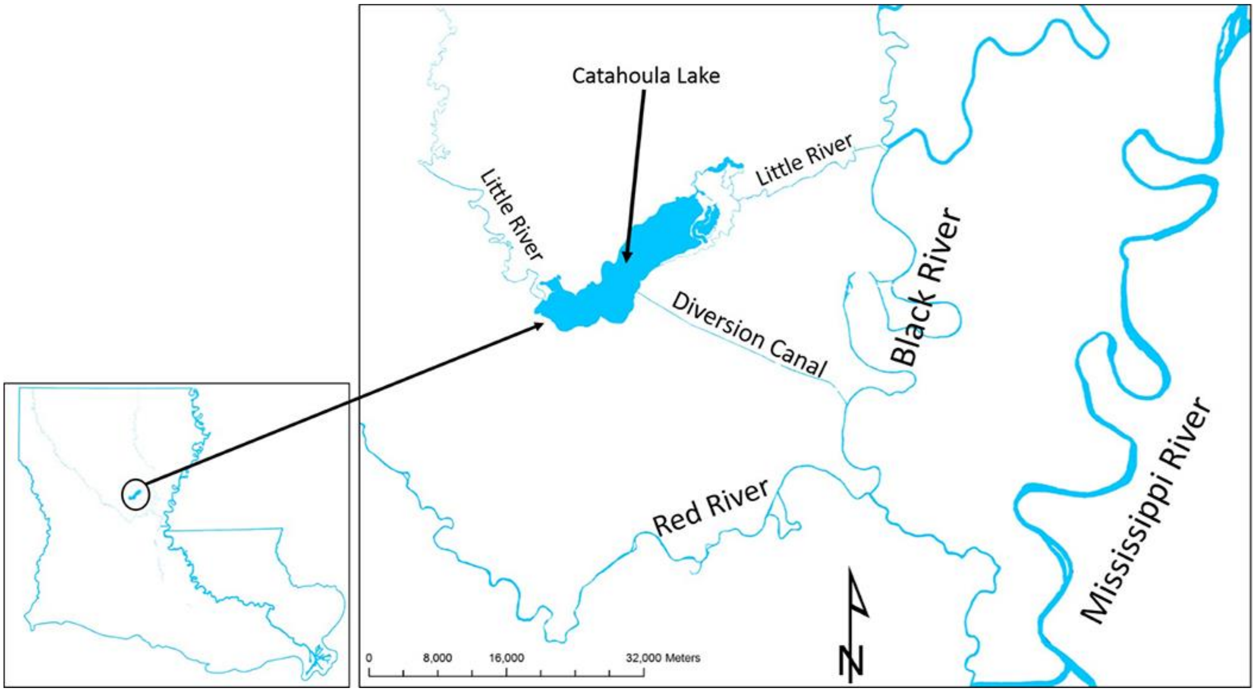

Figure 1—Location of Catahoula Lake in relation to its inputs and outputs: the Little, Red, Black, and Mississippi rivers and the diversion canal.

expansion. We compared aerial imagery to tree ages obtained from tree rings.

METHODS

Aerial Photo Interpretation

We examined USDA aerial imagery from 1940, 1966, and 2007 and Wills' (1963) map of plant types to quantify the extent and amount of expansion of woody vegetation. Imagery was selected by considering time of year, age, resolution, and water level. We scanned the 1940 and 1966 imagery and imported it into ARCMap 9.3.1 where it was georeferenced to the 2007 USDA imagery from the National Agriculture Imagery Program (NAIP). The imagery was then rectified to the Universal Transverse Mercator (UTM) coordinate system and georeferenced in ARCMap. We also scanned and georeferenced Wills' (1963) vegetation map. We analyzed the aerial imagery from 1940, 1966, and 2007 using air photo interpretation and ARCMap. We used Paine's (1981) density classifications to delineate polygons of existing woody vegetation from 20 to 100 percent cover. The resulting polygons were evaluated for expansion and removal of woody vegetation between years.

Tree Ring Data

We established 7 transect lines across the study area to determine tree ages. Five of these transects were approximately the same as those of Wills (1963). The remaining two transects had nearly the same location and azimuth as those of Bruser (1995). Transects varied in length and the number of points. Sampling points were established at intervals along each transect line in the water-elm zone to determine tree ages. We used the point-quarter sampling method to sample trees, selecting the closest tree ≥ 5 cm in diameter at breast height (d.b.h.) in each quadrant within 10 m from the origin. If no trees ≥ 5 cm d.b.h. were present within 10 m, a sample was not taken in that quadrant. We collected cross-sections from the largest stem of the tree as close to the base as possible with a chain saw. For trees > 30 cm d.b.h., we collected two increment cores per tree instead. Cores and cross-sections were dried at 50 ºC and sanded to 600 grit until cells were clearly visible under the microscope. Age of some partially-decayed trees was calculated by estimating the number of rings in the decayed section.

RESULTS

Aerial imagery indicated woody vegetation is expanding into the lakebed, and the rate of expansion has increased since 1966. Between 1940 and 1966, about 1.6 km^2 of woody expansion occurred. We calculated a rate of expansion of 0.17 percent per year from 1940-1966. Between 1966 and 2007, we found that the rate of expansion increased to 0.44 percent per year. Between 1940 and 2007, the lake experienced expansion of 10.4 km^2 or 30 percent (Fig. 2).

Using the tree ring data, we found 35 pre-modification trees and 160 post-modification trees. About 73 percent of the trees sampled were established after the canal was built. However, the oldest trees were 131-years-old, pre-dating any hydrologic changes to the lake.

We observed three patterns of expansion. In the first pattern, younger trees were found at the encroaching margin and older trees behind. This pattern was evident in half of the transects. This indicates a pattern of continuous encroachment (Fig. 3). The second pattern we observed was one of long-term stability (Fig. 4). This represented 25 percent of transects. Here we found young trees along the edge and filling in behind. The final scenario, accounting for 25 percent of transects, was less predictable. We found younger trees filling in behind an advancing front and sites that were complex with no discernible pattern (Fig. 5).

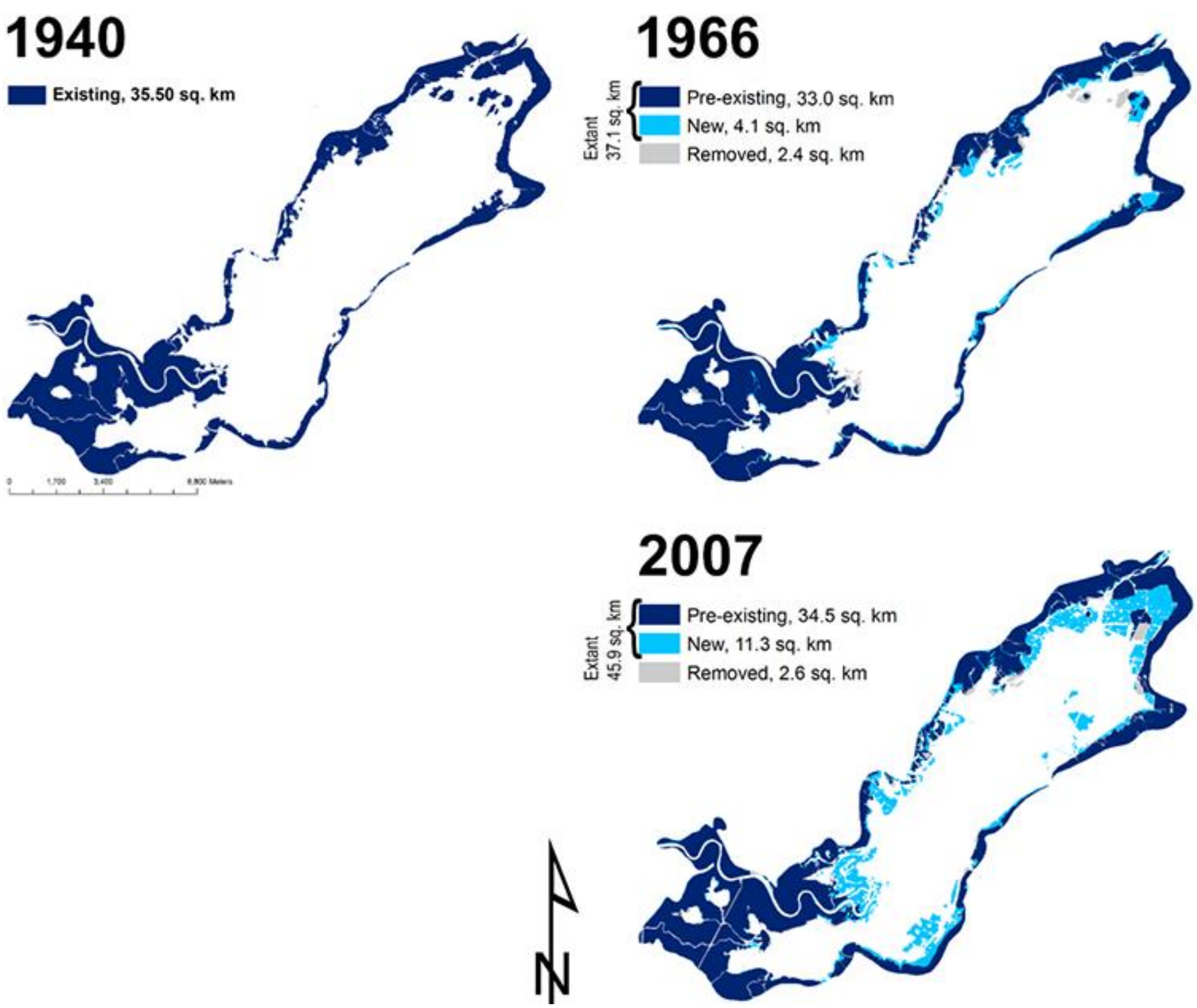

Figure 2—Expansion of woody vegetation at Cathoula Lake from 1940 to 2007.

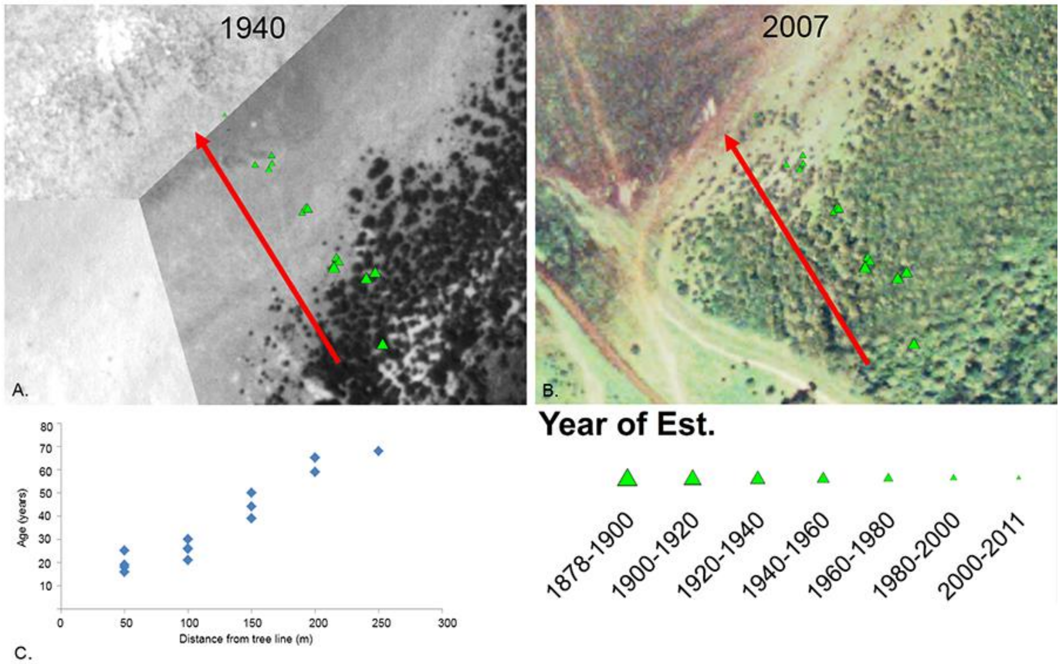

Figure 3—Example of continuous encroachment of woody vegetation. Triangles show location of sampled trees. Red arrows denote lake center. (A) USDA aerial imagery from 1940, (B) USDA aerial imagery from 2007, and (C) age of trees in relation to distance from tree line.

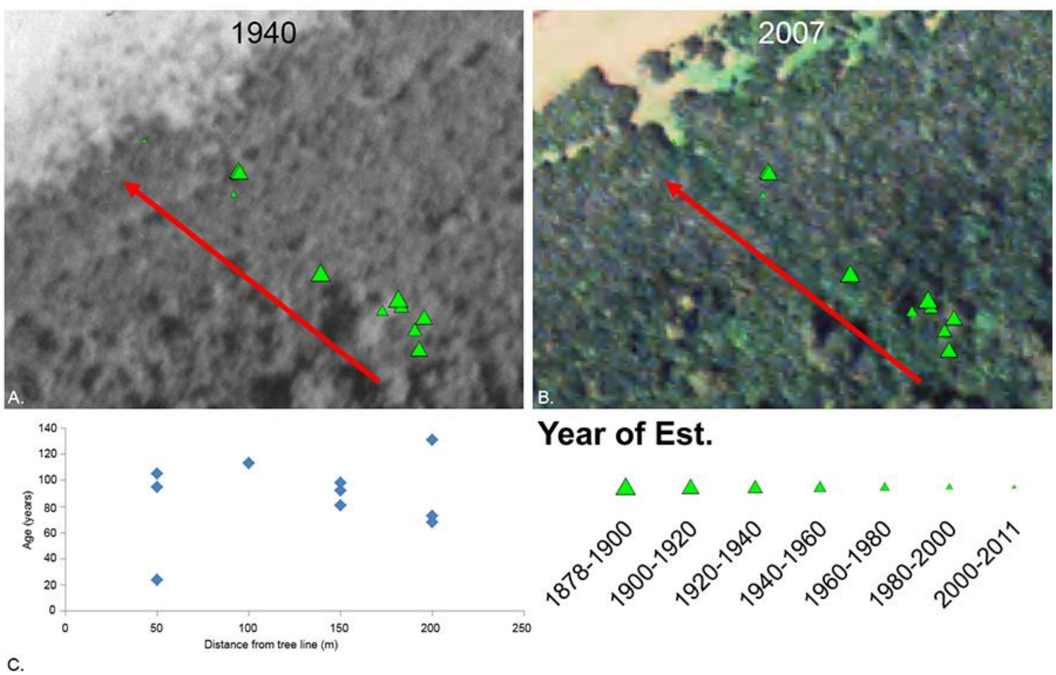

Figure 4—Example of long-term stability of woody vegetation extent. Triangles show location of sampled trees. Red arrows denote lake center. (A) USDA aerial imagery from 1940, (B) USDA aerial imagery from 2007, and (C) age of trees in relation to distance from tree line.

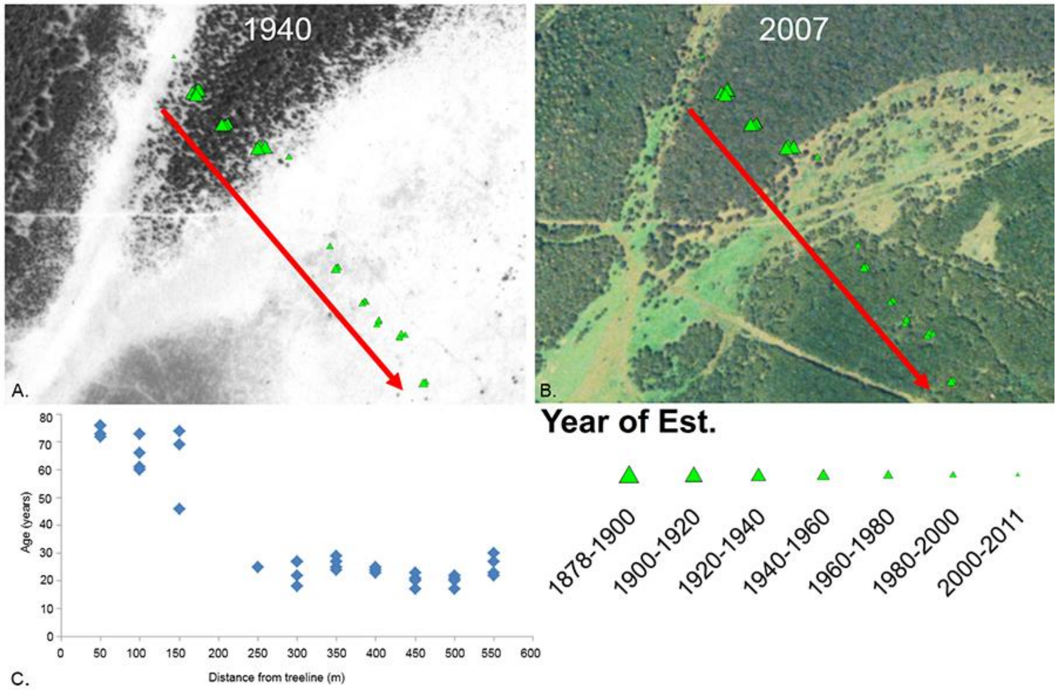

Figure 5—Example of complex pattern of encroachment of woody vegetation. Triangles show location of sampled trees. Red arrows denote lake center. (A) USDA aerial imagery from 1940, (B) USDA aerial imagery from 2007, and (C) age of trees in relation to distance from tree line.

DISCUSSION

We did not investigate soil conditions or possible changes caused by any sediment deposition. If there is substantial sedimentation, it is possible that location and amount of sediment correlates to locations of expansion. It is also possible that sediment composition varies spatially within the lake and helps control establishment of woody species. One complicating factor is that wave action is important for maintaining lake bottom morphology (Brown 1943) which would mute spatial patterns of lake-bottom sediments.

Historic variability in lake chemistry is not well understood. However, there was extensive salt-water pollution from petroleum extraction in the lake itself and upstream in the watershed of the Little River, such that Wills (1963) reported salinity at times of low water approaching that of sea water. Salinity stress would obviously affect plant species composition and may have disrupted normal successsional pathways and edaphic relationships for some period of time.

It remains generally poorly understood why Catahoua Lake is an open body of water and not a forested wetland. Some theories suggest that human disturbance is at least partially responsible for this condition. For example, grazing by cattle and feral hogs has sometimes been cited as a reason Catahoula Lake has remained unforested (Willis 2009). However, Tedford (2009) showed that ecologic conditions (caused by fluctuating water levels) have been similar for the past 4,000 years. Thus, there has likely been a recent change in some main natural process causing the ecosystem to cross a threshold toward forest occupation.

CONCLUSIONS

Woody vegetation has been encroaching into the formerly herbaceous lake bottom at Catahoula Lake. The rate of expansion is apparently faster now than prior to major hydrologic modifications at the lake, which are dominated by the establishment of deeper water on the Black River and establishment of the diversion canal to manage hydrology of the lake. Patterns of establishment do not allow for clear interpretation of the major processes controlling extent of water-elm, so further work is needed to understand the linkage between hydrologic modifications and the ecosystem at the lake margin.

LITERATURE CITED

Brown, C.A. 1943. Vegetation and lake level correlations at Catahoula Lake, Louisiana. The Geographical Review. 33: 435-445.

Bruser, J.F. 1995. Ecology of Catahoula Lake plant communities in relation to an anthropogenic water regime. Baton Rouge: Louisiana State University. 100 p. M.S. thesis.

Paine, D.P. 1981. Aerial photography and image interpretation for resource management. New York: Wiley. 571 p.

Saucier, M. 1998. Water resources development in Louisiana 1998. New Orleans District: U.S. Army Corps of Engineers. 177 p.

Sessums, R.T. 1954. Catahoula Lake area report. Presented to Governor Robert F. Kennon and the Louisiana Legislature in compliance with Act no. 598, 1952 Legislature. Baton Rouge: Louisiana Department of Public Works. 34 p.

Tedford, R.A. 2009. A multi-proxy approach to investigating the latest Holocene (~4,500 yrs. BP) vegetational history of Catahoula Lake, Louisiana. Baton Rouge: Louisiana State University. 391 p. Ph.D. dissertation.

USGS [U.S. Geological Survey]. 2002. Modeling the bathymetry of Catahoula Lake: specialized technology for wetland management. Fact Sheet 098-02. Lafayette, LA: U.S. Dept. of the Interior, U.S. Geological Survey. 2 p.

Willis, F.L. 2009. A multidisciplinary approach for determining the extents of the beds of complex natural lakes in Louisiana. New Orleans: University of New Orleans. 190 p. Ph.D. dissertation.

Wills, D.W. 1963. An investigation of some factors affecting waterfowl and waterfowl habitat on Catahoula Lake, Louisiana. Baton Rouge, LA: Louisiana State University. 164 p. M.S. thesis.

IMPACTS OF INTENSIVE MANAGEMENT AND GENETIC IMPROVEMENT ON SOIL CO_2 EFFLUX AND CARBON CYCLING IN MANAGED LOBLOLLY PINE FORESTS

Chelsea G. Drum, Eric J. Jokela, Jason G. Vogel, Edward A. G. Schuur, and Salvador Gezan[1]

In the southeastern United States, fertilization and weed control treatments, with deployment of genetically improved seedlings planting stock, are routinely used to increase aboveground productivity (Jokela and others 2004). The sustainability of forest productivity under these silvicultural regimes relies, to a certain extent, on the rate of the microbial conversion of plant litter to soil organic matter (SOM) and SOM to carbon dioxide (CO_2). In managed loblolly pine (*Pinus taeda* L.) forests, nearly 50 percent of the carbon (C) is found in the soil to a depth of 1 m, and previous research has indicated that this C can increase with fertilization (Vogel and others 2011). However, in managed forests, SOM dynamics are still unclear (Jandl and others 2007). While we would anticipate an increase in SOM with increasing aboveground productivity, alterations in plant C allocation, understory C inputs, and microbial response to fertilization could lead to unpredicted results (Vogel and others 2011).

This project examined the effects of intensive management and genetic selection of loblolly pine on soil CO_2 efflux and C cycling. In Florida, two field installations at Gainesville, FL (Site A) and Sanderson, FL (Site B) of two families of loblolly pine, one “fast” and one “slow” grower, were studied in a replicated, family block design with two levels of nitrogen and phosphorus fertilization, high and low culture (table 1). Measurements of root biomass and repeated measurements of forest growth, soil CO_2 efflux, and litterfall were used to determine C allocation patterns. Soil CO_2 efflux and litterfall measurements were used to estimate Total Belowground Carbon Flux (TBCF), an estimate of C allocation to roots.

Table 1--Cumulative element nutrient application rates for the high- and low-culture treatments at two sites in north Florida through 2008

Location	Culture	Nitrogen	Phosphorous
		------*pounds per acre*----	
Site A	High	400	90
	Low	45	54
Site B	High	680	160
	Low	196	70

The high culture treatment and family had a significant ($p < 0.01$) effect on loblolly pine mean aboveground biomass increment at both sites, with the fertilization effect being nearly 2 times greater than the family effect (fig. 1). The high culture significantly ($p < 0.05$) decreased soil CO_2 efflux, fine root biomass, and TBCF in three of the four families studied. Litterfall was significantly increased by greater fertilization, and the family effect was significant. The family x fertilization effect for TBCF was significant at Site B, indicating that one family did not significantly decrease TBCF in response to the heavier fertilization treatment. Results from these studies suggest that, with increasing levels of fertilization, belowground allocation and likely C inputs to the soil were reduced.

[1]Graduate Student and Professor, respectively, University of Florida, School of Forest Resources and Conservation, Gainesville, FL 32611; Assistant Professor, Texas A&M University, Department of Ecosystem Science and Management, College Station, TX 77843; Associate Professor, University of Florida, Department of Biology, Gainesville, FL 32611; and Assistant Professor, University of Florida, School of Forest Resources and Conservation, Gainesville, FL 32611.

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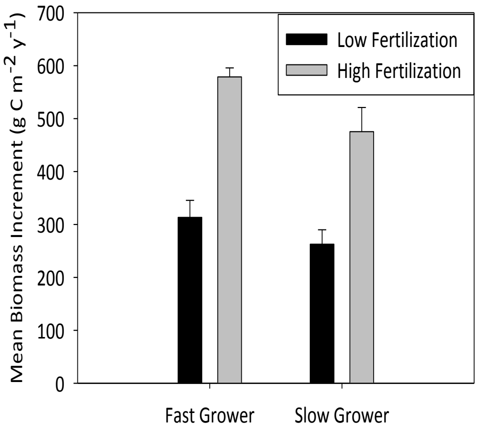

Figure 1--Aboveground mean biomass increment (MBI; g C $m^{-2}year^{-1}$) at Site A. High fertilization and family had a significant ($p < 0.01$) effect on MBI, with the fertilization effect nearly two times greater than the family effect.

LITERATURE CITED

Jandl, R.; Lindner, M.; Vesterdal, L. [and others]. 2007. How strongly can forest management influence soil carbon sequestration? Geoderma. 137: 253-268.

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.

Vogel, J.G.; Suau, L.J.; Martin, T.A.; Jokela, E.J. 2011. Long-term effects of weed control and fertilization on the carbon and nitrogen pools of a slash and loblolly pine forest in north-central Florida. Canadian Journal of Forest Research. 41: 552-567.

GROUNDSTORY VEGETATION RESPONSE TO DIFFERENT THINNING INTENSITIES IN A MINOR STREAM BOTTOM IN MISSISSIPPI: A PRELIMINARY ANALYSIS

Brent R. Frey and Ellen M. Boerger[1]

Abstract--Groundstory vegetation typically accounts for the greatest proportion of plant diversity in temperate forests, representing a critical structural component and mediating numerous ecosystem processes, including tree regeneration. The effects of thinning on groundstory vegetation have received limited study in bottomland hardwood stands. This study investigated groundstory vascular plant development following thinning in a minor stream bottom in Mississippi. Thinning treatments representing a range of residual basal areas were used to assess groundstory herbaceous and woody vine and shrub response to canopy opening. Plant community responses were evaluated in terms of cover, relative abundance, and composition. Groundstory species richness and cover were higher in thinned areas 5 years post-thinning. This was primarily attributable to increases in the cover of grasses, sedges, blackberry (*Rubus argutus* Link), and numerous forbs, likely in response to higher light availability that would favor these less shade tolerant species. Overall, cover appears to increase in direct proportion to the intensity of overstory removal. Improved knowledge of groundstory response to thinning in bottomland hardwood stands should assist management efforts aimed at the maintenance of plant diversity (and its many benefits) and at successfully regenerating desirable hardwood species.

INTRODUCTION

The groundstory, the low-statured non-tree vegetation composed of forbs, graminoids, woody vines and shrubs, is an important structural component of forested ecosystems. Indeed, the majority of plant species diversity in eastern temperate deciduous forests occurs in the groundstory, primarily in the herbaceous layer (Whigham 2004). Moreover, the groundstory mediates key processes, including provision of wildlife habitat (browse, cover), soil dynamics (erosion, sedimentation, nutrient availability), and invasive plant establishment, in addition to its well-recognized competitive effects on tree regeneration (Gilliam 2007). In mature bottomland hardwood forest types, like other closed canopy hardwood stands of the eastern temperate region, understory light tends to be low (Aikens and others 2007, Duguid and others 2013, Jenkins and Chambers 1989), and frequent short-term inundation or prolonged flooding likely limit establishment and development of vegetation. Nonetheless, in bottomland hardwoods systems in the southeastern United States, species diversity has been noted to be exceptionally high. Crouch and Golden (1997) identified over 450 vascular plants of 293 genera and 111 different families across a bottomland topographic sequence in western Alabama.

Forest stand management practices, such as thinning, may have significant impacts on groundstory development in bottomland stands. Canopy removal results in increased light availability to the understory and, in bottomland stands, has been shown to be directly related to degree of canopy removal (Jenkins and Chambers 1989). With increased light availability, an increase in groundstory cover typically occurs, even where canopy openings are small (Castleberry and others 2000, Crouch and Golden 1997). An increase in species richness is also frequently evident following canopy disturbance (Jenkins and Parker 2000, Rapp and others 2001). This is in part because high resource conditions allow opportunities for early successional, shade-intolerant species to establish along with resident understory species (Crouch and Golden 1997). At the same time, high resource conditions prevailing after canopy disturbance may drive reductions in species richness, due to the dominance of a few, vigorous early successional species (Jenkins and Parker 2000). Additionally, ground disturbance associated with logging activities may alter plant communities and establishment conditions, in some cases diminishing diversity (Castleberry and others 2000). Canopy and ground disturbance may also favor invasive species, by providing opportunities for expansion or colonization (e.g. Rosen and others 2006, Ruzicka and others 2010).

While groundstory vegetation typically accounts for the greatest proportion of plant diversity in

[1]Assistant Professor and Graduate Assistant, respectively, Mississippi State University, Department of Forestry, Mississippi State, MS 39762.

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temperate forests, the effects of canopy opening on groundstory vegetation have received only limited study in bottomland hardwood stands (Castleberry and others 2000, Crouch and Golden 1997, Jenkins and Parker 2000, Rapp and others 2001). Moreover, we are unaware of any published literature on the effects of thinning on groundstory response in bottomland systems. The objective of this study was to investigate groundstory vascular plant development 5 years after varying intensities of thinning were applied to a mature hardwood stand within a minor stream bottom in east-central Mississippi.

METHODS

Site Description

The study site is located on the Noxubee River floodplain within the Samuel D. Hamilton Noxubee National Wildlife Refuge in Noxubee County, in east-central Mississippi. The study site has been previously described by Meadows and Skojac (2012). Briefly, the site is a flat, minor stream bottom adjacent to the Noxubee River, which is subject to short-duration flooding events, primarily in winter and spring. The soils are predominantly of the Urbo silty clay loam series (fine, mixed, active, acid, thermic Vertic Epiaquept). Average site indices (base age 50) are 100 feet for cherrybark oak (*Quercus pagoda* Raf.), 96 feet for water oak (*Q. nigra* L.), and 90 feet for sweetgum (*Liquidamber styraciflua* L.). The site supports an approximately 70-year-old mixed species hardwood stand, dominated by red oaks (*Quercus* spp.) and sweetgum. The red oak species [*Q. pagoda*, *Q. nigra*, and to lesser extent *Q. phellos* L. (willow oak)] represented 51 percent of stand basal area prior to treatment. Sweetgum comprised 23 percent of stand basal area prior to treatment, with hickory (*Carya* spp.), green ash (*Fraxinus pennsylvanica* Marsh.), swamp chestnut oak (*Q. michauxii* Nutt.), overcup oak (*Q. lyrata* Walt.), and American elm (*Ulmus americana* L.) comprising the remaining basal area (Meadows and Skojac 2012).

Experimental Design

The study area covers 30 acres, divided into 2-acre rectangular treatment units, each 264 feet by 330 feet. Four thinning treatments and a control were applied in a randomized complete block design, with each treatment replicated three times. Thinning treatments were applied in October 2007. Thinning treatments were based on a stand quality management approach (Meadows and Skojac 2012), which selects trees for retention based on species and stem quality, with little regard for uniformity of spacing that typically dictates thinning treatments in plantations or simple species mixtures. The four thinning approaches differed in their retention levels based on the quality of sawlogs retained (preferred only, or preferred, desirable, and acceptable) and whether they retained pole sized trees that were potential sawlogs in the future (preferred poles or no poles). As a consequence, the different thinning treatments produced varying levels of residual basal area, ranging from 48 to 69 square feet per acre in thinned areas to 108 square feet per acre in the control (table 1).

Table 1--Pre- and post-treatment basal area (BA) by thinning intensity 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Table modified from Meadows and Skojac (2012)

Treatment	Pre-treatment BA	Residual BA
	------------*feet*^{2}*/acre*---------	
Control	113	108
Acceptable/superior	122	69
Acceptable/no pole	108	61
Desirable/superior	126	57
Desirable/no pole	112	48

Measurements and Data Analysis

We used a series of approximately 1/4000th acre (1m^2, 10.76 square feet) plots to measure groundstory vegetation in the summer of 2012 (fifth year post-harvest). There were on average eight vegetation plots per treatment unit. Plots were grouped in sets of four, positioned in four cardinal directions 19.7 feet from a central point located halfway between the bole and dripline of a residual overstory red oak tree. Plots were located in this manner to capture the range of variation in the residual stand. All non-tree herbaceous and woody plants (forbs, graminoids, woody vines, and shrubs) that occurred in the plot were identified by species (or species groups for grasses, sedges, and several genera) and percent cover. For analyses, we assessed rank abundance, estimated species richness using EstimateS Software Version 8.2 (Colwell 2012), and evaluated cover between thinning treatments. Analysis of variance (ANOVA) was performed using the PROC GLM procedure in SAS 9.2 (SAS Institute Inc., Cary, NC) to assess treatment effects on cover and species richness. Tukey's HSD test was used for multiple comparisons. Due to the limited replication, we used $\alpha = 0.10$ for evaluation of significance.

RESULTS AND DISCUSSION

Across the study site, plant abundance reflected the common pattern seen in most plant communities, with relatively few dominant species comprising the majority of groundstory cover and most other species exhibiting low abundance. Across the site, grasses were the most dominant, followed by sedges, blackberry (*Rubus argutus* Link), greenbriar (*Smilax* spp.), crossvine (*Bignonia capreolata* L.), grape (*Vitus* spp.), switchcane [*Arundinaria gigantea* (Walter) Muhl.], and poison ivy [*Toxicodendron radicans* (L.) Kuntze] (fig. 1).

Thinning treatments generally supported greater species richness than the control (figs. 2 and 3). Gap creation generally favors increased species richness, by providing higher resources (light, belowground resources) and a greater range of habitat conditions for shade-intolerant vegetation to establish along with resident vegetation (Crouch and Golden 1997, Rapp and others 2001). The highest intensity thinning (which removed a greater amount of basal area) did not favor higher species richness. This may support the findings of other studies that more intense canopy disturbance may facilitate dominance by a small number of vigorous colonizers, which may limit or exclude other species and reduce species richness (Jenkins and Parker 2000). Alternatively, it may suggest that greater logging disturbance associated with increased canopy removal may have limited species richness (Castleberry and others 2000) relative to the control. Overall, species numbers were lower than observed in several other studies (e.g. Crouch and Golden 1997, Rapp and others 2001); however, our study represents a much narrower range of site conditions. Additionally, we did not identify a number of groups to the species level (grasses, sedges, and several genera) so species richness is undoubtedly higher than estimated.

Treatment effects on cover showed a similar trend to species richness, with higher overall cover in all thinning treatments (fig. 4). Canopy openings tend to favor significant increases in groundstory cover due to enhanced light availability, which typically limits plant development under a closed canopy (Castleberry and others 2000, Jenkins and Parker 2000). This increase appeared to be attributable not only to increases in the cover of woody vines, shrubs, and graminoids following thinning, but also a substantial increase in the cover of forbs (table 2). More specifically, it was evident that grasses, sedges, and blackberry were large contributors to increased cover post-thinning. These species are particularly favored by increased light availability. Ruderal species such as smallspike false nettle [*Boehmeria cylindrica* (L.) Sw.] and other forbs also contributed to increased cover with canopy opening, as evidenced from studies in other bottomland systems (e.g., Rapp and others 2001). The availability of a number of key food plants for wildlife species (e.g., smooth ticktrefoil [*Desmodium laevigatum* (Nutt.) DC.], blackberry) was higher in the thinning treatments (table 2), while several other wildlife species were not (e.g., grape, greenbriar). Also of note, the invasive Japanese honeysuckle (*Lonicera japonica* Thunb.) did not develop higher cover in thinning treatments, despite its recognition as a colonizer of disturbed bottomland sites (Ruzicka and others 2010). Overall, cover increased in linear fashion in relation to residual basal area across the thinning treatments (fig. 5).

SUMMARY

Groundstory species richness and cover were higher in thinned areas 5 years post-thinning. This was primarily attributable to increases in the cover of grasses, sedges, blackberry, and numerous forbs, likely in response to higher light availability that would favor these less shade tolerant species. Wildlife would likely benefit

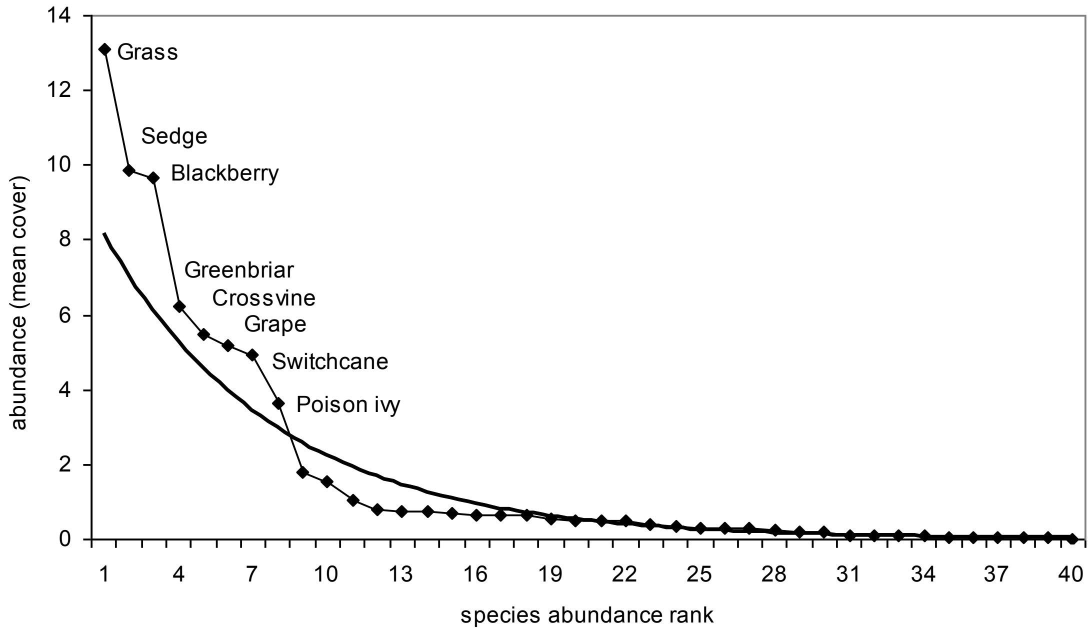

Figure 1--Species abundance rank based upon percent cover 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Dominant species are indicated.

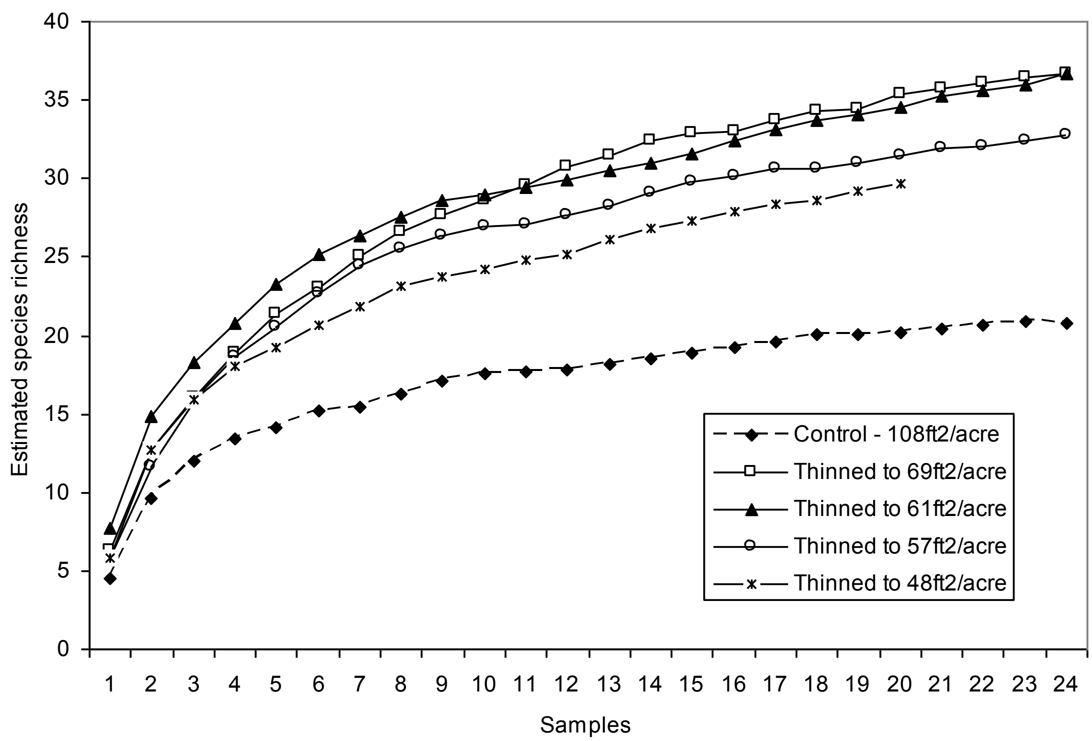

Figure 2--Species richness among thinning treatments 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Richness estimates based on Estimates species richness software (Colwell 2012).

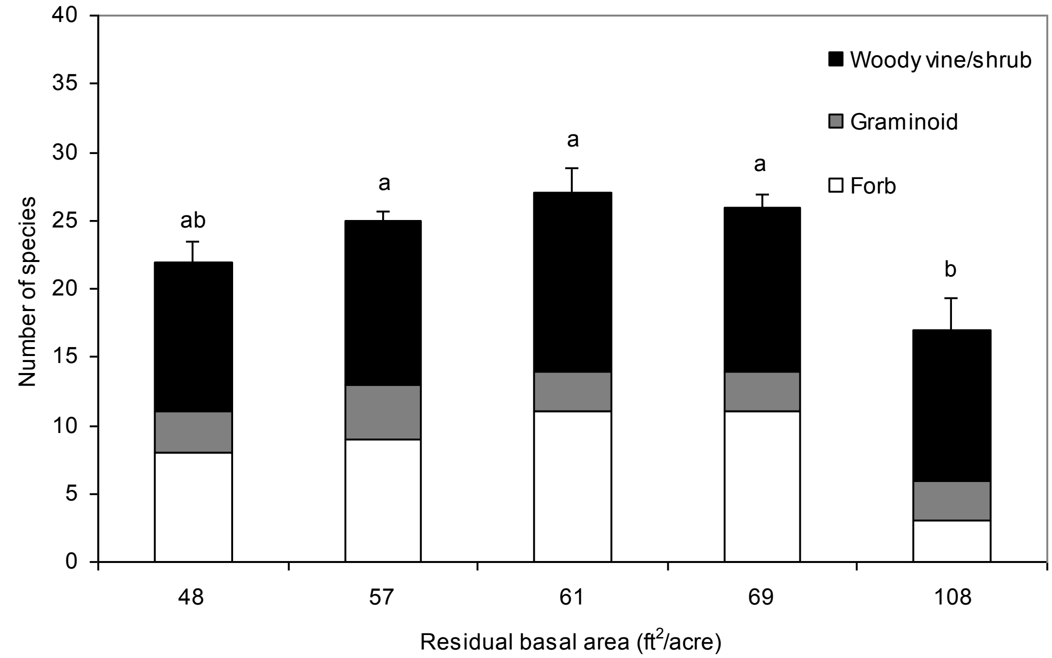

Figure 3--Species richness by treatment for the main groups of groundstory plants (forb, graminoid, woody vine/shrub) 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Significance was analyzed using Tukey's HSD test on the total number of species; columns sharing different letters are significantly different (P < 0.10).

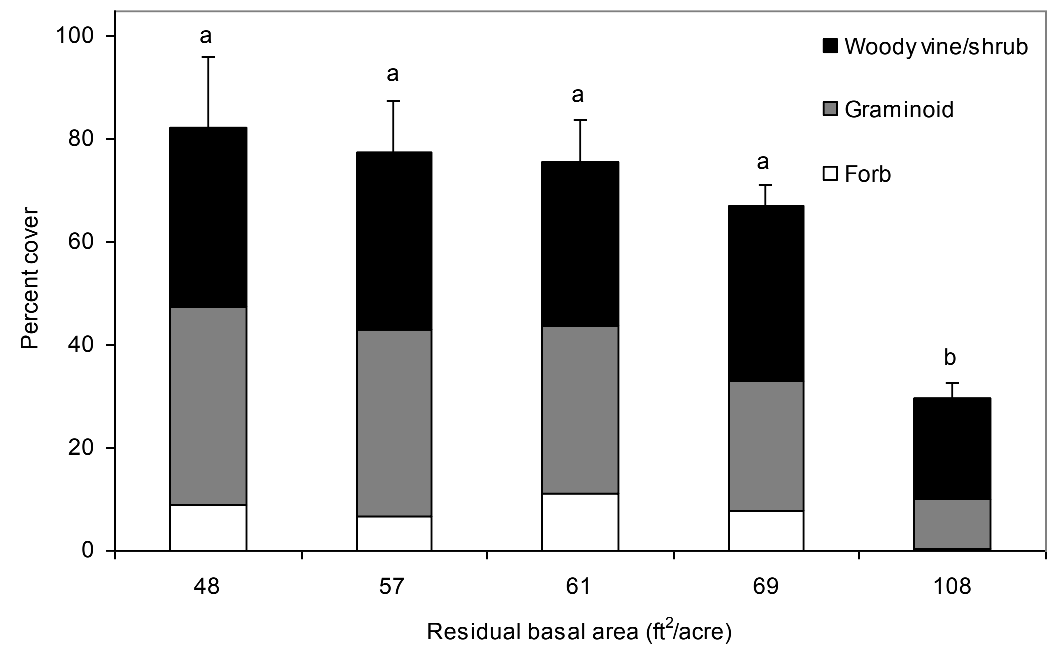

Figure 4--Percent cover by treatment for the main groups of groundstory plants (forb, graminoid, woody vine/shrub) 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Significance was analyzed using Tukey's HSD test on the total number of species; columns sharing different letters are significantly different (P < 0.10).

Table 2--Groundstory percent cover for dominant forb, graminoid, and woody vine/shrub species across the different thinning treatments 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi

	Residual overstory basal area (*feet^2/acre*)				
Taxa	48	57	61	69	108
Forb					
Boehmeria cylindrica	0.6	1.4	0.4	0.6	
Elephantopsus carolinianus	0.3	0.1	4.8	1.4	0.1
Lycopus angustifolius	6.0	0.6	0.0	3.4	
Other	2.2	4.4	6.0	2.3	0.1
Subtotal	9.1	6.5	11.3	7.8	0.3
Graminoid					
Grass*	18.1	17.5	14.7	19.2	4.5
Sedge**	12.9	12.0	10.2	5.3	4.1
Arundinaria gigantea	7.5	3.3	7.8	0.8	1.1
Other	0.0	3.8	0.0	0.0	0.0
Subtotal	38.4	36.6	32.7	25.3	9.8
Woody vine/shrub					
Bignonia capreolata	5.4	5.6	6.4	3.7	4.3
Lonicera japonica	0.7	1.1	1.3	0.6	1.4
Rubus argutus	14.5	4.8	10.1	11.0	0.4
Smilax rotundifolia	3.3	8.4	3.3	8.6	6.2
Toxicodendron radicans	2.8	3.5	2.8	2.9	0.7
Vitis rotundifolia	3.8	6.8	4.7	4.5	4.6
Other	4.7	4.4	3.1	2.7	2.1
Subtotal	35.0	34.5	31.6	34.0	19.6
Grand total	82.5	77.6	75.6	67.0	29.6

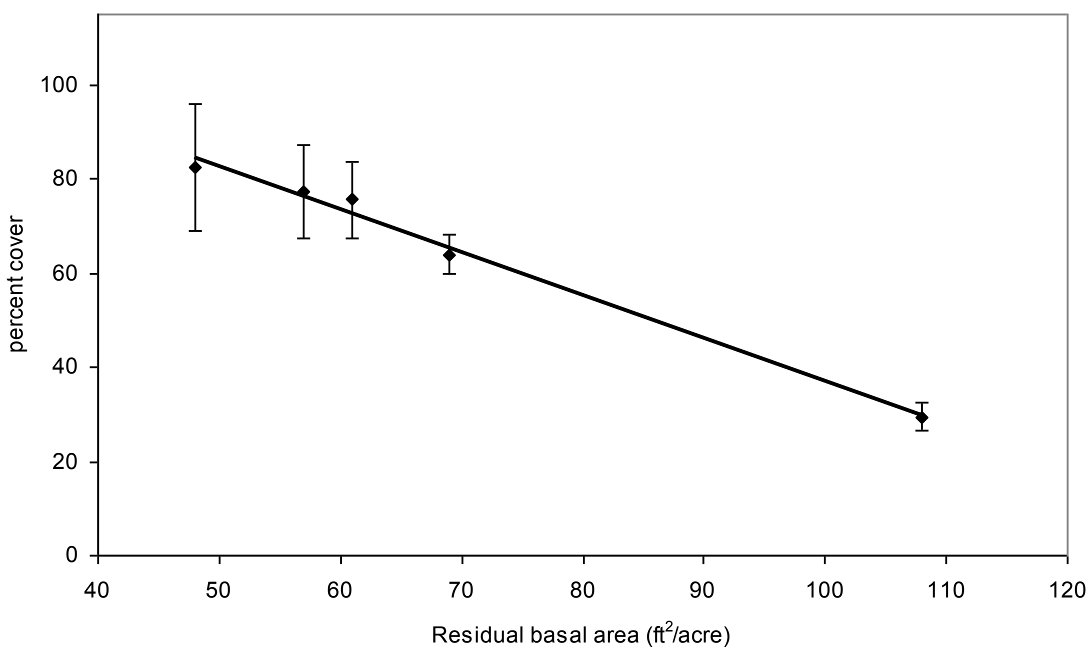

Figure 5--Regression of percent cover by residual basal area (square feet per acre) 5 years after implementing thinning treatments in a minor stream bottom hardwood forest in east-central Mississippi. Standard error bars shown.

from the increased abundance. Overall, cover appears to increase in direct proportion to the intensity of overstory removal. Future work should improve knowledge of groundstory response to thinning in bottomland hardwood stands, and thus assist management efforts aimed at the maintenance of plant diversity (and its many benefits), and at successfully regenerating desirable hardwood species.

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LITERATURE CITED

Aikens, M.L.; Ellum, D.; McKenna, J.J. [and others]. 2007. The effects of disturbance intensity on temporal and spatial patterns of herb colonization in a southern New England mixed-oak forest. Forest Ecology and Management. 252(1-3): 144-158.

Castleberry, S.B.; Ford, W.M.; Miller, K.V. 2000. Influences of herbivory and canopy opening size on forest regeneration in a southern bottomland hardwood forest. Forest Ecology and Management. 131(1-3): 57-64.

Colwell, R.K. 2012. Estimates: statistical estimation of species richness and shared species from samples. Version 8.2. User's guide and application. http://purl.oclc.org/estimates. [Date accessed: October 18, 2012].

Crouch, V.E.; Golden, M.S. 1997. Floristics of a bottomland forest and adjacent uplands near the Tombigbee River, Choctaw County, Alabama. Castanea. 62(4): 219-238.

Duguid, M.C.; Frey, B.R.; Ellum, D.S. [and others]. 2013. The influence of ground disturbance and gap position on understory plant diversity in upland forests of southern New England. Forest Ecology and Management. 303: 148-159.

Gilliam, F.S. 2007. The ecological significance of the herbaceous layer in temperate forest ecosystems. BioScience. 57(10): 845-858.

Jenkins, M.W.; Chambers, J.L. 1989. Understory light levels in mature hardwood stands after partial overstory removal. Forest Ecology and Management. 26(4): 247-256.

Jenkins, M.A.; Parker, G.R. 2000. The response of herbaceous-layer vegetation to anthropogenic disturbance in intermittent stream bottomland forests of southern Indiana, U.S.A. Plant Ecology. 151: 223-227.

Meadows, J.S.; Skojac, D.A., Jr. 2012. Stand quality management in a late-rotation, red oak-sweetgum stand in east Mississippi: preliminary results following thinning. In: Butnor, J.R., ed. 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: 221-229.

Rapp, J.; Shear, T.; Robison, D. 2001. Soil, groundwater, and floristics of a southeastern United States blackwater swamp 8 years after clearcutting with helicopter and skidder extraction of the timber. Forest Ecology and Management. 149(1-3): 241-252.

Rosen, D.J.; Carter, R.; Bryson, C.T. 2006. The recent spread of *Cyperus entrerianus* (Cyperaceae) in the southeastern United States and its invasive potential in bottomland hardwood forests. Southeastern Naturalist. 5(2): 333-344.

Ruzicka, K.J.; Groninger, J.W.; Zaczek, J.J. 2010. Deer browsing, forest edge effects, and vegetation dynamics following bottomland restoration. Restoration Ecology. 18(5): 702-710.

Whigham, D.F. 2004. Ecology of woodland herbs in temperate deciduous forests. Annual Review of Ecology, Evolution, and Systematics. 35: 583-621.

LOBLOLLY PINE HETEROTROPHIC AND AUTOTROPHIC SOIL RESPIRATION AS INFLUENCED BY FERTILIZATION AND REDUCED THROUGHFALL

Brett C. Heim, Brian D. Strahm, and John R. Seiler[1]

Carbon (C) in terrestrial ecosystems is one of the main reservoirs in the global C cycle (Schimel 1995). Within these terrestrial ecosystems, soil C in the form of organic matter and plant biomass are the two largest pools of C. Further, the processes of photosynthesis and respiration that occur in these systems are the two largest fluxes of C globally (Schlesinger 1997). Given their size, even small changes in these pools and fluxes can significantly impact atmospheric CO_2 concentrations. Forest ecosystem management can actively influence global C dynamics by manipulating these pools and fluxes. Afforestation in general, and forest management (silviculture) specifically, can increase terrestrial ecosystem C in soils and biomass (Watson and others 2000). In the southern U.S., intensive management of loblolly pine forests has shown appreciable increases in productivity since widespread establishment of plantations in the 1950s (Fox and others 2007). Understanding the interacting effects of management (e.g., fertilization) and climate variability (e.g., drought) will be critical in guiding the adaptation of these forest ecosystems for the mitigation of negative climate impacts.

In order to quantify the effects of management and climate change, a measure of C storage is necessary. One such measure that forest scientists utilize is known as net ecosystem productivity (NEP). NEP is a measure of the net C accumulated by an ecosystem. For a loblolly pine ecosystem, it represents the C captured by photosynthesis minus the losses due to plant and soil respiration. Direct measurements of NEP are difficult over large geographic areas. Ecosystem C models have the capacity to predict NEP with one modification of their present configuration. There is a need to understand the relative contributions of soil heterotrophic, microbial respiration (R_H) and autotrophic, root respiration (R_A) to the overall belowground, or soil, respiration (R_S). Present estimates suggest R_A and R_H are roughly evenly split (Subke and others 2006), but deviations from this perception could have significant impacts on the estimates C storage in managed forest ecosystems.

In order to attain estimates of R_S components R_H and R_A, R_S needs to be measured in a root-free environment. Such conditions hardly exist in nature. On small scales, however, these conditions can be artificially created by use root-severing collars to cut the supply of plant carbohydrates. Over time, stored carbohydrates in roots (Hogberg and others 2001) used for continued maintenance are depleted, and R_A falls to zero. At this point, a measure of R_S is equal to R_H.

During the 2012 field season, we tested this at the Virginia Tier III PINEMAP installation. This 9-year-old loblolly pine stand is located in the Appomattox-Buckingham State Forest in the Piedmont of Virginia. This location represents the northernmost range of climatic conditions where loblolly pine is intensively managed in the southeastern U.S. The study utilizes large (0.10 ha) plots in a fully replicated (n = 4) 2 x 2 factorial design that includes fertilization (optimal nutrition, no addition) and throughfall exclusion (0, 30 percent). In each treatment three subsample locations were measured, and means were used to estimate R_H.

Respiration measurements were taken approximately every 2 weeks both adjacent to and on top of each root-severing collar to measure the decline in R_A over 90 days. Soil temperature (at 12 cm) and moisture

[1]Graduate Research Assistant, Assistant Professor, and Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061.

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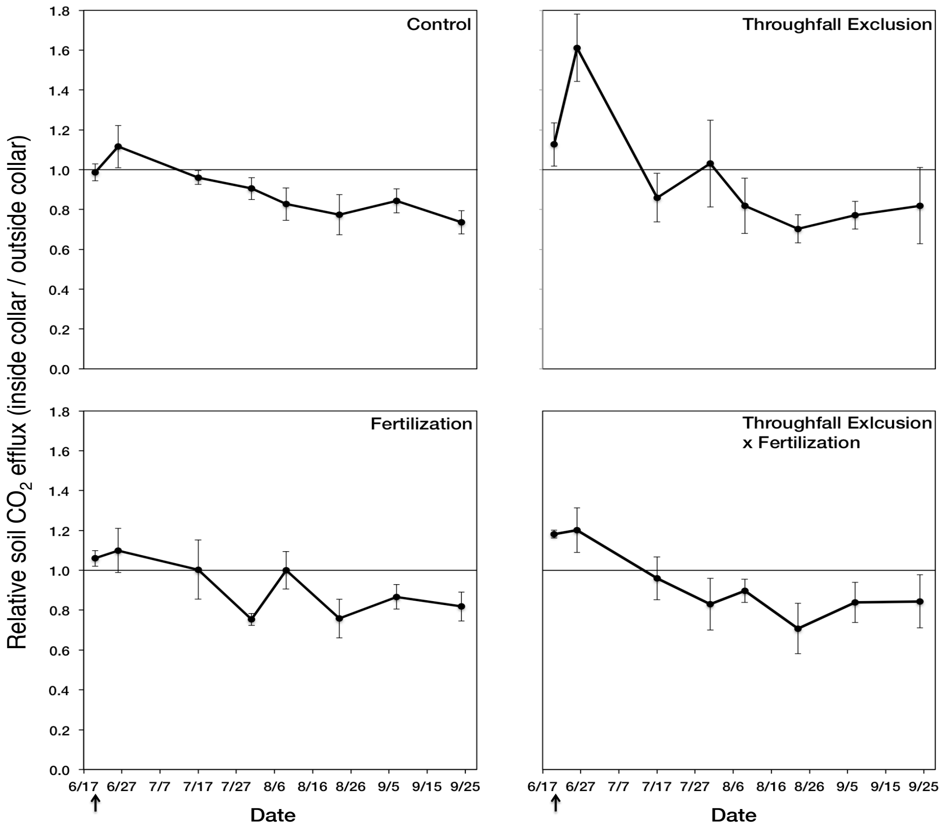

Figure 1--Relative soil CO_2 efflux (inside collar / outside collar) in a 9-year-old loblolly pine stand located on the Virginia Piedmont as influenced by the four treatment combinations. Arrow indicates date of collar installation.

measurements (0 to 12 cm) were also taken adjacent to the collar during each measurement.

Respiration initially increased inside the collars due to the disturbance of collar installation. After a period of equilibration, however, the respiration inside the collar began to decrease relative to outside the collar before stabilizing after approximately 65 days (fig. 1). After stabilization, means for each plot in each treatment were used to estimate R_H. R_H is estimated to account for 76.1, 81.4, 74.0, and 81.4 percent of R_S (table 1). Since there are no significant differences among treatments, a grand partitioning coefficient of 78.2 percent was calculated among all treatments.

Based on these initial results, PINEMAP researchers will be deploying this method at the other Tier III installations as well as in a broader regional context at some Tier II sites to estimate loblolly pine NEP across the range of the species.

LITERATURE CITED

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.

Hogberg, P.; Nordgren, A.; Buchmann, N. [and others]. 2001. Large-scale forest girdling shows that current photosynthesis drives soil respiration. Nature. 411: 789-792.

Schimel, D.S. 1995. Terrestrial ecosystems and the carbon cycle. Global Change Biology. 1: 77-91.

Schlesinger, W.H. 1997. Biogeochemistry: an analysis of global change. 2d ed. San Diego: Academic Press. 588 p.

Subke, J.A.; Inglima, I.; Cotrufo, M.F. 2006. Trends and methodological impacts in soil CO_2 efflux partitioning: a metaanalytical review. Global Change Biology. 12: 921-943.

Watson, R.T.; Noble, I.R.; Bolin, B. [and others]. 2000. IPCC land use, land-use change, and forestry. Cambridge, UK: Cambridge University Press. 375 p.

EFFECTS OF SPRING PRESCRIBED FIRE ON SHORT-TERM, LEAF-LEVEL PHOTOSYNTHESIS AND WATER USE EFFICIENCY IN LONGLEAF PINE

John K. Jackson, Dylan N. Dillaway, Michael C. Tyree, and Mary Anne Sword Sayer[1]

Abstract--Fire is a natural and important environmental disturbance influencing the structure, function, and composition of longleaf pine (*Pinus palustris* Mill.) ecosystems. However, recovery of young pines to leaf scorch may involve changes in leaf physiology, which could influence leaf water-use efficiency (WUE). This work is part of a larger seasonal burning study containing three treatments: spring burn, fall burn, and no burn. The study site is a 21.4-ha (53-acre) longleaf pine plantation planted in 2005 and located on the Winn Ranger District of the Kisatchie National Forest in central Louisiana. This work assessed changes in leaf gas exchange and their resultant WUE in 7-year-old longleaf pine among spring burned and no burn (control) treatments during the 2011 and 2012 growing seasons. Treatment effects on intrinsic WUE [carbon assimilation/ stomatal conductance ($g_{s)}$; iWUE] were assessed using an infrared gas analyzer. We hypothesized that leaf area loss resulting from scorch will: (1) increase photosynthesis, and (2) increase leaf-level iWUE as a result of increased photosynthesis. In the 3 months after the prescribed burn, we observed increases in photosynthesis (A_{Sat}) in the new foliage of scorched trees that decreased below control levels during the following growing season. During the 2012 growing season, 1 year after the burn, there were larger diurnal increases in iWUE in scorched trees than in controls that increased as temperature increased and water availability decreased. This response was due to greater diurnal reductions in g_s in scorched trees.

INTRODUCTION

The longleaf pine (*Pinus palustris* Mill.) ecosystem, once the dominant ecosystem of the southeastern United States, is fire-dependent and currently occupies < 3 percent of its original range (Frost 1993). Prescribed fire is an important management prescription used in longleaf pine stands (Brockway and Lewis 1997). These fires consume the grassy fuel layer and litter layer but do not burn the crowns of large overstory trees. However, the foliage of younger trees and seedlings are often scorched or desiccated resulting in the full or partial defoliation of the tree (Andrews 1917).

Defoliation by fire reduces the photosynthetic area per tree and causes a variety of physiological responses in trees. Some of these responses have been interpreted as recovery and compensatory growth mechanisms, largely because they enhance the rate of photosynthesis and could lead to higher rates of growth relative to a non-defoliated tree (Sword Sayer and Haywood 2009, Vanderklein and Reich 1999). Most studies show that partial or the entire removal of leaf area results in an increase in photosynthetic rates of remaining or new tissue (Detling and others 1979, Heichel and Turner 1983, Painter and Detling 1979, Reich and others 1993, Vanderklein and Reich 1999, Wallace and others 1984). Defoliation studies done with red pine (*Pinus resinosa* Ait.) have shown that partial defoliation stimulated gas exchange rates that diminished with time (Reich and others 1993). Net photosynthesis was 25 to 50 percent higher in defoliated trees than in non-defoliated control trees 6 weeks after defoliation, and this effect continued at a declining rate until there was no longer any difference between defoliated and non-defoliated control trees (Reich and others 1993). A defoliation study done with red pine and Japanese larch (*Larix leptolepis* Sieb. and Zucc.) seedlings showed increases in photosynthesis during the year following a defoliation (Vanderklein and Reich 1999). Red pine also showed photosynthetic increases in the second year after being defoliated (Vanderklein and Reich 1999).

Research has been conducted on changes in water-use efficiency in plant communities that experience fire or defoliation. Cowan and Farquhar (1977) proposed that plants control stomata to optimally satisfy the trade-off between the amount of carbon (C) assimilated and the amount of water transpired (Cowan and Farquhar 1977). Stomatal conductance (g_s)

[1]Graduate Research Assistant, Louisiana Tech University, School of Forestry, Ruston LA 71270; Assistant Professor, Unity College, Center for Natural Resource Management and Protection, Unity, ME 04988; Assistant Professor, Louisiana Tech University, School of Forestry, Ruston, LA 71270; and Research Plant Physiologist, USDA Forest Service, Southern Research Station, Pineville, LA 71360.

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determines both diffusion of CO_2 into the leaf and diffusion of water out of the leaf with the diffusion coefficient being higher for the lighter H_2O molecules. Decreased stomatal apertures restrict diffusion of H_2O about 1.6 times more than CO_2 (Whelan and others 2013). Because this process acts on the leaf water use efficiency (WUE), the ratio between photosynthesis and g_s called intrinsic WUE (iWUE) is more appropriate than WUE for describing the biochemical functions of vascular plants (Beer and others 2009). Whelan and others (2013) showed increased iWUE in longleaf pine in the 30 days after being burned due to reductions in evapotranspiration and g_s. Busch and Smith (1993) examined changes in iWUE in the dominant woody taxa from low-elevation riparian plant communities of the southwestern U.S that were recently burned. These plants experienced higher iWUE when burned compared to unburned control plants. However, Reich and others (1993) reported a tendency for defoliated plants to have lower iWUE than control plants due to large increases in g_s. Sword Sayer and Haywood (2009) have identified increased photosynthesis as a possible recovery mechanism to crown scorch in young longleaf pine and increased iWUE as a function of increased photosynthesis.

When resulting in crown scorch, prescribed fire may increase rates of C capture in new foliage and increase iWUE in longleaf pine during the hot and dry summer months in Louisiana. The overall objective of this study was to determine how spring prescribed fire affects photosynthesis under saturating light (A_{Sat}) and intrinsic water-use efficiency (iWUE) in a young longleaf pine stand. We monitored the changes in photosynthesis and iWUE for 15 months in longleaf pine following a spring prescribed fire. Our first objective was to analyze the immediate changes in photosynthesis and iWUE in new foliage during the same growing season that the burn was prescribed. Our second objective was to analyze the diurnal changes in photosynthesis and iWUE during the growing season 1 year following the spring burn. Understanding the effects of crown scorch on water-use as well as photosynthetic capacity of longleaf pine is needed to develop sustainable management practices to restore these threatened ecosystems and to minimize growth loss in plantation grown longleaf pine.

MATERIALS AND METHODS

Study Site

The study site is located in a 7-year-old longleaf pine stand on the Winn Ranger District of Kisatchie National Forest in central Louisiana (compartment 24, stand 20, T12N, R5W, sec 2, latitude: 32° 3, 10.345"N, longitude: 92° 51' 20.279" W). This region has an average annual temperature of 18.44 °C and receives an average of 116.84 cm of precipitation annually (NOAA 2012). Prior to longleaf pine planting, this site was used as a loblolly pine plantation that was clearcut in 2001. The site is a 21.4-ha (53-acre) longleaf pine plantation that was planted in 2005 and previously burned in 2008. The soil at the site consists of a fine sandy loam (0 to 20 percent slopes) and is classified as the Sacul series (fine, mixed, active, thermic Aquic Hapludults)(NRCS 2012). The Sacul series consists of very deep, moderately well-drained, slowly permeable soils that formed in acid, loamy, and clayey marine sediments. These soils are located on the uplands of the western and southern coastal plains (NRCS 2012).

The study was set up as a randomized complete block (RCB) design replicated three times. Each plot was subjected to either a spring burn or no burn (control) treatment designed to result in significant crown scorch. In September 2010 firebreaks were laid out using GPS equipment (Trimble Juno SD Handheld; Sunnyvale, CA) to prevent the spread of fire to undesired areas. Firebreaks were also installed and maintained around the entire perimeter of the study area.

The spring plots were burned on May 16, 2011, just prior to a significant drought event in Louisiana that lasted until the summer of 2012. The three spring plots that were burned consisted of 9.3 ha (22.98 acres). Fuel samples were collected from three randomly located 0.22-m^2 vegetation subplots per plot (nine total subplots) that were scheduled to be burned. Vegetation was sampled as described by Haywood (2009). Pre-burn fuel loading for the spring burn plots was done on May 12, 2011, and the post-burn sampling was done on May 26, 2011 (table 1). All vegetation was oven dried at 70 °C and weighed to determine the fuel consumption per plot on a dry-weight basis.

Gas Exchange Measurements

Gas exchange data was collected after the spring burn during the growing season. Rates were monitored using a Li-Cor 6400 infrared gas analyzer/portable photosynthesis system (Li-Cor Biosciences, Lincoln, NE). The cuvette of the analyzer was operated under ambient CO_2 concentrations, and saturating light (photosynthetically active radiation of 1800 μmol m^{-2} s^{-1}) was applied so that photosynthesis was maximized throughout the measurement period. All rates were measured at ambient air temperature and ambient relative humidity. Measurements were made on the spring burn and control plots in August and October of 2011, 3 and 5 months after the spring plots were burned, respectively. Morning and afternoon gas exchange measurements were collected three times during the summer of 2012 (June, July, and August). Rates were measured on six individuals per experimental unit during each campaign along with tree height, scorch height, and diameter. Gas exchange rates were monitored on the current year's most recently elongated flush. All needles were excised from the eastern, upper one third of each tree. Two needle fascicles per tree were measured, and the rates were averaged and adjusted for needle area. Needle surface area was calculated by measuring fascicle diameters for all needles measured. Needle area was computed using the following formula from Ginn and others (1991):

$$SA = 3.14159(d)(l) + (n)(d)(l) \qquad (1)$$

where d = fascicle diameter, l = needle length in cuvette, and n = number of needles in the fascicle. The Li-Cor 6400 calculated A_{Sat} and g_S, and iWUE was calculated following each gas exchange campaign by dividing A_{Sat} by g_S.

Statistical Analysis

Statistics were conducted on the absolute values for objective 1, and the results are displayed in a relative differences graph showing the percentage change from the control absolute values. The results for objective 2 are displayed showing the diurnal ratio of change from morning to afternoon measurements for each treatment. All statistics for objective 2 were conducted on the ratio of change from a.m. to p.m. for each variable in each treatment. All analyses were performed using the GLM and REG procedures. The treatment effects were analyzed by ANOVA at $\alpha = 0.05$ significance level. All data were analyzed using the SAS statistical package Version 9.1 (SAS Institute Inc., Cary, NC).

RESULTS AND DISCUSSION

The range of A_{Sat} values we observed were similar to the photosynthetic rates reported for loblolly pine *(Pinus taeda* L.) and white pine (*Pinus strobus* L.) grown in the southeastern United States (Maier and Teskey 1992, Teskey and others 1987). After the initial renewal of leaf area following the prescribed fire we observed short-term (3 to 4 month) increases in A_{Sat} ($p < 0.001$) and g_s ($p < 0.001$) in scorched trees (fig. 1). These increases returned to control levels by October 2011 (5 months post-burn). This short-term increase in A_{Sat} is consistent with most defoliation studies (Detling and others1979, Ericsson and others 1980, Eyles and others 2013, Heichel and Turner 1983, Quentin and others 2010, Painter and Detling 1979, Reich and others 1993, Vanderklein and Reich 1999, Wallace and others 1984). Reich and others (1993) showed that photosynthesis is possibly enhanced via reduced stomatal limitation, showing a tendency for defoliated plants to have greater g_s and lower iWUE than control plants. Increases in A_{Sat} after defoliation may be due to increased leaf nitrogen (N) concentrations, since photosynthetic capacity and leaf N are functionally related (Kozlowski and Keller 1966); however Reich and others (1993) showed that red pine foliar N concentrations were consistently unaffected by defoliation, regardless of intensity of defoliation or the timing of defoliation. The physical structure of a tree canopy can also influence photosynthetic rates. Gold and Caldwell (1990) showed that increases in photosynthesis in defoliated tussock grass were directly related to a higher proportion of younger foliage and a smaller fraction of shaded foliage. Crown scorch on young longleaf pine inherently removes the lower canopy foliage and reallocates new foliage to the upper crown

Table 1--Pre-burn and post-burn fuel loads and levels of consumption for the spring prescribed burning treatment

Treatment	Pre-burn fuel biomass	Standard error	Post-burn fuel biomass	Standard error	Fuel consumption	p-value
	mg ha^{-1}		*mg ha^{-1}*		*percent*	
Spring	16.24	±1.97	1.76	±0.51	89.17	<0.001

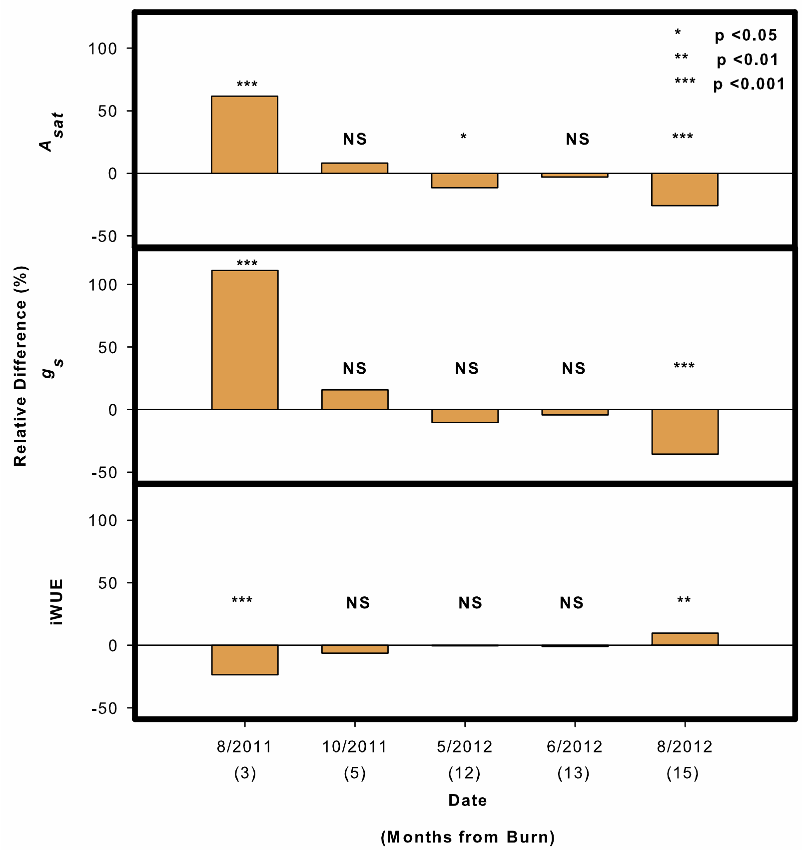

Figure 1-- Comparison of relative changes of A_{Sat}, g_s, and iWUE between spring burned and unburned trees. The bars represent the relative difference of the burned value, showing the percentage change from the control absolute values.

where light levels are more favorable. During May 2012 (the following growing season), A_{Sat} in the scorched trees was slightly lower ($p < 0.05$) than control trees while there were no statistical differences in g_s. As the summer of 2012 progressed, A_{sat} and g_s continually declined to well below control levels ($p < 0.001$; $p < 0.001$) as the climate became increasingly hotter and drier at the end the 15-month study (fig. 1). These findings support our hypothesis for greater A_{Sat} in scorched trees; however, the decrease in A_{Sat} during the 2012 growing season was unexpected. We expected to see increases in A_{sat} during the second growing season (2012) similar to Prudhomme (1982) and Vanderklein and Reich (1999).

Most studies show increased iWUE and decreased g_s in the days after defoliation (Ewers and others 2001, Whelan and others 2013); however, some studies have suggested that iWUE may be reduced in defoliated plants as a

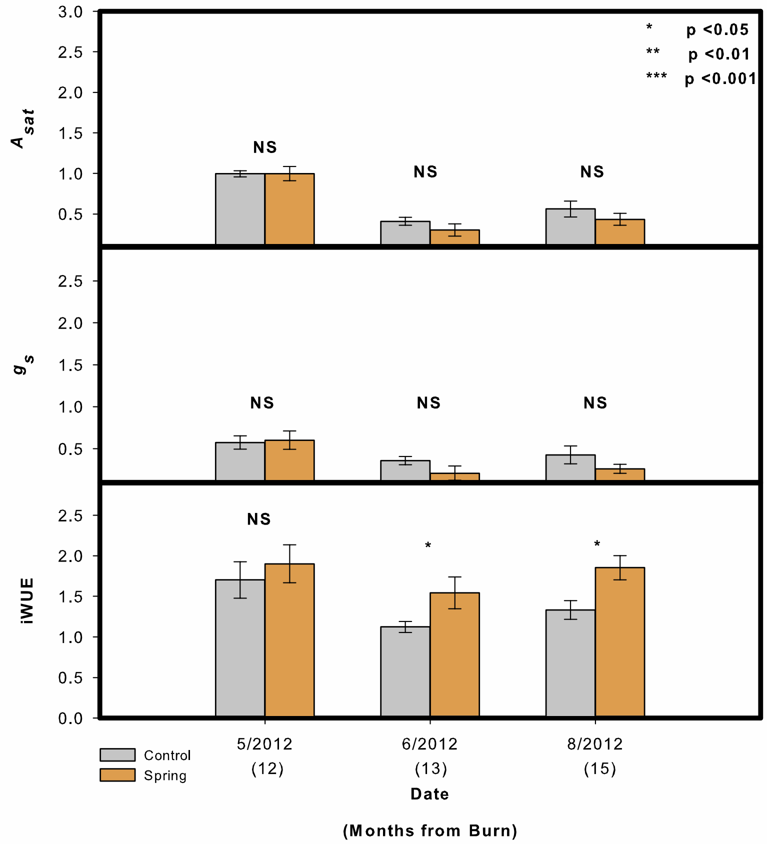

Figure 2-- Comparison between spring burned and unburned diurnal changes in A_{Sat}, g_s, and iWUE. The bars represent the ratio of change for each measurement from a.m. to p.m. A value of 1 indicates no change from a.m. to p.m., a value above 1 indicates an increase from a.m. to p.m., and a value below 1 indicates a decrease from a.m. to p.m. The orange bars represent the spring burned values, and the gray bars represent the unburned values.

result of increased g_s associated with a defoliation-induced shift in plant leaf area per unit root surface (Gold and Caldwell 1990, Reich and others 1993). We observed significant decreases in iWUE ($p < 0.001$) 3 months after the prescribed fire due to large increases in g_s ($p < 0.001$). Five months following the spring burn, these differences were no longer present. These findings are consistent with other defoliation studies conducted on pines (Reich and others 1993). During the summer after the spring plots were burned, there were no differences in iWUE until late summer. In August 2012 (15 months after the prescribed fire), iWUE increased significantly ($p < 0.01$) above control levels due to large decreases in g_s ($p < 0.001$) (fig. 1). Our findings partially support our original hypothesis in that we expected to see increases in iWUE in the scorched trees throughout the summer of 2012; however, we hypothesized these increases would be due to greater A_{Sat}.

During the summer of 2012 we observed no difference in diurnal changes in A_{Sat} or g_s between scorched and un-scorched trees. There were larger diurnal increases in iWUE in the scorched trees than in control trees in June and August as the year progressed into the hot and dry summer months ($p < 0.05$) that occur in central Louisiana (fig. 2). This increase was due to greater diurnal changes in g_s. So as temperature increased and water availability decreased, trees defoliated through crown scorch had significantly greater diurnal increases in iWUE than control trees

CONCLUSIONS

A prolonged drought after the prescribed fire during the 2011 and 2012 study period may have interfered with recovery from crown scorch and may have affected gas exchange rates. Interactions among environmental stresses such as drought and crown scorch may decrease plant resistance to structural damage (Ellsworth and others 1994). The initial short-term increase in A_{Sat} in the new foliage of scorched trees may represent an initial recovery mechanism. However, in the year after the prescribed fire, the new foliage in scorched trees unexpectedly had lower A_{Sat} levels than control trees. Even though new foliage in scorched trees assimilated C at a lower rate than control trees, iWUE increased. This increase was not a result of increased A_{Sat} in new foliage of scorched trees but rather a function of decreased g_s. Our results demonstrate potential reductions in longleaf pine C fixation a year after foliage removal. Future research will continue to monitor how the crown scorch affects gas exchange during different seasonal prescribed fires in young longleaf pine and how leaf area removal influences above and belowground carbohydrate allocation.

ACKNOWLEDGEMENTS

Special thanks to USDA Forest Service, Southern Research Station for funding this project. Thanks to Louisiana Tech University ANS Research Mini-Grant for financial support. Also, thanks to Dustin Dill and Larry Kile with the Forest Service, Winn Ranger District for assisting with study installation, burning, and study site maintenance. We would also like to thank Wesley Palmer for all GPS, GIS, and maps associated with this project. Finally, thanks to Wilson Hood, Brian Byrd, Jessica LaGrone, Savannah Best, Chelsea Stringfield, Tyler Durbin, Jared Simoneaux, and JinYan Yang for assisting with study installation, field, and laboratory measurements.

LITERATURE CITED

Andrews, E.F. 1917. Agency of fire in propogation of longleaf pines. Botanical Gazette. 64: 497-508.

Beer, C.; Ciais, P.; Reichstein, M. [and others]. 2009. Temporal and among-site variability of inherent water use efficiency at the ecosystem level. Global Biogeochemical Cycles. 23: 1-13.

Brockway, D.G.; Lewis C.E. 1997. Long-term effects of dormant-season prescribed fire on plant community diversity, structure and productivity in a longleaf pine wiregrass ecosystem. Forest Ecology and Management. 96: 167-183.

Busch, D.E.; Smith, S.D. 1993. Effects of fire on water and salinity relations of riparian woody taxa. Oecologia. 94(2): 186-194.

Cowan, I.R.; Farquhar, G.D. 1977. Stomatal function in relation to leaf metabolism and environment. In: Jennings, D.H., ed. Integration of activity in the higher plant. Cambridge, UK: Cambridge University Press: 471-505.

Detling, J.K.; Dyer, M.I.; Winn, D.T. 1979. Net photosynthesis, root respiration, and regrowth of *Bouteloua grucilis* following simulated grazing. Oecologia. 41: 127-134.

Ellsworth, D.; Tyree, M.; Parker, B.; Skinner, M. 1994. Photosynthesis and water-use efficiency of sugar maple (*Acer saccharum*) in relation to pear thrips defoliation. Tree Physiology. 14: 619-632.

Ericsson, A.; Hellkvist, J.; Hillerdal-Hagstromer, K. [and others]. 1980. Consumption and pine growth - hypotheses on effects on growth processes by needle-eating insects. Ecological Bulletins. 32: 537-545.

Ewers, B.E.; Oren, R; Phillips, N.; Stromgren, M.; Linder S. 2001. Mean canopy stomatal conductance responses to water and nutrient availabilities in *Picea abies* and *Pinus taeda*. Tree Physiology. 21:841-850.

Eyles, A.; Pinkard, E.A.; Davies, N.W. [and others]. 2013. Whole-plant versus leaf-level regulation of photosynthetic responses after partial defoliation in *Eucalyptus globulus* saplings. Journal of Experimental Botany. 64: 1625-36.

Frost, C.C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. In: Proceedings, 18th Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station. 18: 17-44.

Ginn, S.E.; Seiler, J.R.; Cazell, B.H.; Kreh, R.E. 1991. Physiological growth responses of eight-year-old loblolly pine stands to thinning. Forest Science. 37(4): 1030-1040.

Gold, W.G.; Caldwell, M.M. 1990. The effects of the spatial pattern of defoliation on regrowth of a tussock grass: III. Photosynthesis, canopy structure and light interception. Oecologia 82: 12–17.

Haywood, J.D. 2009. Eight years of seasonal burning and herbicidal brush control influence sapling longleaf pine growth, understory vegetation, and the outcome of an ensuing wildfire. Forest Ecology and Management. 258: 295-305.

Heichel, G.H.; Turner, N.C., 1983. CO_2 assimilation of primary and regrowth foliage of red maple (*Acer rubrum* L.) and red oak (*Quercus rubra* L.): response to defoliation. Oecologia. 57: 14–19.

Kozlowski, T.T.; Keller, T. 1966. Food relations of woody plants. The Botanical Review. 32: 293-382.

Maier, C.; Teskey, R. 1992. Internal and external control of net photosynthesis and stomatal conductance of mature eastern white pine (*Pinus strobus*). Canadian Journal of Forest Research. 22: 1387-1394.

NOAA, N.W.S. 2012. Climate data, Quitmann, LA. http://www.noaa.gov. [Date accessed: January 6, 2013].

NRCS [Natural Resources Conservation Service]. 2012. Web soil survey. Washington, DC: U.S. Department of Agriculture Natural Resources Conservation Service. http://websoilsurvey.nrcs.usda.gov/. [Date accessed: April 4, 2013]

Painter, E.L.; Detling, J.K. 1979. Effects of defoliation on net photosynthesis and regrowth of western wheatgrass. Journal of Range Management. 34: 68-71.

Prudhomme, T.I. 1982. The effect of defoliation history on photosynthesis rates in mountain birch. Report of the Kevo Subarctic Research Station. 18: 5–9.

Quentin, A.G.; Pinkard, E.A.; Beadle, C.L. [and others]. 2010. Do artificial and natural defoliation have similar effects on physiology of *Eucalyptus globulus* Labill. seedlings? Annals of Forest Science. 67: 203-203.

Reich, P.B.; Walters, M.B.; Krause, S.C. [and others]. 1993. Growth, nutrition and gas exchange of *Pinus resinosa* following artificial defoliation. Trees. 7: 67-77.

Sword Sayer, M.A.; Haywood, J.D. 2009. Fire and longleaf pine physiology--does timing affect response? In: Proceedings of the Society of American Foresters national convention. Washington, DC: SAF: 1-11.

Teskey, R.O.; Bongarten, B.C.; Cregg, B.M. [and others]. 1987. Physiology and genetics of tree growth response to moisture and temperature stress: an examination of the characteristics of loblolly pine (*Pinus taeda* L.). Tree Physiology. 3: 41-61.

Vanderklein, D.W.; Reich, P.B. 1999. The effect of defoliation intensity and history on photosynthesis, growth and carbon reserves of two conifers with contrasting leaf lifespans and growth habits. New Phytologist. 144: 121-132.

Wallace, W.; Secor, J.; Schrader, L.E. 1984. Rapid accumulation of gamma-aminobutyric acid and alanine in soybean leaves in response to an abrupt transfer to lower temperature, darkness, or mechanical manipulation. Plant Physiology. 75: 170-175.

Whelan, A.; Mitchell, R.; Staudhammer, C.; Starr, G. 2013. Cyclic occurrence of fire and its role in carbon dynamics along an edaphic moisture gradient in longleaf pine ecosystems. PLOS ONE 8(1): e54045. http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0054045. [Date accessed: March 2, 2013].

LONG-TERM RESPONSE OF YELLOW-POPLAR TO THINNING IN THE SOUTHERN APPALACHIAN MOUNTAINS

Tara L. Keyser and Peter M. Brown[1]

Yellow-poplar (*Liriodendron tulipifera* L.) is the most abundant individual tree species (in terms of volume) in the southern Appalachian Mountains, with Forest Inventory and Analysis (FIA) reports documenting a continuous increase in yellow-poplar over the recent years (Brown 2003, Schweitzer 1999, Thompson 1998). Current management efforts in even-aged yellow-poplar stands rarely include thinning operations. However, thinning prescriptions largely driven by timber-related goals and objectives were once commonplace across the region. As a consequence of shifting objectives, these previously thinned stands, which were traditionally managed on a timber- or financially-related rotation length, are no longer actively managed within the original silvicultural prescription. Despite not being under an active management plan, these previously thinned stands may continue to respond to past treatments. As the focus of forest management on many public lands shifts away from timber production and extraction to ecosystem-based management, it is important to understand the long-term effects that previous management activities have on structure to better inform current management decisions. Because many of these stands were, in the past, harvested prior to their biological rotation age, there is limited quantitative data regarding the long-term effects of previous management activities on long-term growth patterns over time. In this paper, we analyzed 40 years of post-thinning growth data to assess the long-term response of yellow-poplar stands to thinning across a broad age and site quality gradient in the southern Appalachian Mountains.

Between 1960 and 1963, 134 plots 0.1 ha in size were established across an age and site quality gradient in yellow-poplar stands throughout the southern Appalachian Mountains. All plots were thinned from below, with post-thinning relative density ranging between 12 and 56 percent. In 2009, increment cores from five dominant/co-dominant yellow-poplar trees were obtained from all plots. Radial growth was crossdated, measured, and converted to annual basal area increment (BAI; $cm^2\ yr^{-1}$). Using plot-level BAI chronologies, average annual BAI for five time periods was calculated: (1) 10 years prior to thinning (BAI_{pre}); (2) between 1 and 10 years post-thinning (BAI_{post10}); (3) between 11 and 20 years post-thinning (BAI_{post20}); (4) between 21 and 30 years post-thinning (BAI_{post30}); and (5) between 31 and 40 years post-thinning (BAI_{post40}). Plots were classified into three density classes based on post-thinning relative density [Stand Density Index $(SDI)_{observed}/SDI_{maximum}$]: (1) low (relative density < 0.25); (2) moderate (relative density ≥ 0.25 but < 0.35); and (3) high (relative density ≥ 0.35 but < 0.60).

We used analysis of covariance (ANCOVA) to determine the effects of density (low, moderate, high), site index, average age at the time of thinning, and time since thinning (10, 20, 30, and 40 years post-thinning) on relative post-thinning BAI ($RBAI_{post}$). Relative BAI is unitless and is defined as average annual BAI_{post10}, BAI_{post20}, BAI_{post30}, and BAI_{post40} divided by BAI_{pre}. Values of $RBAI_{post} < 1.0$ signify a slow-down in growth relative to pre-thinning rates, whereas values of $RBAI_{post} > 1.0$ indicate an increase in growth.

$RBAI_{post}$ varied across time periods and density classes (table 1). During the first decade post-thinning, 92, 86, and 57 percent of plots in the low-, moderate-, and high-density classes, respectively, contained trees whose RBA_{post} values were ≥ 1.0. During that first decade post-thinning, 23 and 14 percent of the plots in low- and moderate-density classes, respectively, contained trees that experienced at least a 100 percent increase in BAI relative to pre-thinning growth rates. The proportion of plots containing trees that displayed an increase in growth

[1]Research Forester, USDA Forest Service, Southern Research Station, Asheville, NC 28806; and Ecologist, Rocky Mountain Tree Ring Research, Inc., Fort Collins, CO 80526.

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Table 1--Percentage of plots in low (*n*=39), moderate (*n*=42), and high (*n*=53) density classes in each cumulative relative basal area increment ($RBAI_{post}$) category during the 10, 20, 30, and 40 year growth periods post-thinning

Density class	≤1.0	≥1.0	≥1.1	≥1.2	≥1.3	≥1.4	≥1.5	≥2.0	≥2.5	≥3.0
					10 years post-thinning					
Low	8	92	87	82	69	67	67	23	10	3
Moderate	14	86	81	71	64	55	45	14	0	0
High	43	57	38	25	25	15	11	0	0	0
					20 years post-thinning					
Low	5	95	92	79	77	64	56	33	10	3
Moderate	12	88	81	64	57	55	43	12	2	0
High	30	70	62	47	34	28	23	2	0	0
					30 years post-thinning					
Low	36	64	59	46	36	26	21	8	5	3
Moderate	62	38	26	17	17	14	7	0	0	0
High	72	28	17	11	8	6	4	0	0	0
					40 years post-thinning					
Low	29	71	71	68	58	47	37	5	8	3
Moderate	67	33	26	21	19	14	14	2	2	0
High	55	45	32	30	25	17	13	2	2	0

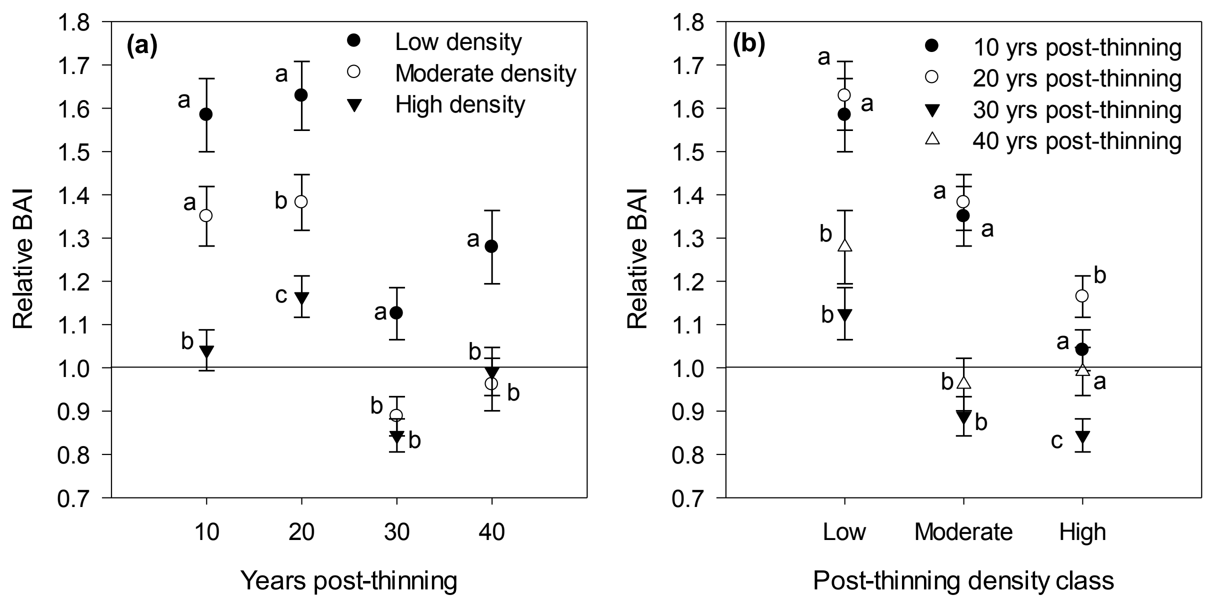

Figure 1--Post-thinning relative basal area increment ($RBAI_{post}$) among time periods within a given density class (a) and among density classes within a given time period (b). Values and error bars represent the lsmeans and standard errors, respectively at the median site index value (site index = 32.3). Lsmeans followed by the same letter are not significantly different. The solid line (y = 1) indicates no change between pre- and post-thinning BAI.

relative to pre-thinning rates generally increased between 11 and 20 years post-thinning. By the fourth decade following thinning, the sample trees in 71, 33, and 45 percent of plots in the low-, moderate-, and high-density classes, respectively, continued to experience an increase in BAI relative to pre-thinning growth rates.

Site index was a significant and positive covariate in the ANCOVA of $RBAI_{post}$, with the effects of site index similar across density classes. During the 10-year time period following thinning, the only significant differences in $RBAI_{post}$ among density classes was between the low- and high- and moderate- and high-density classes (fig. 1a). As time progressed, $RBAI_{post}$ in the low-density class continued to exceed that in the moderate- and high-density classes. In the low- and moderate-density classes, we observed no significant differences in $RBAI_{post}$ between the 10 and 20 year time periods post-thinning (fig. 1b). There was a significant decrease in $RBAI_{post}$ during the third decade post-thinning. The decline in $RBAI_{post}$ during the third decade post-thinning appears to be related to moderate drought conditions across the region during the 1980s.

It is apparent from the results presented here that intermediate silvicultural treatments can have long-lasting effects on stand structure. For yellow-poplar, the increase in tree growth at the lowest residual densities was sustained over the 40-year period encompassed by this study. As these stands approach the traditional rotation age, it is apparent that crop trees continue to benefit from previous thinnings. Beyond traditional timber objectives, the thinning conducted 40 years ago accelerated the development of large trees, a key attribute associated with mixed-mesophytic forests in the later stages (e.g., understory re-initiation and old-growth) of stand development (Greenberg and others 1997). It appears that thinning across a broad age class can accelerate tree growth in the long term, and may serve as an initial restoration treatment in these homogenous yellow-poplar stands.

LITERATURE CITED

Brown, M.J. 2003. Forest statistics for the mountains of North Carolina, 2002. Resour. Bull. SRS-87. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 59 p.

Greenberg, C.H.; McLeod, D.E.; Loftis, D.L. 1997. An old-growth definition for western and mixed mesophytic forests. Gen. Tech. Rep. SRS-16. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 21 p.

Schweitzer, C.J. 1999. Forest statistics for east Tennessee, 1999. Resour. Bull. SRS-51. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 60 p.

Thompson, M.T. 1998. Forest statistics for North Georgia, 1998. Resour. Bull. SRS-35. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 67 p.

INITIAL RESPONSE OF PONDBERRY RELEASED FROM HEAVY SHADE

Brian Roy Lockhart, Emile S. Gardiner, and Theodor D. Leininger[1]

Abstract--Pondberry [*Lindera melissifolia* (Walt.) Blume] is a federally endangered woody shrub in the Lauraceae family. The U.S. Fish and Wildlife Service developed a recovery plan in 1993 that emphasized the need to study pondberry biology and ecology and to develop management practices that would promote recovery and conservation of this species. We initiated a large-scale study in 2005 to determine the effects of soil flooding and light availability on pondberry survival and growth. The study was conducted in Sharkey County, MS in a network of research impoundments known as the Flooding Research Facility. Following the conclusion of our initial research, we implemented a release study to utilize plants which were established in 12 shade houses that provided a light availability near 5 percent of full sunlight. We randomly assigned new shade cloth densities to these 12 houses so that four provided 70 percent of full sunlight, four provided 37 percent of full sunlight, and four provided 5 percent of full sunlight. Our research objective was to quantify survival, stem length, stem diameter, and ramet production of pondberry shrubs released from a heavily shaded environment. Three growing seasons after release, shrub survival averaged 71 percent, and stem length averaged 113 cm regardless of assigned light level. Pondberry released into 70 or 37 percent light developed 58 percent larger stem diameters than those maintained under 5 percent light. Plants released into 70 or 37 percent light also produced 274 percent more ramets than those grown under 5 percent light. These results indicate pondberry growth should respond positively to silvicultural practices such as midstory or overstory canopy treatments that increase light availability in the understory of floodplain forests. However, we speculate that control of competing understory vegetation invigorated by canopy treatment may be necessary to ensure pondberry release.

INTRODUCTION

Pondberry [*Lindera melissifolia* (Walt.) Blume (Lauraceae)] is a deciduous woody shrub endemic to low-lying forests of the southeastern United States (USFWS 1986). In the Lower Mississippi Alluvial Valley (LMAV), this dioecious and rhizomatous plant grows up to 2-m tall on sites flooded for periods ranging from several days to several months. Impacts of deforestation and forest degradation have reduced potential pondberry habitat in the LMAV where it is currently found in isolated colonies within scattered forest patches (Devall and others 2001, Hawkins and others 2010). Consequently, the U.S. Fish and Wildlife Service listed this species as endangered in 1986 (USFWS 1986). A pondberry recovery plan was developed after this listing (USFWS 1993). The plan emphasized the need to study pondberry biology and ecology and to develop management practices that would promote recovery and conservation of this species.

Our knowledge of pondberry biology and ecology is increasing (Aleric and Kirkman 2005a, 2005b; Connor and others 2007; Devall and others 2001; Echt and others 2006, 2011; Fraedrich and others 2011; Godt and Hamrick 1996; Hawkins and others 2009a, 2009b, 2010, 2011; Lockhart and others 2012, 2013; Wright 1990). However, we do not know the resilience of pondberry survival and growth when plants are established in heavily shaded understories; and we do not know how established pondberry responds to release from heavily shaded conditions. Research presented in this manuscript examines: (1) a 7-year progression of survival, stem length, stem diameter, and ramet production for pondberry established under low light availability; and (2) survival, stem length, stem diameter, and ramet production for pondberry established and raised for 4 years under low light availability then subjected to release into higher light environments. An understanding of how pondberry tolerates shade and responds to release from heavily shaded environments should inform the design of silvicultural practices to recover and conserve this endangered species.

MATERIALS AND METHODS

The study was conducted in the Flooding Research Facility (FRF) in Sharkey County, MS, on the Theodore Roosevelt National Wildlife Refuge Complex (32° 58' N, 90° 44' W). The FRF is situated about 7 km from natural pondberry colonies growing on the Delta National Forest. Mean annual temperature at the FRF averages 17.3 °C with a range from 27.3 °C in July to 5.6 °C in January, and mean annual precipitation averages 1350 mm (WorldClimate 2008). The FRF was constructed on Sharkey soils (very-fine, smectitic and thermic Chromic Epiaquerts) which are shrink-swell clay soils that

[1]Research Forester, Research Forester, and Supervisory Research Plant Pathologist, respectively, USDA Forest Service, Southern Research Station, Stoneville, MS 38776.

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developed from alluvium deposited on slackwater areas.

The FRF is comprised of 12, 0.4-ha rectangular impoundments that can be independently flooded or drained [see Lockhart and others (2006) for more details about design and operation of the FRF]. Pondberry research conducted between 2005 and 2007 called for flooding the FRF impoundments according to three different hydroperiod regimes [see Lockhart and others (2013) for more information about the initial pondberry study]. Additionally, three rectangular shade houses (25.6-m long by 7.3-m wide by 2.4-m tall) were constructed in each impoundment to provide light levels of 70, 37, or 5 percent of full sunlight. Light availability within these shade houses was controlled using neutral density shade cloth (PAK Unlimited, Inc., Cornelia, GA). Following completion of the 2005-2007 research, established pondberry plants were maintained under their assigned light levels, but flooding treatments were suspended. Prior to the 2009 growing season, we implemented a new study (named the pondberry release study) that utilized plants established in the 12 houses assigned the 5 percent light level (one house in each of the 12 impoundments). We randomly assigned new shade-cloth densities to these 12 shade houses to emulate a release treatment such that four houses provided 70 percent of full sunlight, four houses provided 37 percent of full sunlight, and four houses provided 5 percent of full sunlight.

Pondberry plants established in the FRF included 20 LMAV genotypes that were outplanted during April 2005. Ninety-six, single-stemmed stecklings (rooted cuttings) were planted on a 1.2- by 1.2-m spacing in each shade house. Stecklings were randomly assigned in each shade house so that each genotype was well represented. Transplants were maintained free of competing vegetation for the duration of the research by hand hoeing and directed applications of herbicides.

Plants were measured immediately after planting in 2005 and at the end of each growing season through 2011. Measurements included stem length, stem diameter, and a count of ramets produced by the original steckling. Stem length was measured from the groundline to the base of the terminal bud. Stem diameter was measured 2 cm above the groundline with dial calipers. Two diameter measurements perpendicular to each other were collected to calculate an average diameter for each stem.

Data were analyzed according to a completely randomized design with light availability as a fixed-effect treatment. Analyses were conducted using PROC GLM in SAS 9.3 (SAS Institute, Inc., Cary, NC). Response variables analyzed included survival (percent), length (cm) and diameter (mm) of the longest stem, and ramet number for each shrub. Data transformation was used as needed to normalize model residual errors. Statistical significance among treatment means for each response variable was determined at $\alpha = 0.05$. Duncan's Multiple Range Test was used for mean separations and untransformed data are presented in all figures.

RESULTS

Pondberry mortality was greatest the first 3 years following planting as survival for plants declined to 73 percent (fig. 1). Mortality stabilized following the third growing season, and survival remained near 71 percent for all treatment levels 7 years after planting ($p = 0.53$; fig. 1).

Stem length of pondberry planting stock averaged 21 cm at outplanting, and increased 371 percent to 99 cm when receiving 5 percent light for seven growing seasons (fig. 2). Release of pondberry into higher light environments did not influence stem length until the third growing season after release (fig. 2). At that time, the longest stems on shrubs receiving 37 percent light were 23 percent longer than those receiving 70 or 5 percent light ($p = 0.01$; fig. 2).

Pondberry stem diameter averaged 1.5 mm at outplanting (fig. 3). Stem diameter increased 360 percent to 6.9 mm through 7 years of growth under 5 percent light (fig. 3). Pondberry stem diameter responded quickly to increased light

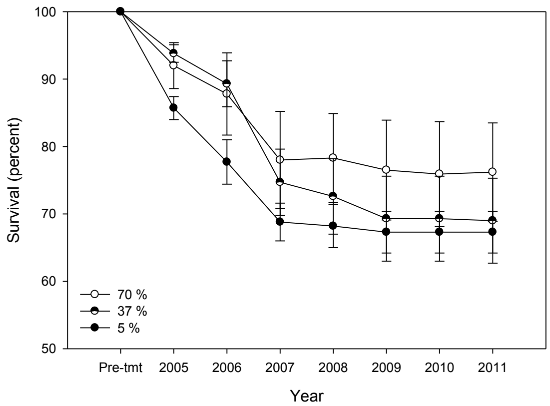

Figure 1--Pondberry survival by light availability at the Flooding Research Facility, Sharkey County, MS. Plants were grown in 5 percent light from 2005 through 2008, though they are shown by release treatments for information purposes. Vertical bars represent one standard error.

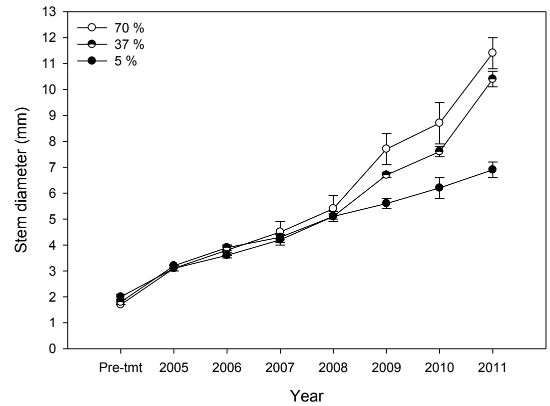

Figure 3--Pondberry stem diameter by light availability at the Flooding Research Facility, Sharkey County, MS. Plants were grown in 5 percent light from 2005 through 2008, though they are shown by release treatments for information purposes. Vertical bars represent one standard error.

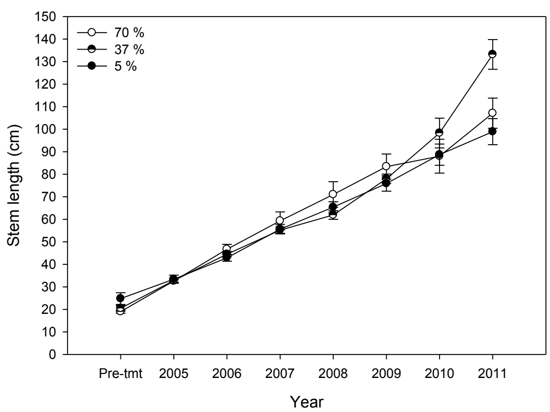

Figure 2--Pondberry stem length by light availability at the Flooding Research Facility, Sharkey County, MS. Plants were grown in 5 percent light from 2005 through 2008, though they are shown by release treatments for information purposes. Vertical bars represent one standard error.

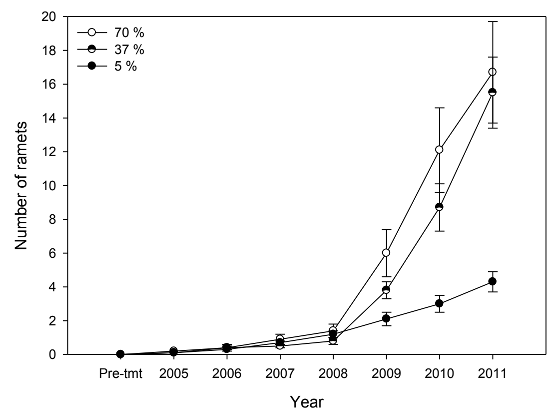

Figure 4--Number of pondberry ramets produced and maintained by light availability at the Flooding Research Facility, Sharkey County, MS. Plants were grown in 5 percent light from 2005 through 2008, though they are shown by release treatments for information purposes. Vertical bars represent one standard error.

availability, as shrubs released to 70 or 37 percent light grew 290 percent more in diameter the first year after release than those maintained under 5 percent light. Three years after release, pondberry stem diameter was 58 percent larger when receiving 70 or 37 percent light compared to shrubs receiving 5 percent light ($p < 0.01$; fig. 3).

The number of ramets produced and maintained by pondberry plants receiving 5 percent light increased from 0.1 per plant during the first growing season to 4.3 per plant by the end of the seventh growing season (fig. 4). As with stem diameter, ramet production quickly responded to increased light availability. During the first growing season after release, shrubs released into 70 or 37 percent light produced and maintained 322 percent more ramets than those maintained under 5 percent light. Three years after release, the number of ramets observed on pondberry shrubs receiving the two highest light levels had increased from an average of 1.1 to 16.1 ramets per plant. At this time, shrubs maintained under 5 percent light had 73 percent fewer ramets per plant than those released into 70 or 37 percent light ($p = 0.01$; fig. 4).

DISCUSSION

Hawkins and others (2009b) reported that pondberry colonies found in the LMAV were associated with forest compositions and structures reflective of hydrologic regime, topography, historical disturbance, and an absence of recent disturbance. Common overstory tree species found in LMAV forests supporting pondberry colonies included Nuttall oak (*Quercus nuttallii* Palmer), willow oak (*Q. phellos* L.), overcup oak (*Q. lyrata* Walt.), American elm (*Ulmus americana* L.), green ash (*Fraxinus pennsylvanica* Marsh.), sweetgum (*Liquidambar styraciflua* L.), and common persimmon (*Diospyros virginiana* L.). Red maple (*Acer rubrum* L.), sugarberry (*Celtis laevigata* Willd.), deciduous holly (*Ilex decidua* Walt.), green ash, and American elm were among the midstory species abundant in these forests (Hawkins and others 2009b). The stratified overstory and midstory canopies characteristic of these forests intercept most ambient sunlight, leaving relatively little available to understory woody plants. For example, ambient light levels less than 7 percent of full sunlight have been routinely observed in mature floodplain forest understories across the southeastern U.S. (Cunningham and others 2011, Jenkins and Chambers 1989, Lhotka and Loewenstein 2006, Lockhart and others 2000).

We found that pondberry was able to survive and grow for seven growing seasons in a low light environment. Plants receiving 5 percent light had a slow but steady increase in stem length, stem diameter, and ramet production. Pondberry acclimated to this low light environment exhibited typical shade leaf-blade characteristics, including dark green leaf blades with large surface areas and a horizontal leaf-blade display that minimizes leaf-blade overlap and maximizes light gathering capabilities (Aleric and Kirkman 2005a, Lockhart and others 2012). Other research conducted at our study site confirmed that plants receiving 5 percent light had lower total leaf mass, thinner leaf blades, lower stomatal densities, thinner palisade layers, and thinner spongy mesophyll layers than plants grown under higher light environments (data not shown). Leaves acclimated to low light availability were also able to maintain low, but positive, rates of net photosynthesis through the growing season (data not shown). Findings from this study confirmed that pondberry can survive and exhibit positive growth in light environments characteristic of heavily shaded understories in bottomland hardwood forests (Aleric and Kirkman 2005a, Lockhart and others 2012).

Research into silvicultural practices that increase the amount of ambient sunlight reaching the understory of floodplain forests has focused on applications to enhance oak reproduction (Cunningham and others 2011, Guttery and others 2011, Lhotka and Loewenstein 2006, Lockhart and others 2000). Yet results should be equally as relevant to other understory species. Deadening midstory canopy trees, through mechanical or chemical operations, will increase the amount of ambient sunlight reaching the forest understory. Researchers have shown that control of midstory canopy competition can increase light levels reaching the forest understory to between 25 and 40 percent of full sunlight (Cunningham and others 2011, Guttery and others 2011, Lockhart and others 2000).

This research documents pondberry responding favorably to increases in light availability following several years of growth suppression under low light environments. Enhanced stem diameter and ramet production were the earliest responses observed and were recorded during the first growing season for plants released into 70 or 37 percent light. Stem length also increased for plants receiving 37 percent light, but this response was not observed until three growing seasons after release from 5 percent light. The increases in stem length, stem diameter, and ramet production for pondberry plants receiving 70 or 37 percent light appeared to be supported by morphological and physiological acclimation of shrubs to the higher light environments.

Our observations of pondberry in the current experiment indicate this plant can tolerate relatively low light availability for several years then acclimate to and improve growth when released into a higher light environment. Light availability observed under the 37 percent light level of this experiment is within the range of light levels reported by researchers following deadening of midstory canopy trees in floodplain forests (Cunningham and others 2011, Guttery and others 2011, Lockhart and others 2000). Thus, our observations provide a starting point for future research aimed at developing silvicultural practices to enhance understory light availability for pondberry recovery and restoration in bottomland hardwood forests.

Glitzenstein (2007) reported on an earlier effort to restore a declining pondberry population in South Carolina. He observed a rapid increase in the number of pondberry stems and the sum of pondberry stem lengths 2 years after established shrubs were released with partial removal of overstory and midstory trees. However, practitioners understand that release of competing vegetation can be a primary concern when performing practices that increase understory light availability. Glitzenstein (2007) accounted for this by clipping competing vegetation during the release treatment, but the effects of this practice on competing vegetation were not reported.

In our study, all vegetation growing in pondberry plots was controlled with mechanical or chemical treatments, so we cannot speculate how pondberry would respond relative to competing vegetation. Natural pondberry colonies in the LMAV occur in areas subject to seasonal soil flooding, and researchers have hypothesized that the flooding likely reduces understory plant competition (Wright 1990). It is reasonable to speculate that vegetation control practices may be necessary to ensure pondberry survival and growth where soil flooding is not expected to reduce competing vegetation invigorated by canopy treatment.

Our research confirmed that pondberry is capable of tolerating the heavily shaded conditions characteristic of understories in many mature bottomland hardwood forests. Though this species was able to persist and even grow under heavy shade, we demonstrated that above-ground growth was suppressed by low light availability. Following 4 years of suppression beneath heavy shade, pondberry growth increased after simulated release into higher light environments. The responses to release we observed indicate that pondberry would likely react favorably to canopy disturbance created through silvicultural practices. Our findings support the premise that active management could be used to promote vigor and therefore sustainability of extant pondberry colonies. Future research should focus on development of silvicultural practices that provide for the biological requirements of this endangered species.

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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.

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.

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: 984-992.

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.

Echt, C.S.; Demeer, D.; Gustafson, D. 2011. Patterns of differentiation among endangered pondberry populations. Conservation Genetics. 12: 1015-1026.

Echt, C.S.; Demeer, D.; Nelson, C.D. 2006. Microsatellites for *Lindera* species. Molecular Ecology Notes. 6: 1171-1173.

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.

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.; 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.A.; 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 Research 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 the 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.

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.; Gardiner, E.S.; Leininger, T.D. [and others]. 2006. Flooding facility helps scientists examine the ecophysiology of floodplain species used in bottomland hardwood restorations. Ecological Restoration. 23: 151-157.

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.; 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.; 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.

USFWS [U.S. Fish and Wildlife Service]. 1986. Endangered and threatened wildlife and plants: determination of endangered status for *Lindera melissifolia*. Federal Register. 51: 27495-27500.

USFWS [U.S. Fish and Wildlife Service]. 1993. Recovery plan for pondberry (*Lindera melissifolia*). Atlanta: U.S. Department of the Interior Fish and Wildlife Service. 56 p.

WorldClimate. 2008. Rolling Fork, Sharkey County, Mississippi USA. http://www.worldclimate.com. [Date accessed: September 6, 2009].

Wright, R.D. 1990. Photosynthetic competence of an endangered shrub, *Lindera melissifolia*. Proceedings of the Arkansas Academy of Science. 44: 118-120.

GREENHOUSE GAS FLUXES AND ROOT PRODUCTIVITY IN A SWITCHGRASS AND LOBLOLLY PINE INTERCROPPING SYSTEM FOR BIOENERGY PRODUCTION

Paliza Shrestha, John R. Seiler, Brian D. Strahm, Eric B. Sucre, and Zakiya H. Leggett[1]

This study is part of a larger collaborative effort to determine the overall environmental sustainability of intercropping pine (*Pinus taeda* L.) and switchgrass (*Panicum virgatum* L.), both of which are promising feedstock for bioenergy production in the Lower Coastal Plain in North Carolina. We measured soil CO_2 efflux (R_S) every six weeks from January 2012 to March 2013 in four-year-old monoculture and intercropped stands of loblolly pine and switchgrass (Fig. 1). R_S is primarily the result of root respiration (R_A) and microbial decomposition of organic matter (R_H) releasing CO_2 as a by-product and is an important and large part of the global carbon (C) cycle. Accurate estimates of the two components of total soil respiration (R_S) are required as they are functionally different processes and vary greatly spatially and temporally with species composition, temperature, moisture, productivity, and management activities. We quantified R_A and R_H components of R_S by using a root exclusion core technique based on root carbohydrate depletion, which eliminates R_A within the cores over time. We determined the relationship between R_S, R_A and R_H measurements and roots collected from the cores. We took fresh soil cores in July 2012 to compare root productivity of loblolly pine and switchgrass in monoculture versus the co-culture. Additionally, CH_4 and N_2O fluxes were monitored quarterly using vented static chambers. Pure switchgrass had significantly higher R_S rates (July, August, September), root biomass and root length in the top 0-35 cm relative to switchgrass in the co-culture, while loblolly pine with and without switchgrass had no significant changes in R_S and roots (Table 1). Correlations between R_A and roots showed significantly positive correlation of R_A to grass root biomass ($r = 0.37$, $p \leq 0.001$), fine ($r = 0.26$, $p \leq 0.05$) and medium root surface area ($r = 0.20$, $p \leq 0.1$). The estimated portions of R_S attributed to R_A in the intercrop stand were 31% and 22% in the summer and fall, respectively. No significant treatment differences were observed in either CH_4 or N_2O flux. Our study indicates a decrease in switchgrass root productivity in the intercropped stand versus the monoculture stand which could account for differences in the observed R_S.

[1]Graduate Student, Professor, and Assistant Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forestry and Environmental Conservation, Blacksburg, VA 24061; and Sustainability Scientists, Weyerhaeuser Company, Southern Timberlands Research and Development, Vanceboro, NC 28586.

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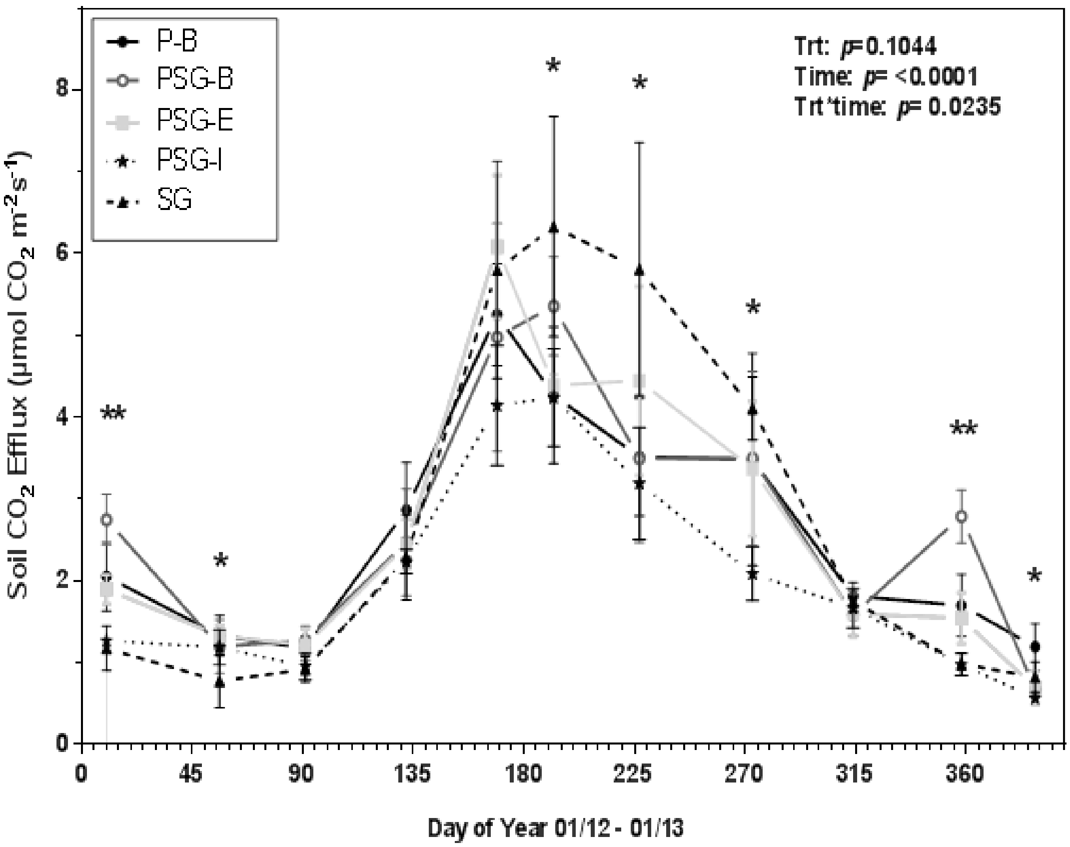

	Jan10	Feb25	Mar30	May10	Jun16	Jul10	Aug14	Sep15	Oct26	Dec09	Jan19
P-B	b	a	a	a	a	b	ab	ab	a	b	a
PSG-B	a	ab	a	a	a	ab	ab	ab	a	a	b
PSG-E	b	a	a	a	a	b	ab	ab	a	bc	b
PSG-I	c	ab	a	a	a	b	b	b	a	c	b
SG	c	b	a	a	a	a	a	a	a	c	ab

Figure 1--Mean soil CO_2 efflux rates (μmol m^{-2} s^{-1}) measured approximately every 6 weeks between January 10, 2012 to January 20, 2013 in a 4-year-old switchgrass and loblolly pine agroforestry system on the lower coastal plain of North Carolina. Error bars represent ±1 standard error from the mean. Stars indicate sampling dates with significant differences between treatments as determined using repeated measures analysis (α = 0.10). The accompanying matrix represents mean separation using Tukey-Kramer HSD where different letters within each treatment date indicate significant differences. Terms with a single asterisks (*) are significant at α = 0.1 level and double asterisks (**) at α = 0.05. P represents traditional pine treatments, SG represents flat planted switchgrass, and PSG represents pine intercropped with switchgrass. Additional treatment designations indicate the microtopographical position of the sample location where B represents the bedded row, I represents the interbed space, and E represents the edge where an aboveground transition from switchgrass to pine can be observed.

Table 1--Switchgrass and loblolly pine root biomass, average root length, and average root surface area in intercrop versus pure stands in 0 to 35 cm soil depth measured in July on the lower coastal plain of North Carolina. Means are followed by ±1 standard errors.

Measurement	Type[a]	Pine roots[b]	Switchgrass roots[b]	Total
Biomass (*g m*$^{-3}$)	P-B	3262[a] ± 501.1	NA	3262 ± 501.1
	PSG-B	3835[a] ± 628.7	182.8[b] ± 51.36	4018 ± 680.06
	PSG-E	1666[b] ± 197.0	1247[b] ± 424.4	2913 ± 621.4
	PSG-I	325.9[c] ± 56.28	1481[b] ± 418.7	1807 ± 475.0
	SG	411.0[c] ± 178.2	5359[a] ± 1842	5770 ± 2020
Length (*cm dm*$^{-3}$)	P-B	2335[a] ± 532	NA	2335 ± 532
	PSG-B	1045[b] ± 170.7	1150[c] ± 330.1	2195 ± 500.8
	PSG-E	844.1[bc] ± 60.38	2941[bc] ± 565.7	3785 ± 626.1
	PSG-I	312.0[cd] ± 56.65	4503[ab] ± 1163	4815 ± 1220
	SG	158.4[d] ± 76.94	7665[a] ± 640.9	7823 ± 717.9
Surface area (*cm*^{2} *dm*$^{-3}$)	P-B	35.76[a] ± 7.287	NA	35.76 ± 7.287
	PSG-B	19.50[ab] ± 3.457	10.47[c] ± 2.787	29.97 ± 6.244
	PSG-E	16.03[bc] ± 1.712	31.50[bc] ± 6.147	47.53 ± 7.859
	PSG-I	6.188[cd] ± 1.239	42.02[b] ± 9.801	48.21 ± 11.04
	SG	2.900[d] ± 1.270	90.41[a] ± 4.807	93.31 ± 6.077

[a]P-B = pine bed; PSG-B = pine + switchgrass bed, PSG-E = pine + switchgrass edge, PSG-I = pine + switchgrass interbed; SG = switchgrass.
[b]Numbers in the same column followed by different letters are significantly different at $\alpha = 0.05$.

DISTRIBUTION OF LONGLEAF PINE IN THE SOUTHEASTERN UNITED STATES AND ITS ASSOCIATION WITH CLIMATIC CONDITIONS

Zhen Sui, Zhaofei Fan, Michael K. Crosby, and Xingang Fan[1]

Abstract--Longleaf pine (*Pinus palustris* Mill.) has irreplaceable ecological value in the southeastern United States. However, longleaf pine-grassland ecosystems have been dramatically declining since European settlement. From the aspect of longleaf pine restoration and management, this study calculated longleaf pine importance values in each southern county and then conducted preliminary analysis based on spatial autocorrelation statistics and quantile regression. This study estimated current longleaf pine spatial distribution characteristics and the relationship between species dominance and climatic conditions. Even though longleaf pine has declined across counties over the past 40 years, clusters remain in the states of Florida, Georgia, North Carolina, Alabama, Louisana, South Carolina, and Mississippi. Quantile regression modeling predicted broader levels than conventional least square regression.

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.)-grassland ecosystems have existed in the coastal plain for thousands of years. Historical climatic conditions influenced longleaf pine distribution (Van Lear and others 2005). Longleaf pine is one of the most important tree species in the southeastern United States because of its economic and ecological value (Brockway and others 2005, Friedenberg and others 2007, Gilliam and Platt 1999, Johnsen and others 2009, Roise and others 1991). Unfortunately, the extent of longleaf pine ecosystems has dramatically declined (Outcalt and Sheffield 1996, Van Lear and others 2005). Before European settlement, longleaf pine forests occupied over 60 million acres in the southeastern United States; only about 3 million acres remain. The loss was due to logging, land use conversion, fire exclusion, and lack of regeneration. Longleaf pine forests have become the third most endangered ecosystem in the United States (Noss and others 1995). Many approaches have been proposed for the restoration of longleaf pine, such as maintaining the overstory of longleaf pine, reducing midstory hardwood trees, reducing non-native species, and re-establishing native plant and animals (Varner and others 2005).

To assist longleaf pine-grassland restoration, ecological factors that drive species distributional response need to be considered. Iverson and others (1999) applied regression-tree analysis and identified that mean January temperature plays a significant role in affecting longleaf pine importance value in the eastern United States. Samuelson and others (2012) measured leaf physiological traits of southern pines and found that longleaf pine has higher water-use efficiency and greater drought tolerance than other pines. In general, climatic variables have the most influence in species spatial pattern at large scales (Woodward 1987), while more local effects, such as soil factors, determine the local variations in distribution (Iverson and others 1999). However, few studies investigated the relationship between longleaf pine distribution and climate effects. The objectives of this study are: (1) to assess spatial distribution of longleaf pine by decade over the past 40 years; and (2) to determine the relationship between longleaf pine importance value and minimum temperature, maximum temperature, and annual precipitation. Such information will assist future restoration efforts for longleaf pine in various climate zones and will help in planning of longleaf pine restoration.

METHODS

The study area includes almost 2,360 counties in 13 southeastern states (fig. 1). The importance values of longleaf pine by county were calculated using the Forest Inventory and

[1]Ph.D. Student, Assistant Professor, and Post-doctoral Research Associate, respectively, Mississippi State University, College of Forest Resources, Department of Forestry, Mississippi State, MS 39762; and Assistant Professor, Western Kentucky University, Department of Geography and Geology, Bowling Green, KY 42101.

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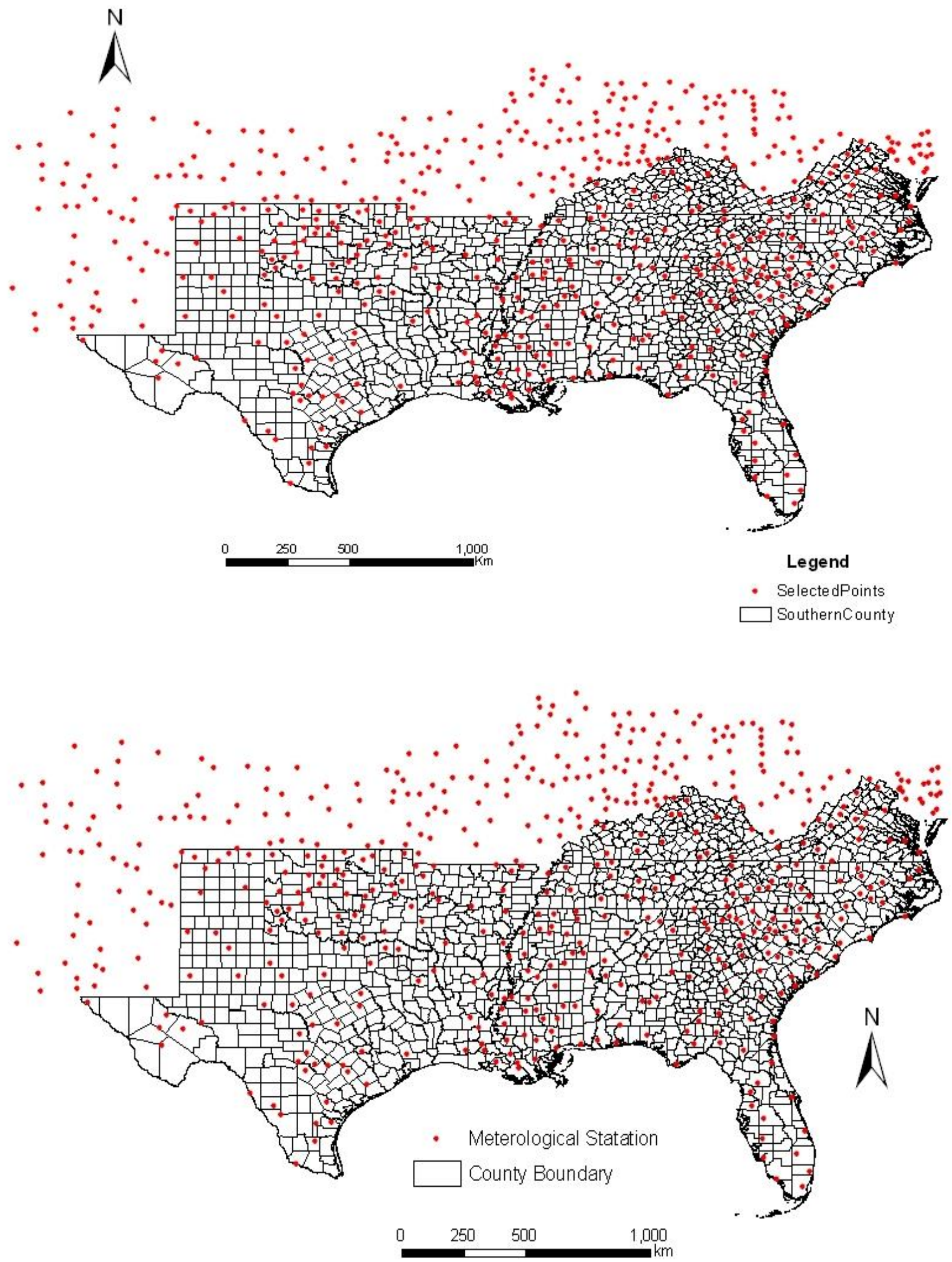

Figure 1--Study area and locations of selected meteorological stations.

Analysis (FIA) database. We divided the dataset into four decades (the 1970s, 1980s, 1990s, and 2000s). If one plot was measured twice in one decade, the latest measurements were used in our calculations.

Observed meteorological data is from the U.S. Historical Climatology Network (USHCN version 2) and contains monthly mean maximum temperature, mean minimum temperature, and total precipitation since 1897. A total of 526 meteorological stations were selected within 13 southeastern states as well as adjacent states that had data from 1970 to 2009. We then interpolated station data into climatic surfaces (raster) in ArcGIS by the Inverse Distance Weight (IDW) algorithm. Choosing stations from the adjacent states can reduce the errors from spatial interpolation of climate variables. After interpolation, zonal statistics in ArcGIS were used to aggregate climate surfaces to each county. The zonal layer was the county boundary, which was obtained from The National Atlas of the United States of America (www.nationalatlas.gov). Lastly, importance values were calculated and paired with three climatic variables by county and decade for further analysis.

Global Moran's I index estimates overall degrees of spatial autocorrelation. The index was applied in this study to test spatial autocorrelation in order to evaluate spatial clustering patterns. This index uses the randomization assumption to test for normality. In general, the values of Moran's I range from -1 to 1. Negative values of Moran's I indicate negative spatial autocorrelation; positive values of Moran's I indicate positive spatial autocorrelation; a zero value indicates no spatial autocorrelation. Moran's I is calculated by

$$I = \frac{N}{S_o} \frac{\sum_i \sum_j w_{ij}(x_i - u)(x_j - u)}{\sum_i (x_i - u)^2}$$
$$S_o = \sum_i \sum_j w_{ij} \tag{1}$$

where:
N is the number of counties;
w_{ij} is the element in the spatial weight matrix corresponding to the observation pair (i, j);
x_i and x_j are observations for counties i and j, respectively;
u is observed mean over all counties.

Local spatial autocorrelation statistics, or G-statistic, provide estimates of dependency relationships in different areas. In this study, the local G-statistic was used to make autocorrelation comparisons in different neighborhoods and test the statistical significance of local clusters of the importance of longleaf pine over the 4 decades. Detailed information of local G-statistic can be found in Getis and Ord (1992).

Quantile regression was used to evaluate how different parts of the dependent variable respond to predictors, in that any quantile of a response is able to be fitted by respective linear models (Cade and others 1999). Quantile regression not only specifies the predictor as the conventional regression model but also has more ecological rationale without abrupt thresholds and unexpected shapes (Austin 2007). Considering longleaf pine restoration, the primary goal is more stems and large trees. In this study, quantile regression was chosen to compare with conventional least square regression to estimate the different levels of responses of importance value to the climatic conditions.

RESULTS AND DISCUSSION

Even though the declining dominance of longleaf pine in the southeastern United States has been reported (Oswalt and others 2012, Outcalt and Sheffield 1996, Van Lear and others 2005), few studies calculate important values at the county level. We found that longleaf pine existed in 778, 653, 668, and 649 out of 2,360 counties in the 1970s, 1980s, 1990s, and 2000s, respectively. Thus, numbers of counties occupied by longleaf pine generally decreased in the past 40 years despite a slight rebound in the 1990s. Table 1 shows the tendency of longleaf pine importance values at different quantile levels. Overall, our results showed a general decreasing trend in both the number of counties with longleaf pine and the importance values at various quantiles over the past 40 years.

Nearby counties overall have similar longleaf pine occupation conditions. For the calculation of longleaf pine importance value distribution in the southeastern United States, the results of the global autocorrelation statistics by global Moran's I are 0.2717, 0.2466, 0.3017, and 0.2292 for the 1970s, 1980s, 1990s, and 2000s, respectively. The global Moran's test related to the importance values of longleaf pine are statistically significant (p-value < 0.05) and indicate spatial heterogeneity. For the local G-statistics (fig. 2), counties shaded in red (hotspots) are spatial clusters with a 95 percent significance level from a two-tailed normal distribution. In general, most of identifed hotspots were mainly distributed in Florida, Georgia, North Carolina, Alabama, Louisana, South Carolina, and Mississippi. Hotspots provide ecological insights for longleaf pine restoration because detected counties have relatively greater degrees of longleaf pine dominance than other counties. In the future, more detailed survey of these identified hotspots may reveal suitable habitats and preservation refuges for longleaf pine restoration.

Table 1--Statistics of longleaf pine importance value (percent; 0^{th} = minimum, 100^{th} = maximum) at various quantiles by 4 decades

	0^{th}	25^{th}	50^{th}	75^{th}	100^{th}
1970s	0.07	2.00	6.00	14.00	34.00
1980s	0.16	2.00	5.00	9.00	29.00
1990s	0.13	1.00	4.00	7.00	28.00
2000s	0.09	1.00	4.00	8.00	28.00

Multiple covariates regression was applied under 19 distinct quantiles ranging from 5^{th} to 95^{th} in order to compare estimation coefficients of quantile regression with conventional linear regression by the least square estimate. Figure 3 presents the change of intercept and partial slope with estimated conditional quantiles for three climatic variables (minimum temperature, maximum temperature, and annual precipitation) predicting longleaf pine importance value. In each panel, the solid red line shows the least squares regression line; the dashed red lines represent the 95 percent confidence interval of the least squares regression. The black dots are estimates at each conditional quantile, and the gray areas are the 95 percent confidence interval for the quantiles. For each covariate, these point estimates may be interpreted as the impact of a one-unit change of the covariate on longleaf pine important value holding other covariates fixed. From figure 3, the quantile regression estimates are outside the confidence interval of the conventional least square estimates. This indicates that the ordinary regression performs well in predicting the relationship between longleaf pine importance values and given variables of minimum

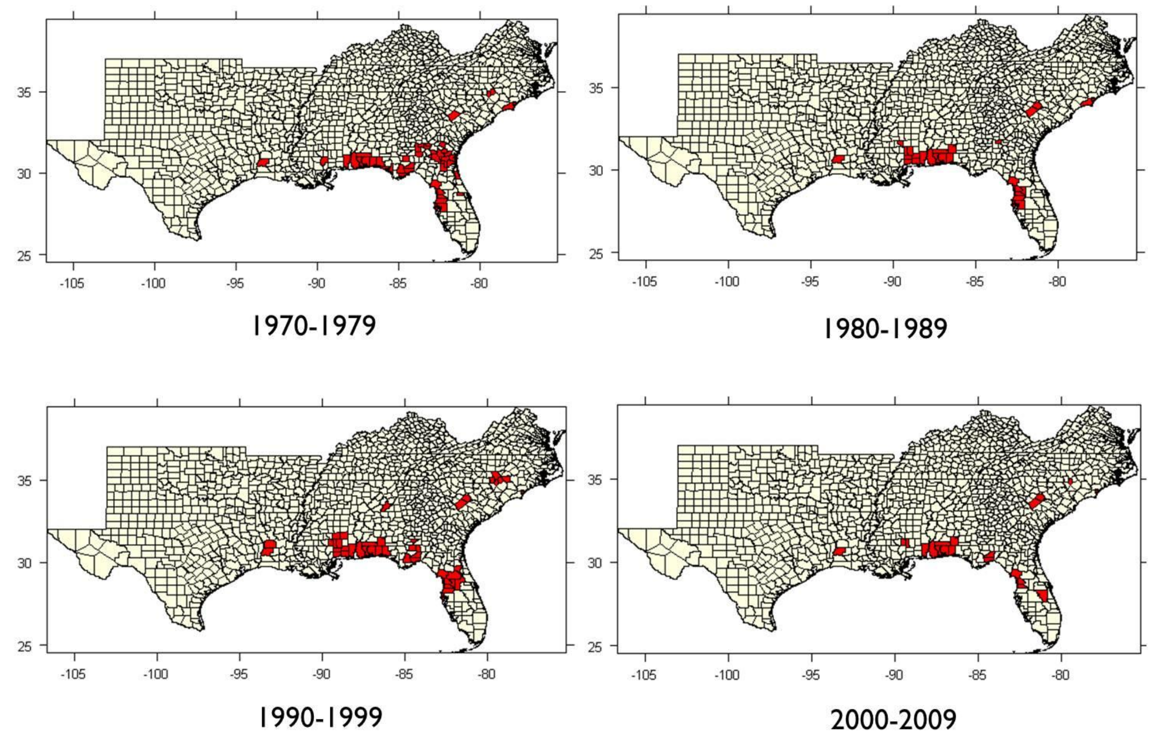

Figure 2--Shaded counties representing spatial clusters (hotspots) of longleaf pine dominance over the past 40 years.

temperature, maximum temperature, and annual precipitation at the quantiles of 60^{th}-75^{th}, 60^{th}-80^{th}, and 50^{th}-75^{th}, respectively. We observed that the ordinary regression only represents the center of data distribution but fails in presenting other parts of the response distribution. Unfortunately, the center approximation is not consistent with the ecological theory of limiting factors because real limiting function is often reflected by the upper or lower boundaries. Compared to conventional regression, quantile regression is able to serve as a useful tool with considering various levels of the response. For the future management, quantile regression is helpful for identifying high dominance of longleaf pine associated with climatic conditions. Thus, quantile regression is a promising statistical method in selecting locations for high restoration success under changing environment.

CONCLUSIONS

Based on the calculation in this study, the numbers of counties with longleaf pine are decreasing over the past 4 decades in the southeastern United States. In addition, the importance values of longleaf pine are declining at various quantiles of the 0th (minimum), 25th, 50th, 75th, and 100th (maximum). Analyzing clusters by spatial autocorrelation statistics, we found that, in general, most of spatial clusters are distributed in Florida, Georgia, North Carolina, Alabama, Louisana, South Carolina, and Mississippi which can serve as the source of longleaf pine refuge and experimental sites for further detailed studies.

Quantile regression covered broader longleaf pine dominance levels than conventional least square regression in assessing the relationships between longleaf pine dominance and climatic variables of minimum temperature, maximum temperature, and annual precipitation. Thus, quantile regression could help with predicting longleaf pine dominance under limiting factors in future studies. In addition, we are able to utilize quantile regression modeling to evaluate potential restoration success (e.g., how much longleaf pine dominance can be achieved) before taking action.

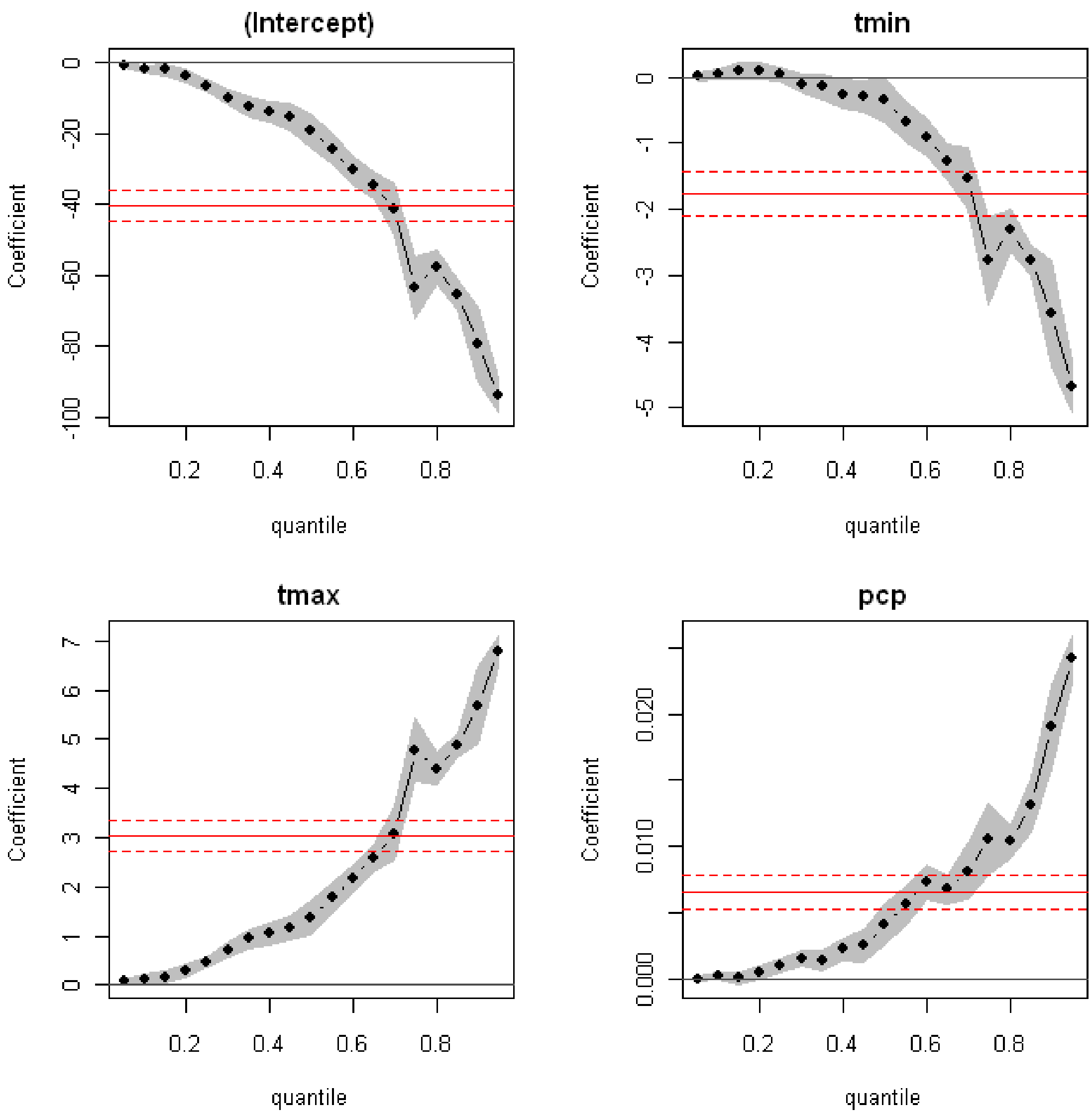

Figure 3--Coefficients of the quadratic terms for multiple quantiles. The variable names are: tmin = mean minimum temperature; tmax = mean maximum temperature; and pcp = annual precipitation.

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LITERATURE CITED

Austin, M. 2007. Species distribution models and ecological theory: a critical assessment and some possible new approaches. Ecological Modelling. 200(1-2): 1-19.

Brockway, D.G.; Outcalt, K.W.; Tomczak, D.J.; Johnson, E.E. 2005. Restoration of longleaf pine ecosystems. Gen. Tech. Rep. SRS-83. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 34 p.

Cade, B.S.; Terrell, J.W.; Schroeder, R.L. 1999. Estimating effects of limiting factors with regression quantiles. Ecology. 80: 311-323.

Friedenberg, N.A.; Whited, B.M.; Slone, D.H. [and others]. 2007. Differential impacts of the southern pine beetle, *Dendroctonus frontalis*, on *Pinus palustris* and *Pinus taeda*. Canadian Journal of Forest Research. 37: 1427-1437.

Getis, A.; Ord, J.K. 1992. The analysis of spatial association by use of distance statistics. Geographical Analysis. 24: 189-206.

Gilliam, F.S.; Platt, W.J. 1999. Effects of long-term fire exclusion on tree species composition and stand structure in an old-growth *Pinus palustris* (longleaf pine) forest. Plant Ecology. 140: 15-26.

Iverson, L.R.; Prasad, A.M.; Hale, B.J. Sutherland, E.K. 1999. Atlas of current and potential future distributions of common trees of the eastern United States. Gen. Tech. Rep. NE-265. Radnor, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station. 245 p.

Johnsen, K.H.; Butnor, J.R.; Kush, J.S. [and others]. 2009. Hurricane Katrina winds damaged longleaf pine less than loblolly pine. Southern Journal of Applied Forestry. 33: 178-181.

Noss, R.F.; LaRoe, E.T.; Scott, J.M. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. http://noss.cos.ucf.edu/papers/Noss%20et%20al%201995.pdf. [Date accessed: May 27, 2013]

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.

Outcalt, K.W.; Sheffield, R.M. 1996. The longleaf pine forest: trends and current conditions. Resour. Bull. SRS-9. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 28 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: 88-92.

Samuelson, L.J.; Stokes, T.A.; Johnsen, K.H. 2012. Ecophysiological comparison of 50-year-old longleaf pine, slash pine and loblolly pine. Forest Ecology and Management. 274: 108-115.

Van Lear, D.H.; Carroll, W.D.; Kapeluck, P.R.; Johnson, R. 2005. History and restoration of the longleaf pine-grassland ecosystem: implications for species at risk. Forest Ecology and Management. 211: 150-165.

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.

Woodward, F.I. 1987. Climate and plant distribution. Cambridge, UK: Cambridge University Press. 174 p.

IS THERE A MORPHOLOGICAL OR PHYSIOLOGICAL EXPLANATION FOR THE DRAMATIC INCREASE IN HYBRIDIZATION BETWEEN LOBLOLLY AND SHORTLEAF PINE?

Rodney E. Will, Curtis J. Lilly, John F. Stewart, C. Dana Nelson, and Charles G. Tauer[1]

Hybrids between shortleaf pine (*Pinus echinata* Mill.) and loblolly pine (*P. taeda* L.) have dramatically increased since the 1950s (Stewart and others 2012). Fire suppression, planting nonnative seed sources, and other anthropogenic activities have the potential to break down ecological barriers that previously kept these species from interbreeding (Tauer and others 2012). We compared artificial F1 shortleaf x loblolly pine hybrids to their parents in a 3-year study in Oklahoma. Loblolly and hybrid seedlings had superior establishment and growth rates compared to shortleaf pine. When topkilled before and during the third growing season using a combination of topclipping and girdling with fire, resprouting was greatest in shortleaf (94 percent) and lower in hybrids (77 percent) and loblolly pine (35 percent). Number of sprouts for surviving seedlings followed the same pattern, 32, 23, and 12 percent, respectively, for shortleaf, hybrids, and loblolly pine. Formation of a basal crook, a presumed adaptation to protect dormant buds from fire, was greatest in shortleaf (82 percent) and lower in hybrids (35 percent) and loblolly pine (6 percent). In large part due to the crook, height to the lowest sprout was shortest in shortleaf (4 mm), intermediate in hybrids (8 mm), and greatest in loblolly pine (21 mm). Water-use efficiency of hybrid pine was similar to shortleaf pine and higher than loblolly pine. In the absence of fire, the hybrid seedlings perform at least as well as the parent species. In contrast, shortleaf pine has superior traits related to potential survival following topkill by fire. Fire appears necessary to eliminate hybrids and maintain the genetic integrity of shortleaf pine.

LITERATURE CITED

Stewart J.S.; 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 Genomes. 8: 725-735.

Tauer, C.G.; Stewart, J.F.; Will, R. [and others]. 2012. Hybridization leads to loss of genetic stability in shortleaf pine: unexpected consequences of pine management and fire suppression. Journal of Forestry. 110: 216-224.

[1]Professor, Oklahoma State University, Department of Natural Resource Ecology and Management, Stillwater, OK 74078; Analyst, Campbell Group, Portland, OR 97258; Post-doctoral Research Associate, Oklahoma State University, Department of Natural Resource Ecology and Management, Stillwater, OK 74078; Research Geneticist, USDA Forest Service, Southern Research Station, Saucier, MS 39574; and Researcher (retired), Silverton, OR 97381.

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EFFECTS OF THE SILVICULTURAL INTENSITY ON THE 4-YEARS GROWTH AND LEAF-LEVEL PHYSIOLOGY OF LOBLOLLY PINE VARIETIES

Marco Yanez, John Seiler, and Thomas Fox[1]

The role that genetic improvement plays in the increase of productivity in loblolly pine (*Pinus taeda* L.) in the South has been recognized (McKeand and others 2003). Varietal forestry has the potential to improve the productivity and quality of loblolly pine stands, and higher genetic gains can be achieved in volume and stand uniformity (Zobel and Talbert 1984). However, to achieve this potential productivity, the use of elite genotypes must be linked with silvicultural management. Although the positive growth responses due to intensive silviculture have been reported extensively, much less is known about the physiological processes that drive these responses. It is expected that increasing the genetic uniformity, from open-pollinated (OP) to control mass-pollination (CMP) families and finally to varieties, is associated to a higher uniformity in both growth and physiological processes occurring among the trees. Aspinwall and others (2011) found no relationship among the different level of genetic and phenotypic uniformity. Their results are based in single-tree plot trials, and little information exists about the performance of loblolly pine varieties In monoclonal blocks, which is the standard way to deploy elite genotypes.

In order to gain major insight about the genetic x silviculture interaction on the growth and leaf-level physiology in loblolly pine, a study was established in 2009 on the Virginia Piedmont and North Carolina Coastal Plain of the U.S. Each trial was designed as a split-split plot, with two levels of silviculture (operational and intensive) as the whole plot and six genotypes entries (one OP, one CMP, and four clones) as a split-plot factor. Two of the clones selected have a wide-crown ideotype, and two have a moderately broad crown. Three different planting densities (250, 500, and 750 trees per acre) were arranged as the split-split plot treatments. The study was on 4-year growth data and leaf-level physiology in a subsample of trees in the spacing of 500 trees per acre. Leaf-level physiology was also sampled in the upper and lower crown. We assessed the interaction effects of the genotypes with the silvicultural treatments and site conditions.

Preliminary results indicate that the relation between growth and leaf-level physiology is not clear. There were interaction effects among site by silvicultural treatments and site by genotype. At the Piedmont site, the OP family had the lower averaged-photosynthetic rate (fig. 1). Future analyses will focus on distinguishing differences among crown traits and leaf area of the varieties.

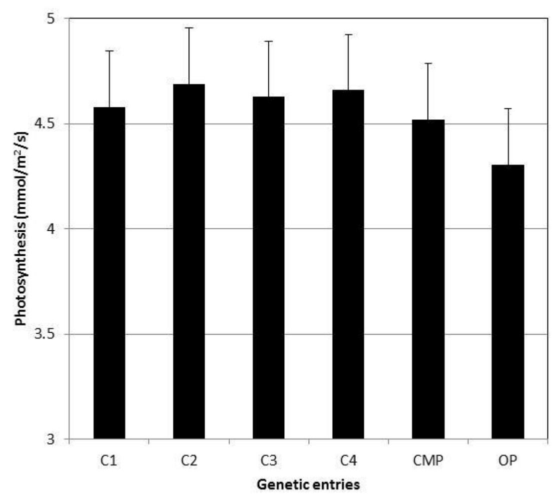

Figure 1—Photosynthetic rates of the different genotypes at the Piedmont site in Virginia.

LITERATURE CITED

Aspinwall, M.; King, J.; Domec, J.; Mckeand, S. 2011. Genetic effects on transpiration, canopy conductance, stomatal sensitivity to vapor pressure deficit, and cavitation resistance in loblolly pine. Ecohydrology. 4(2): 168-182.

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

Zobel, B.; Talbert, J. 1984. Applied forest tree improvement. New York: John Wiley and Sons. 505 p.

[1]Graduate Student and Professors, respectively, Virginia Polytechnic Institute and State University, Department of Forestry and Environmental Conservation, Blacksburg, VA 24061.

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Forest Nutrition

Moderators:

Eric Jokela
University of Florida
School of Forest Resources and Conservation

and

Jeremy Stovall
Stephen F. Austin State University
Arthur Temple College of Forestry and Agriculture

EXPLAINING THE APPARENT RESILIENCY OF LOBLOLLY PINE PLANTATION TO ORGANIC MATTER REMOVAL

Jeff A. Hatten, Eric B. Sucre, Zakiya Leggett, Jason Mack, Scott D. Roberts, Janet Dewey, and Brian Strahm[1]

Abstract -- We utilized 15-year measurements from an organic matter manipulation experiment in a loblolly pine plantation in the Upper Coastal Plain of Alabama to examine the apparent resiliency of a loblolly pine stand to organic matter removal. Treatments included complete removal of harvest residues and forest floor (removed), doubling of harvest residues and forest floor (added), and a standard harvest residue management (reference). Mineral soil and O horizons were sampled and analyzed for carbon (C) and nitrogen (N) using dry combustion. The $\delta^{15}N$ and CuO oxidation biomarkers were assessed in year 15. At year 15, there was no difference in volume between the reference and removed treatments while the added treatment exhibited higher volume. The $\delta^{15}N$ composition of the mineral soil from the removed site was enriched, suggesting that the removed treatment has experienced higher rates of N mineralization. Biomarkers from the CuO oxidation procedure support these assertions. These results have implications for long-term site productivity since recovery of labile N after severe removal of organic matter may be slow, and productivity of subsequent rotations may be negatively impacted if not ameliorated through management.

INTRODUCTION

Forest harvesting intrinsically removes organic matter and associated nutrients; these exports may impact long-term soil productivity and soil carbon (C) stores of managed forests. The Energy Independence and Security Act (EISA) of 2007 set a goal of 36 billion gallons per year of biofuels produced for consumption in the United States by 2022 and, in general, set goals and regulations that reduce U.S. dependency on foreign oil sources (EISA 2007). There are also developing biomass markets for electricity generation. In total, this means more intensive utilization of harvested materials from managed forests. In the U.S., the focus has been on whole-tree harvesting (i.e. boles, branches, tops, needles) (Powers and others 2005) which is associated with higher levels of nutrient and C removal than with standard harvesting operations and may impact long-term soil productivity and soil C stores.

Litter and slash are sources of soil organic matter (SOM), which is a potential long-term sink for atmospheric carbon dioxide. SOM plays a significant role in soil, enhancing cation exchange capacity, soil structure, aeration, water-holding capacity, and soil strength. Decomposing SOM provides the majority of mineralized forms of nitrogen (N) and phosphorus (P). Collectively, these characteristics of SOM play an integral role in sustaining site productivity (Fisher and Binkley 2000).

Most studies of organic matter removal associated with whole-tree harvesting have shown no significant effect on stand productivity (Ponder 2008, Powers and others 2005, Zerpa and others 2010). The North American Long-Term Soil Productivity (LTSP) network found no significant differences in tree productivity at age 10 following whole-tree harvesting and forest floor removal on 18 sites across North America (Powers and others 2005). To date, there has been surprisingly little research on mechanisms of the lack of negative response to stand productivity as a result of a relatively extreme treatment, such as slash and O-horizon removals. This apparent resiliency is the focus of this paper. Huang and others (2011) demonstrated that light-fraction (LF) C concentration can be reduced after whole-tree harvesting of *Pinus radiata* D. Don plantations. While not reported, there was probably a parallel response to LF (or labile) N concentration, suggesting that a stand may resist changes in growth if there is enough labile N to compensate for the losses. However, this response is not universal. Crow and others (2009) report that after 5 years of no above ground litter inputs (belowground inputs still allowed), there was no

[1]Assistant Professor, Oregon State University, Department of Forest Engineering, Resources and Management, Corvallis, OR 97330; Manager of Production Forestry and Biomass Research and Sustainability Scientist, respectively, Weyerhaeuser Company, Vanceboro, NC 28586; Professor and Research Associate, respectively, Mississippi State University, Department of Forestry, Mississippi State, MS 39759; Assistant Research Scientist, University of Wyoming, Analytical Geochemistry Department, Laramie, WY 82072; and Assistant Professor, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061.

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significant effect on LF C concentration relative to the control at the H.J. Andrews Experimental Forest in Oregon. However, when roots and litter were excluded for 5 years, there was a significant decline in LF C and likely a significant impact on labile sources of nutrients associated with organic matter (i.e. N). While there is much uncertainty regarding factors that control resiliency of stand productivity to organic matter removal, we hypothesize that the ability of a soil to be resilient to severe organic matter removals is a site property that could be assessed and monitored by measuring the labile pool of nutrients.

The objective of this research was to assess changes in site productivity by examining the effect of manipulating forest floor and harvest residue inputs. Specifically, we evaluated how these manipulations affected indicators of nutrient cycling and organic biomarkers in the context of intensive forest management.

MATERIALS AND METHODS

Site Description

The Millport Organic Matter Study is located in Lamar County, AL (33° 32' 22.87" N, 88° 77.53" W). The site is located on the Upper Coastal Plain physiographic province. Soils are classified as deep, well-drained Ruston series fine-loam, siliceous, semiactive, thermic Typic Paleudults (NRCS 2011). The nearest weather station with at least 23 years of precipitation and temperature data was in Tuscaloosa, AL (60 kilometers southeast). Average annual temperature from 1987 to 2010 was 16.9 ^{o}C and ranged from 6.3 ^{o}C in January to 27.1 ^{o}C in July (NOAA 2010). Mean annual precipitation was 1320 mm with the wettest month occurring in February (137 mm) and driest month in September (80 mm) (NOAA 2010). Columbus Air Force Base weather station was located 17.7 kilometers northwest of the study site, so it was be used to characterize precipitation and temperatures occurring during the period of study.

A 34-year-old loblolly pine stand was clearcut in 1994 preceding establishment of the current stand. Three treatments were established at the site: added, removed, and reference. Whole trees were harvested, and all slash and the entire forest floor was removed from the removed treatment. The added treatment had bole-only harvest with the slash and forest floor from the removed treatment transferred from an adjacent removed plot (e.g., plot 1 forest floor and residues were transferred to plot 2). This transfer of material preserved the order of O horizons (i.e. Oi, Oe, and Oa). The reference treatment had a bole-only harvest and standard harvest-residue management. A randomized complete block design was utilized containing three treatments and four replicates. Loblolly pine was planted on 4.3- by 3-m spacing in each 0.16-ha plot.

Site and Soil Sample and Data Collection

Stand volume was determined every 3 years through age 15. Diameter at breast height (d.b.h.) was measured on all trees, while 10 randomly selected trees per plot were subsampled to determine average plot height (10 percent of all trees) (Burkhart 1977).

Each plot contained five sampling locations consisting of the four plot corners and the plot center where soil and organic-horizon (O-horizon) data were collected. O-horizon was collected at age 15 from five points per plot and composited. Samples were oven dried at 55 to 60 ^{o}C for 24 hours. O-horizon bulk density was determined by multiplying the surface of the area collected (412 cm^2) by the average thickness of the horizon collected at this point. O-horizon composites were mixed thoroughly, sub-sampled, ground, and analyzed for C and N.

Mineral soils were sampled one time during the fall/winter of 2010-2011 at five locations per plot at 0- to 20-cm, 20- to 40-cm, and 40- to 60-cm depth with a hammer corer. Five samples were composited by plot for a total of 12 soil samples for each depth (48 total soil samples). Soil samples were oven dried at 55 to 60 ^{o}C. Composite samples at each depth were then sub-sampled, ground (Dyna-Crush Soil Grinder, Customer Laboratory Inc.), and analyzed for C and N. Additional soil samples were collected for bulk density determination. Bulk density samples were oven dried at 105 °C for 48 hours.

Density fractionation of mineral soil is a common procedure to determine dynamics of SOM. We utilized it to examine relative lability or recalcitrance of organic forms of N. A low-density (light) fraction in the mineral soil indicates a more labile N pool, while a higher density (heavy) fraction (HF) indicates more recalcitrant forms of N (Six and others 1999). Collected mineral soil samples were separated into LF and HF with a 1.64 g/cm^3 density sodium polytungstate (SPT) solution. Oven-dried soil samples (3 g each) were mixed with 5 g of SPT solution and put in a centrifuge at 3,000 rpm for 10 minutes. After each centrifuge run, LF material floated to the top of the vial and was aspirated and collected in a separate vial. This process was done six times to assure all of the < $1.64 g/cm^3$ particles were collected. The SPT solution containing LF was filtered through 0.47-μm combusted (3 hours at 350 °C) glass-fiber filters, oven dried overnight (55 to 60 °C), processed by randomly punching holes in the filters, and analyzed for total C and N. HF samples were lyophilized and analyzed for total C and N.

Laboratory Analyses

Subsamples of oven-dried O horizon were ground with a Thomas Wiley Laboratory Mill Model 4 using a 60-mesh sieve prior to C and N analysis. Dried mineral soils were subsampled and ground with a mortar and pestle. Organic and mineral samples were analyzed for C and N using dry combustion (Costech ECS 4010).

Samples were subjected to a variety of analyses designed to characterize the composition of the SOM. Details of all analytical techniques used in this study are provided by Hatten and others (2012). The stable isotopic compositions of C and N in the whole soils, and fractions of the 0- to 20-cm horizon were determined by isotope ratio mass spectrometry after high-temperature combustion of pre-acidified samples. In soils, the lighter forms of N are mineralized and taken up or leached, leaving behind the residual heavier N. This leads to a steady enrichment in ^{15}N with depth. We expected that if mineralization rates were impacted by organic matter removal or addition, the alteration would be reflected in the $\delta^{15}N$ signature. We hypothesized that if there were higher mineralization rates (faster cycling N) in mineral soil of the removed plot, it would have an enriched stable isotopic N signature relative to the reference. Lower mineralization rates on the added plot would lead to a relatively depleted signature relative to the reference.

We used alkaline CuO oxidation to determine concentrations of organic compounds derived from different biochemical precursors. In this paper we only report lignin-derived vanillyl phenols and parahydroxybenzenes, but many other biomarkers were measured. Vanillyl phenols are derived only from vascular plants while parahydroxybenzoic acids can be derived from both vascular plants and microorganisms. For this analysis, we used the ratio of parahydroxybenzoic acids to vanillyl phenols to examine the contribution of microorganisms to the organic C pool of the soils. Higher activities of microorganisms relative to the total organic C pool should result in a higher ratio.

Statistical Analysis

Analysis of variance (ANOVA) using a general linear models approach on a completely randomized block design was used to assess treatment effects of all parameters collected once during the duration of the study. A critical value of $\alpha = 0.05$ was used to test for significant differences. All ANOVA and Pearson correlation tests were run using SPSS (IBM SPSS Statistics, Version 21).

RESULTS AND DISCUSSION

Stand Volume and Periodic Annual Increment

Standing tree volume at age 15 in the added treatment was 31 and 22 percent higher than the removed and reference, respectively. No significant difference was found between the removed and reference treatments (fig. 1a). There was no significant difference between the treatments in the net volume increment from the last time these stands were measured at age 12. However, volume increment was significantly different between the ages of 5 and 8.

Whole Soils and Stand Productivity

Before describing the trends between treatments with respect to N, we needed to demonstrate that N is a limiting nutrient on these sites. Total N in the O (fig. 2a) and A (fig. 2b) horizons correlates significantly with stand volume at age 15. This suggests that N on these sites has some control over forest productivity. However,

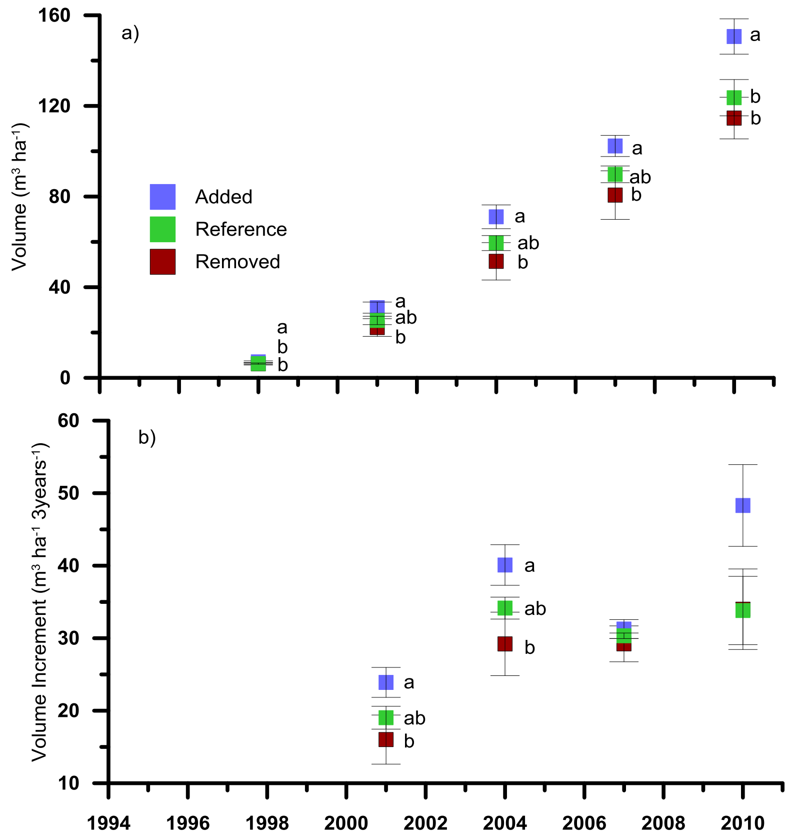

Figure 1--(a) Total bole volume, and (b) periodic annual increment from tree measurements made every year since stand initiation in 1994.

whole soil N content did not differ between treatments (total soil p = 0.395; fig. 3), although the added plots showed a trend of higher N content and the removed plots consistently showed a trend of lower N relative to the reference plots to a depths of 40 cm in the soil profile. It is possible that these small changes in N content result in trends in productivity.

By examining differences in the added and removed plots relative to the reference plots, we found that there was a significant relationship between the change in O-horizon N content and the change in productivity, suggesting that there was an increase in growth on the added treatments and a possible decrease in growth on some of the removed treatments (fig. 4). However, neither the change in volume nor the change in O-horizon N content was significantly different from zero, while only the change in volume was significantly different from zero for the added treatment.

We did not find any significant differences among the mass of LF N as we had hypothesized (data not shown). There were significant differences in HF N between added and removed treatments within the 0- to 20-cm horizon (fig. 5). However, neither treatment was different from the reference. This is possibly a result of recovery that has occurred since treatment application. The addition/removal of N in the slash and O horizon of the added/removed treatment was approximately ±300 kg/ha. In just the HF of 0 to 20 cm, the difference between the removed and reference treatments was about 151 kg/ha or 51 percent of the treatment effect. The HF of the 0- to 20-cm horizon of the added treatment contained 222 kg/ha more than the reference or 74 percent of the added N.

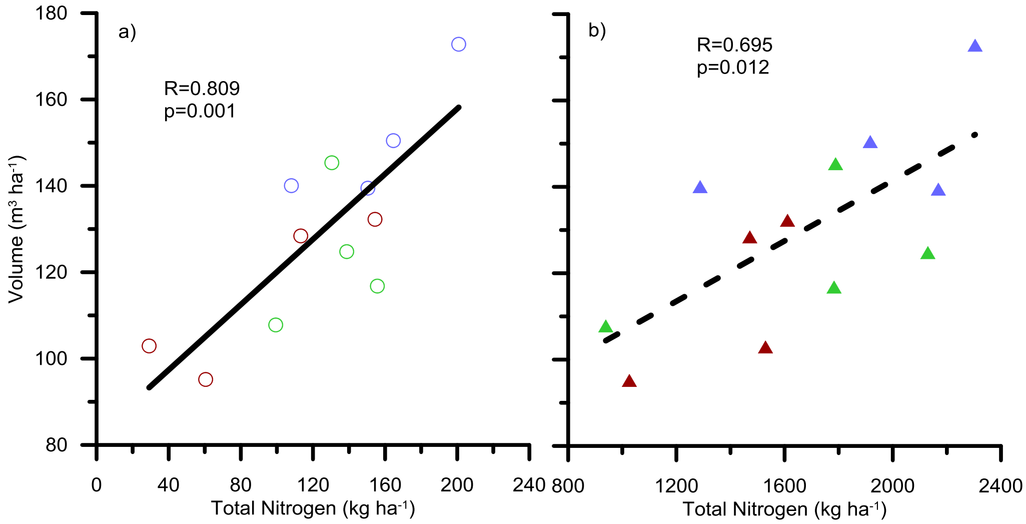

Figure 2--Relationship of total N in: (a) O horizon, and (b) 0- to 20-cm horizon. Colors of symbols represent the same treatments as in figure 1.

This suggests that the HF was at least partly responsible for resiliency of the stand, and this fraction should contain the signal of what has occurred in the soil since treatment.

In soils, depleted forms of N are mineralized and taken up by organisms or leached. The heavier isotope (^{15}N) is left behind leading to enrichment with depth. We expected a similar process to lead to enriched stable isotopic signatures in the removed treatment if N was mineralized faster. This faster rate of N mineralization may have partially relieved a N limitation caused by removing organic matter. Table 1 shows that there was not a significant difference in the HF stable isotopic N signature at $\alpha = 0.05$; however it was significant at $\alpha = 0.10$. In concordance with our hypothesis, HF from the removed treatments was enriched in ^{15}N, suggesting N was cycled faster on the removed plots. Interestingly, the added plots displayed a depleted stable isotopic signature relative to the reference, suggesting that N was cycled at a relatively slower rate. It should be noted that the O horizon has a depleted ^{15}N signature, so removal or addition of this material (i.e. the treatments) could have resulted in these trends. Additional measurements would be needed to determine if the signal in ^{15}N was a direct effect of the treatments or a result of a change in N mineralization rates and a mechanism for soil resiliency.

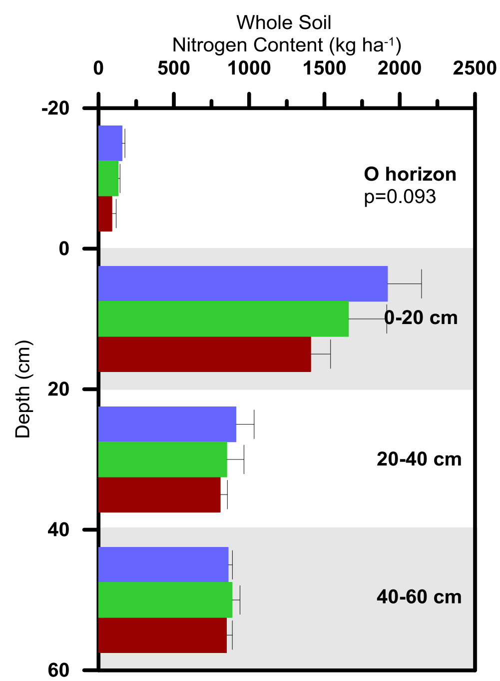

Figure 3--Total N in each soil horizon. Colors of bars represent the same treatments as in figure 1.

The ratio of parahydroxybenzoic acid to vannillyl is an indicator of microbial contributions to SOM. Elevated ratios suggest that there are higher rates of microbial activity relative to the total SOM pool. We found that the stable isotopic signature of the 0- to 20-cm whole soil and the parahydroxybenzoic acid to vannilyl ratio were

Table 1--Stable isotopic N signature of the HF from the 0 to 20 cm horizon. Enriched values of $\delta^{15}N$ suggest faster rates of N cycling

Treatment	$\delta^{15}N$
	ppm
Added	-0.67±0.38
Reference	0.17±0.17
Removed	1.07±0.85
p	0.07

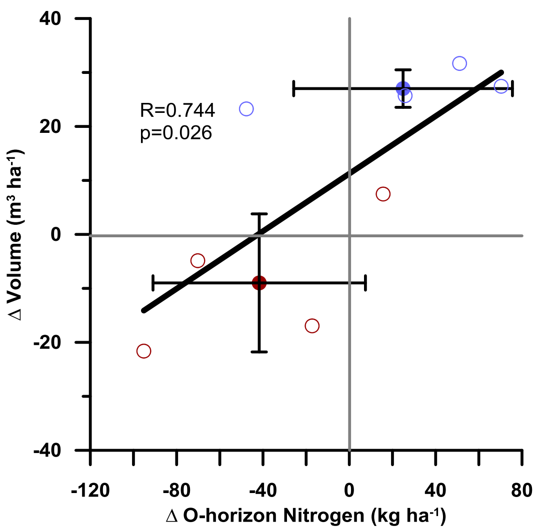

Figure 4--Relationship of the change in O-horizon N content and change in volume relative to the reference plot within each block. Solid circles represent averages with 95 percent confidence interval. Colors of circles represent the same treatments as in figure 1.

significantly correlated, suggesting that a portion of the stable isotopic signature was a result of higher rates of microbial activity (fig. 6a). We also found that there were significant differences in this indicator ratio among the treatments at α = 0.10 (fig. 6b). This trend is not the result of the addition or removal of organic matter since O horizons had a much higher ratio of parahydroxybenzoic acids to vannilyl ratio. This result was similar across several other biomarkers including fatty acids, diacids, and benzene dicarboxylic acids (all markers of relative contributions of microbes to SOM).

We found evidence for a slight nutrient limitation in the removed treatment early in stand history, but this nutrient limitation was alleviated by higher rates of mineralization of soil N pools. There is the possibility that the removed treatment could have been affected by higher competition due to the favorable seedbed conditions created by the removal of slash and O-horizon material. The added treatment responded to the addition of nutrients by increasing growth rates relative to the control; however this could have also been confounded by differences in competition early on in the stand.

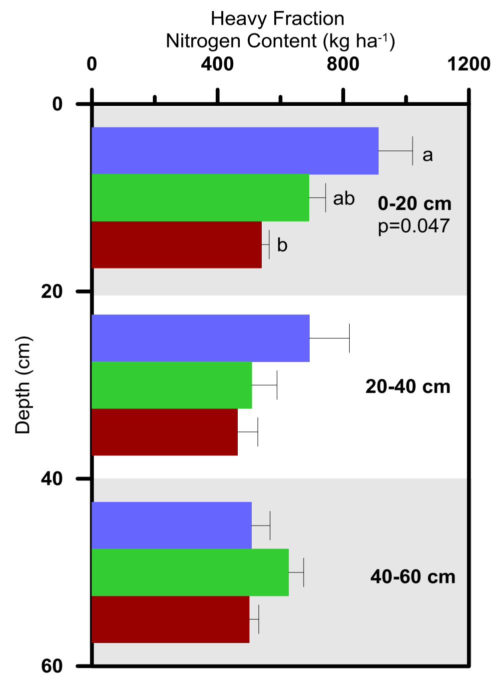

Figure 5--Total heavy fraction N in each soil horizon. The HF represents what is thought to be a slow cycling pool of N in the soil. Colors of bars represent the same treatments as in figure 1.

Despite uncertainties regarding the levels of competition experienced by the developing stands, we found compelling evidence of a mechanism for resiliency in these soils that includes the mineralization of what is thought to be a recalcitrant pool of organic nutrients. What drove the mineralization of this recalcitrant pool? Possibly changes in soil moisture and temperature as a result of exposing the mineral soil directly to precipitation and sunlight, changes to the microbial community, or possibly some other conceived mechanism. More research is necessary to more certainly determine the mechanisms of this resiliency.

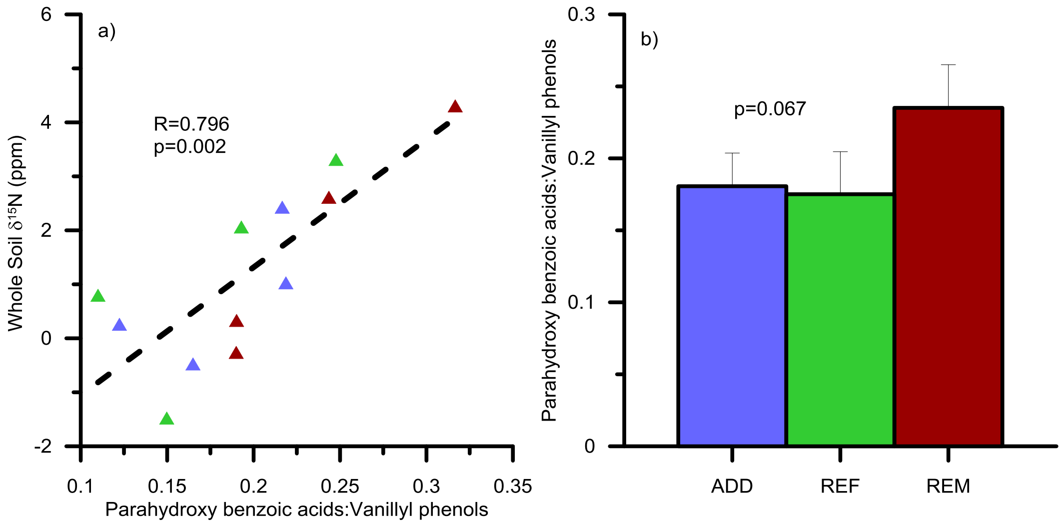

Figure 6--(a) Relationship of 0 to 20 cm $\delta^{15}N$ and parahydroxy benzoic acids in the 0- to 20-cm soil horizon; and (b) ratio of parahydroxy benzoic acids to vanilyl phenols in the 0- to 20-cm horizon. Parahydroxy benzoic acids are a biomarker of microbial contributions to SOM. Colors of symbols represent the same treatments as in figure 1.

CONCLUSIONS

We found evidence for a mechanism that alleviated a possible nutrient limitation in a loblolly pine plantation with a severe organic-matter-removal treatment. This mechanism appeared to be higher rates of mineralization of slow cycling soil N pools. These results have implications for long-term site productivity since the recovery of labile N after severe removal of organic matter may be slow, and productivity of subsequent rotations may be negatively impacted if not ameliorated through management.

LITERATURE CITED

Burkhart, H. 1977. Cubic-foot volume of loblolly pine to any merchantable top limit. Southern Journal of Applied Forestry. 1: 7-9.

Crow, S.E.; Lajtha, K.; Filley, T.R. [and others]. 2009. Sources of plant-derived carbon and stability of organic matter in soil: implications for global change. Global Change Biology. 15: 2003-2019.

EISA [Energy Independence and Security Act]. 2007. House Resolution 6. 110th Congress, 1st Session.

Fisher, R.F.; Binkley, D. 2000. Ecology and management of forest soils. 3rd ed. New York: John Wiley & Sons. 512 p.

Hatten, J.A.; Goni, M.A.; Wheatcroft, R.A. 2012. Chemical characteristics of particulate organic matter from a small, mountainous river system in the Oregon Coast Range, USA. Biogeochemistry. 107: 43-66.

Huang, Z.; Clinton, P.W.; Davis, M.R.; Yang, Y. 2011. Impacts of plantation forest management on soil organic matter quality. Journal of Soils and Sediments. 11: 1309-1316.

NOAA. 2010. NOWData, NOAA Online Weather Data. National Weather Service Forecast Office. Birmingham, AL. http://www.nws.noaa.gov/climate/xmacis.php?wfo=bmx. [Date accessed: August 8, 2011].

NRCS [Natural Resources Conservation Service]. 2012. Web soil survey. Washington, DC: U.S. Department of Agriculture Natural Resources Conservation Service. http://websoilsurvey.nrcs.usda.gov/. [Date accessed: August 21, 2011].

Ponder, F.J. 2008. Nine-year response of hardwood understory to organic matter removal and soil compaction. Northern Journal of Applied Forestry. 25: 25-31.

Powers, R.F.; Scott, D.A.; Sanchez, F.G. [and others]. 2005. The North American Long-Term Soil Productivity Experiment: findings from the first decade of research. Forest Ecology and Management. 220: 31-50.

Six, J.; Schultz, P.; Jastrow, J.; Merckx, R. 1999. Recycling of sodium polytungstate used in soil organic matter studies. Soil Biology and Biochemistry. 31(8): 1193-1196.

Zerpa, J.; Allen, H.; Campbell, R. [and others]. 2010. Influence of variable organic matter retention on nutrient availability in a 10-year-old loblolly pine plantation. Forest Ecology and Management. 259: 1480-1489.

SOIL ORGANIC MATTER FRACTIONS IN LOBLOLLY PINE FORESTS OF COASTAL NORTH CAROLINA MANAGED FOR BIOENERGY PRODUCTION

Kevan J. Minick, Brian D. Strahm, Thomas R. Fox, Eric B. Sucre, and Zakiya H. Leggett[1]

Dependence on foreign oil continues to increase, and concern over rising atmospheric CO_2 and other greenhouse gases has intensified research into sustainable biofuel production. Intercropping switchgrass (*Panicum virgatum* L.) between planted rows of loblolly pine (*Pinus taeda* L.) offers an opportunity to utilize inter-row space that typically contains herbaceous and weedy competition for the production of a biomass feedstock. However, introduction of a dedicated energy crop to these forests may influence soil carbon (C) and nitrogen (N) availability through various effects on soil organic matter (SOM) quality, quantity, and stabilization and release of N through microbially-mediated decomposition processes. Our overall objective of this research was to investigate the influence of forest-based bioenergy production on soil C and nutrient biogeochemistry to enhance our understanding of the sustainability of this type of forest management. In this study specifically, we investigated the effects of loblolly pine-switchgrass intercropping on SOM fractions in an intensively managed loblolly pine plantation. Investigating SOM stabilization and destabilization mechanisms will give insight into the long-term soil C storage potential in these forested ecosystems.

The Lenoir I Intercropping Sustainability Study was located in the Lower Coastal Plain physiographic province near Dover, NC. Soils were mapped as Pantego (fine-loamy, siliceous, semiactive, thermic Umbric Paleaquults) or Rains (fine-loamy, siliceous, semiactive, thermic Typic Paleaquults) soil series. Soils at this site were very poorly drained with a seasonal water table at or near the surface. Ditching was maintained to lower the water table and reduce soil water saturation. Bedding was implemented as part of site preparation to raise root systems of young loblolly pines above the water table, increase soil aeration, and reduce competition. Switchgrass planted at this research site was the Alamo lowland variety.

In summer 2008, four blocks of seven plots (0.8-ha treatment plots with 0.4-ha measurement plots with a minimum 15-m outer buffer) were established on a recently harvested 34-year-old loblolly pine plantation. Treatments included: (1) traditional pine establishment with biomass left in place, (2) traditional pine establishment with biomass removed, (3) pine intercropped with switchgrass between bed rows with biomass left in place, (4) pine intercropped with switchgrass between bed rows with biomass removed, (5) pine establishment with an “extra” row of trees flat-planted in between crop tree beds with biomass left in place, (6) pine establishment with an “extra” row of trees flat-planted in between crop tree beds with biomass removed, and (7) switchgrass only. One of four replicate plots per treatment was randomly located in each block. Treatments with biomass left in place reflect standard site preparation following harvest. Removal of biomass included all harvest residuals > 5 cm diameter, not including sheared stumps. All plots were equipped with a weather station which collected hourly data on soil temperature (°C) and soil moisture at 10 cm and 30 cm depths.

[1]Ph.D. Candidate, Assistant Professor, and Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061; and Sustainability Scientists, Weyerhaeuser Company, Southern Timberlands Technology, Vanceboro, NC 28586.

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Table 1--Total soil organic carbon (C) and nitrogen (N) measured from soil samples collected from the bed and interbed of each treatment and at four discrete depth increments. Means of replicates are shown with standard error in parenthesis (n = 4). Treatment effects at each location and depth were tested using one-way ANOVA

	--------------------Percent C--------------------			--------------------Percent N--------------------		
Location	P+	P-	PS+	P+	P-	PS+
Bed						
0- 5 cm	4.54 (0.32)	4.59 (0.53)	3.71 (0.53)	0.18 (0.02)	0.18 (0.03)	0.14 (0.02)
5-15 cm	4.57 (0.47)	5.39 (0.96)	5.00 (0.53)	0.18 (0.03)	0.21 (0.05)	0.17 (0.02)
15-30 cm	3.73 (0.52)	4.27 (1.25)	2.94 (0.24)	0.14 (0.02)	0.18 (0.06)	0.11 (0.01)
30-45 cm	1.24 (0.36)	1.31 (0.19)	1.08 (0.10)	0.05 (0.01)	0.06 (0.01)	0.05 (0.01)
Interbed						
0- 5 cm	7.43 (0.81)	7.11 (0.81)	6.56 (1.08)	0.31 (0.02)	0.31 (0.05)	0.26 (0.05)
5-15 cm	4.46 (0.37)	4.98 (0.69)	4.18 (0.52)	0.18 (0.02)	0.20 (0.04)	0.16 (0.02)
15-30 cm	1.94 (0.57)	1.98 (0.26)	1.76 (0.25)	0.08 (0.02)	0.08 (0.01)	0.07 (0.01)
30-45 cm	0.69 (0.10)	0.67 (0.04)	0.78 (0.13)	0.04 (0.01)	0.04 (0.01)	0.04 (0.01)

Of the seven treatments, three were included in this study: (1) traditional pine establishment with biomass left in place (P+); (2) traditional pine establishment with biomass removed (P-); and (3) pine intercropped with switchgrass between bed rows with biomass left in place (PS+). In May 2011, soil samples were collected from the beds and interbeds in each treatment plot using a 7-cm-diameter PVC core and divided into 0-5, 5-15, 15-30, and 30-45 cm depth increments. Chemical, biochemical, and physical protection of SOM were tested using acid hydrolysis (Leavitt and others 1996, Paul and others 2006), density fractionation (Cambardella and Elliot 1993, Golchin and others 1994, Sollins and others 2006, von Lützow and others 2007), and aggregate fractionation methods (Cambardella and Elliot 1993, Jastrow and others 1996, Six and others 1998), respectively. All whole soil and fractionated samples were dried at 50 ºC for 12 hours and subsequently ground and analyzed for C and N concentrations and $\delta^{13}C$ and $\delta^{15}N$ isotopic signature by an IsoPrime 100 stable isotope ratio mass spectrometer (IRMS) interfaced with an elementar vario MICRO cube dry-combustion elemental analyzer (Elementar, Hanau, Germany).

As expected, total C and N decreased with sampling depth and ranged from >7 percent C and 0.3 percent N at the surface to <1 percent C and 0.05 percent N at the 30 to 45 cm depth (table 1). No significant treatment differences were found for total C or N at any depth or for the bed or interbed (table 1). We found no evidence that intercropping of switchgrass or residual removal altered hydrolyzable or non-hydrolyzable C fractions in the interbed (fig. 1). Intercropping treatments did not influence C or N content associated with aggregates in each of three size classes measured.

Our results suggest that at this early stage of forest development there has been no impact of imposed bioenergy management regimes on total C and N pools or the various SOM fractions examined. Presence of grass in forested ecosystems may lead to increased soil C stocks in the case of switchgrass (Liebig and others 2008, Ma and others 2000a) or decrease soil C stocks in the case of an invasive grass (Strickland and others 2011). Soil C pools at this site are hypothesized to increase due to switchgrass production of extensive and deep fine-root networks, therefore providing a potentially substantial input of root-derived C into the soil (Frank and others 2004, Garten and Wullschleger 1999, Liebig and others 2005, Ma and others 2000b). It is typical that detectable changes in SOM pools will not manifest themselves until at least 15 years since afforestation (Nave and others 2013), and therefore it may be too early in this study to detect changes in SOM. Although we did not find any differences in C fractions 3 years after the site was established, this research provides a detailed baseline of SOM dynamics in young loblolly pine stands upon which future long-term changes in SOM pools can be assessed. Changes in the chemical nature of SOM and physical protection within aggregates can have important effects on decomposition dynamics and long-term soil C storage (von Lützow and others 2007), as well as nutrient cycling and availability. Therefore, understanding how forest-based bioenergy production will influence soil C dynamics is essential in our ability to sustainably manage forested ecosystems.

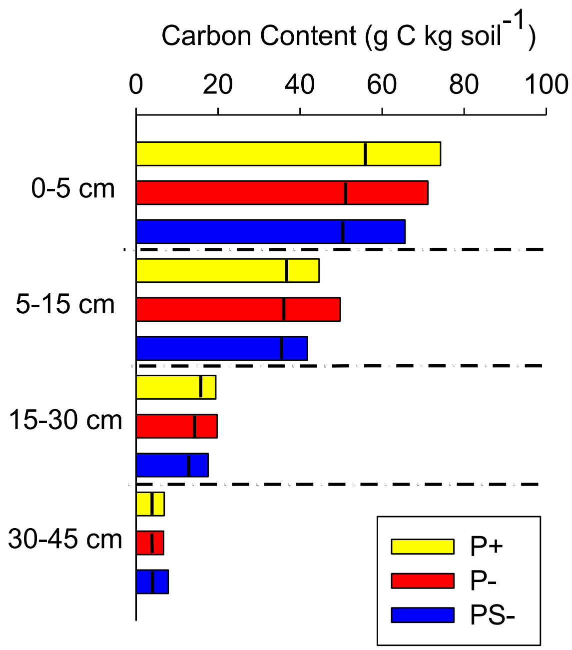

Figure 1--Carbon (C) content in soils collected from the interbed of each treatment plot at four depth increments. Stacked bars represent the non-hydrolyzable C content as the left bar and hydrolyzable C content as the right bar.

LITERATURE CITED

Cambardella, C.A. Elliott, E.T. 1993. Methods for physical separation and characterization of soil organic matter. Geoderma. 56: 449-457.

Frank, A.B.; Berdahl, J.D.; Hanson, J.D. [and others]. 2004. Biomass and carbon partitioning in switchgrass. Crop Science. 44: 1391-1396.

Garten, C.T., Jr.; Wullschleger, S.D. 1999. Soil carbon inventories under a bioenergy crop (switchgrass): measurement limitations. Journal of Environmental Quality. 28: 1359-1365.

Golchin, A.; Oades, J.M.; Skjemstad, J.O.; Clarke, P. 1994. Soil structure and carbon cycling. Australian Journal of Soil Research. 32: 1043-1068.

Jastrow, J.D.; Boutton, T.W.; Miller, R.M. 1996. Carbon dynamics of aggregate-associated organic matter estimated by carbon-13 natural abundance. Soil Science Society of America Journal. 60: 801-807.

Leavitt, S.W.; Follett, R.F.; Paul, E.A. 1996. Estimation of slow- and fast-cycling soil organic matter pools from 6N HCl hydrolysis. Radiocarbon. 38: 231-239.

Liebig, M.A.; Johnson, H.A.; Hanson, J.D.; Frank, A.B. 2005. Soil carbon under switchgrass stands and cultivated cropland. Biomass and Bioenergy. 28: 347-354.

Liebig, M.A.; Schmer, M.R.; Vogel, K.P.; Mitchell, R.B. 2008. Soil carbon storage by switchgrass grown for bioenergy. Bioenergy Research. 1: 215-222.

Ma, Z.; Wood, C.W.; Bransby, D.I. 2000a. Soil management impacts on soil carbon sequestration by switchgrass. Biomass and Bioenergy. 18: 469-477.

Ma, Z.; Wood, C.W.; Bransby, D.I. 2000b. Impact of soil management on root characteristics of switchgrass. Biomass and Bioenergy. 18: 105-112.

Nave, L.E.; Swanston, C.W.; Mishra, U.; Nadelhoffer, K.J. 2013. Afforestation effects on soil carbon storage in the United States: a synthesis. Soil Science Society of America Journal. 77: 1035-1047.

Paul, E.A.; Morris, S.J.; Conant, R.T.; Plante, A.F. 2006. Does the acid hydrolysis-incubation method measure meaningful soil organic carbon pools? Soil Science Society of America Journal. 70: 1023–1035.

Six, J.; Elliot, E.T.; Paustian, K.; Doran, J.W. 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. Soil Science Society of America Journal. 62: 1367-1377.

Sollins, P.; Swanston, C.; Kleber, M. [and others]. 2006. Organic C and N stabilization in a forest soil: evidence from sequential density fractionation. Soil Biology and Biogeochemistry. 38: 3133-3324.

Strickland, M.S.; DeVore, J.L.; Maerz, J.C.; Bradford, M.A. 2011. Loss of faster-cycling soil carbon pools following grass invasion across multiple forest sites. Soil Biology and Biochemistry. 43: 452-454.

von Lützow, M.; Kögel-Knabner, I.; Ekschmitt, K. [and others]. 2007. SOM fractionation methods: relevance to functional pools and to stabilization mechanisms. Soil Biology and Biochemistry. 39: 2183-2207.

UPTAKE OF ^{15}N LABELED FERTILIZER IN LOBLOLLY PINE PLANTATIONS OF THE SOUTHERN UNITED STATES

Jay E. Raymond, Thomas Fox, and Brian Strahm[1]

Forests in the southeastern United States managed extensively (minimal silvicultural treatments) generally have low productivity rates (Fox and others 2007). Conversely, intensively managed stands have significantly increased productivity in this region, from 2 m^3 ha^{-1} $year^{-1}$ to a range of 6 to 10 m^3 ha^{-1} $year^{-1}$ (Fox and others 2007). Yet theoretical models and empirical field trials indicate that production in excess of 20 m^3 ha^{-1} $year^{-1}$, with stand rotations of < 15 years, are biologically possible (Allen and others 2005, Borders and Bailey 2001, Fox and others 2007). These productivity increases can occur if forest plantations are intensively managed as agroecosystems through the development of site-specific silvicultural prescriptions to address factors limiting growth. The growth rates of extensively managed forests are not adequate to sustainably produce the raw materials required to support the current and future forest products and bioenergy industries, and more intensively managed forest plantations will be required to adequately and sustainably supply the raw materials required (Sedjo and Botkin 1997).

One factor contributing to low forest productivity is low levels of available soil nutrients, principally nitrogen (N) (Albaugh and others 1998, 2004; Jokela and Martin 2000; Sampson and Allen 1999). Low nutrient availability restricts leaf area production resulting in reduced photosynthetic capacity and growth. Soil N availability is normally greatest following harvesting and site preparation early in the rotation but gradually decreases as canopy closure occurs and N becomes immobilized in the forest floor and plant biomass (Fox and others 1986, Miller 1981, Vitousek and Matson 1985). Because of this pattern, N fertilization is needed in many plantations to maintain rapid growth. Fertilization can increase soil nutrient availability, gradually increase the leaf area of the stand, and improve the overall growth of the forest. Empirical results from fertilization trials in loblolly pine indicate that most nutrient limitations can be easily and cost-effectively ameliorated with fertilization (Fox and others 2007).

The precise fate of fertilizer-applied N in loblolly pine plantations is not well understood. It is estimated that 10 to 25 percent fertilizer-applied N is incorporated by crop trees (Blazier and others 2006, Mead and Pritchett 1975). Yet the remainder of the fertilizer N is assimilated in various ecosystem components (forest floor, mineral soil, understory competition, etc.) or lost (N volatilization, leaching, etc.) from the system. The variability in fertilizer-N uptake in the crop trees may contribute to the variability in growth response observed in fertilizer studies. A better understanding of the fate of applied N fertilizer in forests is needed to improve economic and environmental decisions.

Fertilizer enriched with the stable isotope ^{15}N is a technique that can accurately and precisely measure N uptake and the fate of applied N fertilizer in forest ecosystems. We used fertilizers enriched with ^{15}N (0.5 atom percent; approximately 370 per mil, 224 kg ha^{-1} N) to compare N uptake following fertilization with four conventional and enhanced-efficiency N fertilizers [urea, polymer-coated urea (PCU), urea+NBPT (NBPT), and controlled release N (CUF+NBPT)] in loblolly pine (*Pinus taeda* L.) plantations. The primary research objective of this study is to improve the fundamental understanding of N dynamics in forested systems by: (1) determining the fate of N in mid-rotation loblolly pine stands across the southern United States by using ^{15}N enriched fertilizers; and (2) comparing N dynamics following fertilization with conventional (urea) and enhanced-efficiency fertilizers enriched with ^{15}N.

Five 100-m^2 circular plots were installed at sites during 2011 and 2012 in a wide range of

[1]Graduate Research Assistant, Professor, and Assistant Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA, 24061.

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representative soil and stand conditions in the primary physiographic regions of the South. The rate of N uptake from different fertilizers was evaluated by periodic foliage sampling in 2011. Total N uptake in the growing season was determined by biomass sampling of individual tree components (foliage, fine branches, coarse branches, and stem) to determine the N mass and concentration in each component. Additional ecosystem components analyzed to determine the fate of applied N included understory vegetation, forest floor, mineral soil, litter, and roots. N volatilized following fertilization was measured using a microplot experiment.

Preliminary results at a site in the Virginia Piedmont indicate potential differences exist between urea and enhanced-efficiency fertilizers after winter fertilizer application. For example, when comparing the ^{15}N isotopic signature (per mil) in the forest floor in the N-volatilization microplots, urea had a lower value (105 per mil) compared to all enhanced-efficiency fertilizers (CUF=210 per mil, NBPT=204 per mil, PCU=186 per mil) in the forest floor 1 day after the winter fertilizer application. A larger ^{15}N value may indicate less volatilization from enhanced-efficiency fertilizers when compared to urea. Ongoing analysis is investigating if this finding is consistent across the major physiographic regions of the southern United States that this study encompasses.

LITERATURE CITED

Albaugh, T.J.; Allen, H.L.; Dougherty, P.M. [and others]. 1998. Leaf-area and above- and belowground growth responses of loblolly pine to nutrient and water additions. Forest Science. 44: 317-328.

Albaugh, T.J.; Allen, H.L.; Dougherty, P.M.; Johnsen, K.H. 2004. Long term growth responses of loblolly pine to optimal nutrient and water resource availability. Forest Ecology and Management. 192: 3-19.

Allen, H.L.; Fox, T.R.; Campbell, R.G. 2005. What's ahead for intensive pine plantation silviculture in the South? Southern Journal of Applied Forestry. 29(2): 62-69.

Blazier, M.A.; Hennessey, T.C.; Dougherty, P.; Campbell, R. 2006. Nitrogen accumulation and use by a young loblolly pine plantation in southeast Oklahoma: effects of fertilizer formulation and date of application. Southern Journal of Applied Forestry. 30: 66-78.

Borders, B.; Bailey, R.L. 2001. Loblolly pine - pushing the limits of growth. Southern Journal of Applied Forestry. 25(2): 69–74.

Fox, T.R.; Burger, J.A.; Kreh, R.E. 1986. Effects of site preparation on nitrogen dynamics in the southern Piedmont. Forest Ecology and Management. 15: 241-256.

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(5): 337-347.

Jokela, E.J.; Martin, T.A. 2000. Effects of ontogeny and soil nutrient supply on production, allocation, and leaf area efficiency in loblolly and slash pine stands. Canadian Journal of Forest Research. 30: 1511-1524.

Mead, D.J.; Pritchett, W.L. 1975. Fertilizer movement in a slash pine ecosystem. 1. Uptake of N and P and N movement in the soil. Plant and Soil. 43: 451-465.

Miller, H.G. 1981. Forest fertilization: some guiding concepts. Forestry. 54(2): 157-157.

Sampson, D.A.; Allen, H.L. 1999. Regional influences of soil available water and climate, and leaf area index on simulated loblolly pine productivity. Forest Ecology and Management. 124: 1-12.

Sedjo, R.A.; Botkin, D. 1997. Using forest plantations to spare natural forests. Environment. 30: 15-20.

Vitousek, P.M.; Matson, P.A. 1985. Intensive harvesting and site preparation decrease soil nitrogen availability in young plantations. Southern Journal of Applied Forestry. 9: 120-125.

EFFECTS OF FERTILIZATION AND WEED CONTROL ON SECOND ROTATION GROWTH AND SOIL NUTRIENT AVAILABILITY IN JUVENILE LOBLOLLY PINE PLANTATIONS IN NORTH FLORIDA

Praveen Subedi, Eric J. Jokela, Timothy A. Martin, and Jason G. Vogel[1]

Evolution in silvicultural practices during the past few decades has resulted in increased productivity over a wide range of southern pine sites (Fox and others 2007). Improved site preparation, competing vegetation control, fertilization, and deployment of genetically superior planting stock have all enhanced the productive potential and yield of these forests (Colbert and others 1990, Jokela and others 2004). Although the costs for adopting intensive management systems for southern pine plantations are high, financial returns from these short-rotation and high-yielding systems are promising (Allen and others 2005). As a result, southern pine plantations in the southern United States are now among the most intensively managed forests in the world.

At the same time, concerns over the sustained productivity of intensively managed forests is increasing as a result of possible site nutrient depletion from frequent harvests, alteration of soil properties (Powers 1999), and depletion of soil carbon from sustained elimination of competing vegetation (Vogel and others 2011). Fertilization and weed control treatments are commonly adopted in southern pine plantations to enhance overall productivity (Fox and others 2007). The areal extent of annual fertilization in southern pine plantations has, thus, increased by almost 5-fold when compared to the early 1990s (Albaugh and others 2007). In the last decade, fertilizer prices have increased by almost 3-fold due to changing global supply and demand (USDA 2012). In that context, understanding the role that historic silvicultural treatments like nutrient additions and competing vegetation control have on growth dynamics and soil nutrient availability in successive rotations is critical to improve our understanding and development of intensive forest management systems.

On a north Florida Spodosol, we investigated the inter-rotational effects of fertilization and weed-control treatments on the growth and soil nutrient availability of juvenile loblolly pine (*Pinus taeda* L.) stands using two randomized complete block design experiments, each consisting of three replicates. These experiments were established on the same site (Ultic alaquods) and treatment plots as the first rotation. The first rotation's treatments were: control (C); fertilizer only (F); weed control only (W); and fertilizer + weed control (FW). Total nutrient additions over the first rotation for the F and FW treatments were (kg ha^{-1}): 1,088 N; 230 P; 430 K; 108 Ca; 72 Mg; 72 S; 4.1 Mn; 5.4 Fe; 0.9 Cu; 4 Zn; and 0.9 B (Jokela and others 2010). Competing understory vegetation in the first rotation was controlled mechanically and chemically in the W and FW treatments for the first 10 years until canopy closure suppressed further establishment (Vogel and others 2011). Prior to the establishment of the second rotation, the understory vegetation was mulched in the C and F treatments to retain this nutrient pool within the plot boundaries. Mulching was not done in the W and FW plots because of the history of sustained understory competition control from the first rotation. The original experiment was whole-tree harvested in May 2009, with harvested trees processed off the treatment plots. Following harvest, a single full-sib loblolly pine family was planted at a 1.8- by 3.0-m spacing in both experiments using containerized seedlings. One experiment was actively retreated (C, F, W, and FW – Actively Managed, Retreated) as in the previous rotation,

[1]Graduate Student, Professor, and Professor, respectively, University of Florida, School of Forest Resources and Conservation, Gainesville, FL 32611; and Assistant Professor, Texas A&M University, Department of Ecosystem Science and Management, College Station, TX 77843.

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Table 1--Analysis of variance of total aboveground biomass accumulation by stand age[a] for juvenile loblolly pine stands growing on Spodosols in north Florida

		Stand age (*years*)		
Experiments	Treatments	1	2	3
Actively managed retreated	Control (C)	A	A	A
	Fertilizer only (F)	AB	AB	B
	Fertilizer+ weed control (FW)	B	B	C
	Weed control only (W)	AB	A	AB
	p-Value	0.048	0.004	<0.001
Untreated carryover	Control (C_C)	A	A	A
	Fertilizer only (C_F)	B	B	B
	Fertilizer+ weed control (C_{FW})	A	A	A
	Weed control only (C_W)	A	A	A
	p-value	<0.001	0.001	<0.001

[a]Within a given stand age, treatments followed by same letter are not significantly different (Tukey's HSD at α = 0.05).

while the second experiment was left untreated (C_C, C_F, C_W, and C_{FW} – Untreated Carryover).

We estimated total aboveground biomass and nutrient accumulation in loblolly pine by using existing allometric equations developed for the same family growing on similar soil types, along with treatment-specific nutrient concentrations for the various biomass components (Adegbidi and others 2002). In addition, destructive sampling of understory vegetation was conducted to estimate aboveground biomass and nutrient pools among all treatments. Soil nutrient supply rates were estimated for 8 weeks during the growing season using PRS™-probes (Western Ag Innovations, Inc., Saskatoon, SK, Canada) that were buried in the upper 15 cm of the soil. We also estimated the Mehlich III extractable soil nutrient concentrations in the upper 100 cm of soil in the untreated carryover experiment.

Early results, through age 3 years, showed that loblolly pine growth in the second rotation consistently out-performed the first rotation. While the actively retreated FW treatment had significantly higher aboveground biomass accumulation (27.9 Mg ha^{-1}) than the F (17.7 Mg ha^{-1}), W (14.5 Mg ha^{-1}), and C (7.7 Mg ha^{-1}) treatments, the untreated C_F treatment (17.9 Mg ha^{-1}) had higher aboveground pine biomass than the C_{FW} (12.3 Mg ha^{-1}), C_C (9.8 Mg ha^{-1}), and C_W (9.6 Mg ha^{-1}) treatments (fig. 1, table 1). Nutrient accumulations in the pine mostly followed the biomass accumulation trends [e.g. N (kg ha^{-1}): 124 in the FW versus 37 in the C treatment].

We also observed a shrub-dominated community [e.g. *Ilex glabra* (L.) A. Gray and *Serenoa repens* (Bartr.) Small] in the F and C treatments and a grass-dominated community (e.g. *Andropogan* spp. and *Dicanthelium* spp.) in the FW and W treatments. Similar community composition differences were observed in the untreated carryover experiment. *Ilex glabra* was a major accumulator of nutrients (e.g. N, 45 percent; B, 62 percent; Mn, 82 percent of total understory pool) in the F treatment, and it affected loblolly pine growth. For instance, control of competing vegetation in the FW treatment resulted in an almost 1.5-fold gain in aboveground pine biomass compared to the F treatment.

In the untreated carryover experiment, early differences in pine biomass accumulation between the C_F and C_{FW} treatments was unexpected given the history of comparable nutrient additions during the first rotation. However, higher soil P availability (20.8 μg/10cm^2/8 weeks in C_F versus 8.2 μg/10cm^2/8 weeks in C_{FW}) in the surface soil horizons and its strong correlation with pine growth ($r = 0.8$; $p < 0.01$) suggested that the nutrient pools present in the forest floor and understory vegetation from the first rotation (Vogel and others 2011) served as an important nutrient source, especially for P, upon their mineralization in the second rotation.

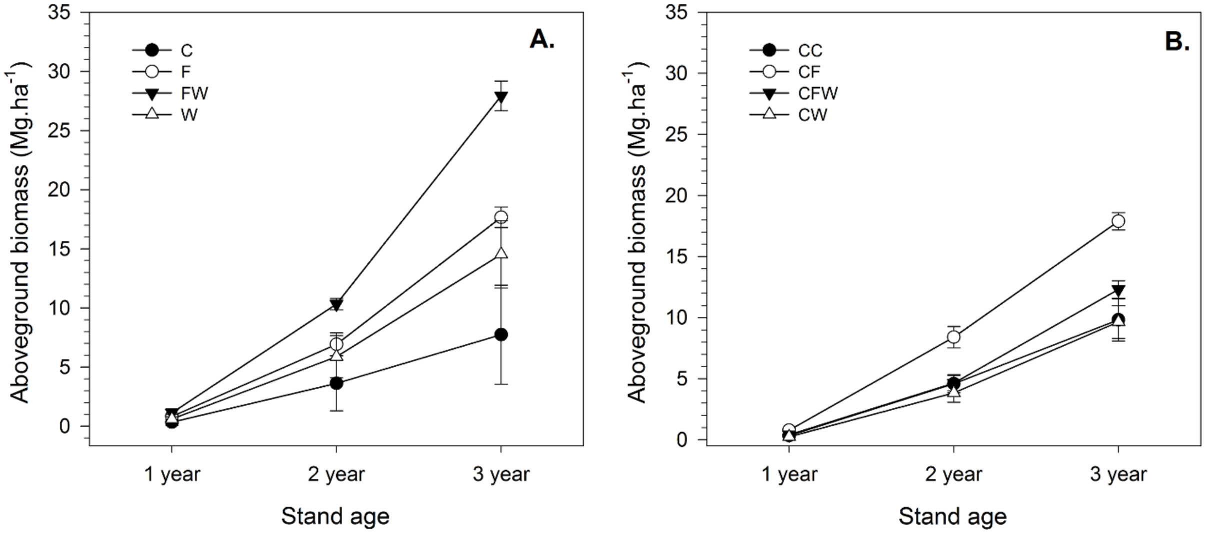

Figure 1--Total aboveground biomass accumulation for second rotation loblolly pine stands growing in (A) the actively managed retreated, and (B) untreated carryover experiments on Spodosols in north Florida. The notations C, F, FW, and W represent, respectively, the plots that received control, fertilizer only, fertilizer + weed control, and weed control only treatments in both rotations. The notations C_C, C_F, C_{FW}, and C_W, respectively, represent the untreated carryover plots that only received treatments in the first rotation: control, fertilization only, fertilization + weed control, and weed control only treatments. Error bars represent standard deviations.

In addition, historical P movement from the E to the Bh and Bt horizons [P (mg kg^{-1}): 11.4 in 0 to 20 cm, 27.5 in 50 to 100 cm], in the absence of understory vegetation, especially for the C_{FW} treatment, may have contributed to early P limitations and reduced growth. Our results suggest that understory mulching and forest floor incorporation may alleviate the need for P fertilization during stand establishment on sites previously fertilized with P.

LITERATURE CITED

Adegbidi, H.G.; Jokela, E.J.; Comerford, N.B.; Barros, N.F. 2002. Biomass development for intensively managed loblolly pine plantations growing on spodosols in the southeastern USA. Forest Ecology and Management. 167: 91-102.

Albaugh, T.J.; Allen, H.L.; Fox, T.R. 2007. Historical patterns of forest fertilization in the southeastern United States from 1969 to 2004. Southern Journal of Applied Forestry. 31: 129-137.

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: 62-69.

Colbert, S.R.; Jokela, E.J.; Neary, D.G. 1990. Effects of annual fertilization and sustained weed control on dry matter partitioning, leaf area, and growth efficiency of juvenile loblolly and slash pine. Forest Science. 36: 995-1014.

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.

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.

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: 338-347.

Powers, R.F. 1999. On the sustainable productivity of planted forests. New Forests. 17: 263-306.

USDA. 2012. USDA Economic Research Service - fertilizer use and price. http://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx#26727. [Date accessed: March 3, 2013].

Vogel, J.G.; Suau, L.J.; Martin, T.A.; Jokela, E.J. 2011. Long-term effects of weed control and fertilization on the carbon and nitrogen pools of a slash and loblolly pine forest in north-central Florida. Canadian Journal of Forest Research. 41: 552-567.

IMPACTS OF WATER AND NUTRIENT AVAILABILITY ON LOBLOLLY PINE FUNCTION

Maxwell Wightman, Timothy Martin, Eric Jokela, and Carlos Gonzalez-Benecke[1]

The impact of climate change on temperature and precipitation patterns in the southeastern United States are likely to have important effects on southern pine systems. A 2009 summary from the U.S. Global Change Research Program indicated that the southeastern U.S. will experience an increase in average temperature of 2.5 to 5 °C by the 2080s. Predictions for changes in precipitation for the Southeast, although less certain, generally indicate a 10- to 30-percent reduction in summertime precipitation. The objective of this research, part of a larger project on climate change and planted southern pine (Pine Integrated Network: Education, Mitigation and Adaptation Project, or PINEMAP, http://pinemap.org), is to quantify the impact of artificial drought conditions on loblolly pine (*Pinus taeda* L.) water relations and productivity in both fertilized and unfertilized plantations. The study uses a randomized complete block design containing two levels of fertilization and throughfall exclusion in a 2 by 2 factorial arrangement, replicated four times. Treatments include: fertilization with (kg/ha) 224 N, 28 P, 56 K, and a micronutrient blend to eliminate nutrient deficiencies, and exclusion of 33 percent of incoming throughfall. Measurements of productivity and tree water relations (including whole-tree sap flow and foliar water potential) will be used to characterize response to the treatments. The specific objectives are to: (1) quantify the impact of both 33 percent rainfall exclusion and fertilization treatments on loblolly pine productivity, whole-crown stomatal conductance, and whole tree hydraulic conductivity; (2) investigate the relationship between vapor pressure deficit, soil moisture, and stomatal conductance; and (3) provide water relations parameters for the Physiological Principles in Predicting Growth (3-PG) model.

Initial results after one summer of treatment applications show that the 30 percent reduction in throughfall did not significantly affect the growth or water use of loblolly pine plantations. The throughfall exclusion treatment did not significantly impact height growth ($p = 0.41$) or basal area production ($p = 0.36$). Mean daily transpiration was also not affected by throughfall exclusion ($p = 0.68$).

The application of fertilizer increased growth and water use of loblolly pine. Basal area growth was significantly higher in stands that received fertilizer ($p = 0.05$); however, there was no apparent impact of fertilization on height growth ($p = 0.84$). Plots that received fertilizer also had higher rates of mean daily transpiration although this relationship was not statistically significant.

The study will continue to be monitored to determine whether increased leaf area from fertilizer will impact the response of the stands to altered precipitation.

[1]Graduate Research Assistant, Professor, Professor, and Research Associate, University of Florida, School of Forest Resources and Conservation, Gainesville, FL 32611.

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Vegetation Management

Moderators:

Jimmie Yeiser
University of Arkansas
Provost/Vice Chancellor of Academic Affairs

and

Joshua Adams
University of Arkansas
School of Forest Resources

EFFECTS OF COMPETING VEGETATION ON GROWTH OF LOBLOLLY PINE PLANTATIONS IN THE WEST GULF COASTAL PLAIN

Dean W. Coble[1]

Competing woody vegetation negatively affects the growth of planted loblolly pine (*Pinus taeda* L.) trees by seizing site resources that otherwise would be used by the planted trees (Burkhart and Sprinz 1984). The West Gulf coastal plain represents a range of growing conditions from lower coastal plain to upland sites that host a variety of vegetation that compete with planted loblolly pine. Estimation of the productivity of these plantations requires that the effect of woody competition on planted tree growth be better understood across this range of growing conditions. In order to quantify the effects of competing woody vegetation on planted loblolly pine, competing woody vegetation has been measured on 127 permanent growth and yield plots since 2004. These study sites are part of the East Texas Pine Plantation Research Project and 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. Woody competing vegetation was tallied on four nested subplots [1/40th acre circular plots for woody competition > 1-inch d.b.h. (saplings) and 1/200th acre circular plots for all other woody competition < 1-inch d.b.h. (understory)] located within each 0.23-acre plot. Woody competition was quantified four ways to evaluate the influence on planted pine basal area per acre: (1) ratio of sapling basal area per acre to the total (planted pine plus sapling) basal area per acre (Burkhart and others 1987, Lee and Coble 2002); (2) basal area per acre of saplings; (3) linear feet of understory; and (4) total linear feet of saplings and understory. Regression analysis was used to compare these measures of woody competition to the basal area per acre of planted pine. Dominant height (feet) and plantation age (years) were included in the analysis as covariates to account for variability in site productivity and stage of development, respectively. The analysis found that total linear feet of all woody competition was the strongest measure of competition that negatively impacted planted pine basal area per acre. Total linear feet of all woody competition most likely represented woody competition better than the other measures in this study because yaupon (*Ilex vomitoria* Aiton) was the most abundant competing species in these young plantations (average age = 5.4 years). At younger ages, yaupon rootstocks produce abundant multiple stems that compete for available growing space. These stems are not large in diameter and therefore have low basal area; however, they can overtop and out-compete planted pine. Total linear feet best represented the effects of competing woody vegetation on planted pine in the early stages of development in plantations represented in this study. Future research will focus on refining methodologies for quantifying competing woody vegetation and its effects on loblolly pine growth. Total linear feet of competing woody vegetation may be used as an additional parameter in future growth models.

LITERATURE CITED

Burkhart, H.E.; Sprinz, P.T. 1984. A model for assessing hardwood competition effects on yields of loblolly pine plantations. Publ. No. FWS-3-84. Blacksburg, VA: Virginia Polytechnic Institute and State University. 55 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. Publ. No. FWS-1-82. Blacksburg, VA: Virginia Polytechnic Institute and State University. 62 p.

Lee, Y.J.; Coble, D.W. 2002. A survival model for unthinned loblolly pine plantations that incorporates non-planted tree competition, site quality, and incidence of fusiform rust. Bioresource Technology. 85(3): 301-308.

[1]Professor, Stephen F. Austin State University, Arthur Temple College of Forestry and Agriculture, Nacogdoches, TX 75962.

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HERBACEOUS WEED CONTROL IN LOBLOLLY PINE PLANTATIONS USING FLAZASULFURON

Andrew W. Ezell and Jimmie L. Yeiser[1]

Abstract--A total of 13 treatments were applied at four sites (two in Mississippi and two in Texas) to evaluate the efficacy of flazasulfuron applied alone or in mixtures for providing control of herbaceous weeds. All sites were newly established loblolly pine (*Pinus taeda* L.) plantations. Plots were evaluated monthly until 180 days after treatment. No phytotoxicity on pine seedlings was observed. Overall, flazasulfuron has potential for use in forestry herbaceous weed control applications but only in mixtures with other products.

INTRODUCTION

The benefits of herbaceous weed control (HWC) in recently planted pine plantations are well established. For more than 20 years, different herbicides have been evaluated for such applications. While there is no "silver bullet" which will provide complete control on every site, the combination of biological control and economics have narrowed the operational applications across the South to a relatively few choices of tank mixtures. These applications are all highly cost effective and therefore make it difficult for new products to enter the market. Combined with the established applications is the fact that very few new products are proposed for use in forestry applications. Thus, when a new product is proposed for use in forestry, it is worthwhile to conduct a thorough evaluation.

OBJECTIVES

The objectives of this study were as follows: (1) to evaluate the efficacy of flazasulfuron for herbaceous weed control in first-year loblolly pine (*Pinus taeda* L.) plantations; and (2) to evaluate the crop tolerance of loblolly pine seedlings to applications of Flazasulfuron.

STUDY SITES

Two study sites were utilized in Mississippi and in Texas. In each state, one site received all treatments with non-ionic surfactant added (0.25 percent v/v), and one site had treatments with no surfactant. In Mississippi, the first site was on forest industry land with Prentiss silt loam soil (pH = 5.3). The site was harvested April 2009, received a shear/combination plow treatment in August 2010, and was hand planted January 2011. The second site in Mississippi was on forest industry land with Smithdale-Ruston sandy clay loam soil (pH = 5.0). The area had been harvested June 2010, subsoiled August 2010, received chemical site preparation treatment September 2010, and was hand planted in January 2011.

In Texas, the first site was near the town of St. Augustine. It had clay soils and had been machine planted in winter 2011. The second site was near Forest, Texas, had sandy loam soils, and had been machine planted in winter 2011.

TREATMENTS

A complete list of treatments is provided in table 1. While the addition of non-ionic surfactant is presented in the table, it was included for treatments on only one site per state as noted earlier. The SL-160 listed in the table is flazasulfuron.

Treatments were applied on March 16 (without surfactant) and March 21, 2011 (with surfactant) in Mississippi. Treatments were applied on March 17 (without surfactant) and April 2, 2011 (with surfactant) in Texas. All treatments were applied as a 5-foot swath over the top of the planted seedlings using a CO_2-powered backpack sprayer and hand-held wand.

Each treatment was replicated four times at each site. A completely randomized design or randomized complete block design was utilized depending on site conditions. Each plot (replication) consisted of 100 linear feet of the planted row with the 5-foot spray swath.

[1]Professor, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; and Provost/Vice Chancellor for Academic Affairs, University Arkansas at Monticello, Monticello, AR 71656.

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Table 1--List of treatments in 2011 ISK field trials using SL-160 (flazasulfuron) and non-ionic surfactant (NIS)

Treatment number	Herbicides (rates of product/A)
1	Untreated check
2	SL-160 (3 oz) + NIS (0.25% v/v)
3	SL-160 (6 oz) + NIS (0.25% v/v)
4	SL-160 (9 oz) + NIS (0.25% v/v)
5	Oust (2 oz) + NIS (0.25% v/v)
6	SL-160 (6 oz) + Arsenal AC (4 oz) + NIS (0.25% v/v)
7	SL-160 (9 oz) + Arsenal AC (4 oz) + NIS (0.25% v/v)
8	SL-160 (6 oz) + Velpar L (32 oz) + NIS (0.25% v/v)
9	SL-160 (9 oz) + Velpar L (32 oz) + NIS (0.25% v/v)
10	Oust (2 oz) + Arsenal AC (4 oz) + NIS (0.25% v/v)
11	Oust (2 oz) + Velar L (32 oz) + NIS (0.25% v/v)
12	Oust Extra (3 oz) + NIS (0.25% v/v)
13	SL-160 (6 oz) + Escort (1.07 oz) + NIS (0.25%)

EVALUATIONS

Plots were evaluated at 30, 60, 90, 120, 150, and 180 days after treatment (DAT). For each evaluation, an ocular estimate of ground cover was recorded by vegetation type of either grasses, broadleaf forbs, or vines. Loblolly pine seedlings were also evaluated for phytotoxic symptoms at each timing.

Data were analyzed using ANOVA procedures. Means were separated using Duncan's New Multiple Range Test (DNMRT).

RESULTS

No phytotoxic symptoms were observed on any seedlings at any evaluation timing in either state. Flazasulfuron is safe to use over loblolly pines at the rates tested in this study.

Mississippi--Differences were observed between sites in Mississippi. These are attributed to variation in site preparation and the ensuing weed complex which occupied each site. As might be expected, coverage by grasses and forbs was appreciably lower on the site which had received chemical site preparation.

On the site with no surfactant, grass coverage in untreated plots decreased through the growing season as forb coverage increased (tables 2 and 3). Grass control in treatments with flazasulfuron alone (treatments 2, 3, and 4) was not as good as areas treated with mixtures of flazasulfuron plus Arsenal AC® or Velpar L® (treatments 6, 7, 8, and 9). The best control was provided by treatment 10 (Arsenal AC® + Oust XP®) and 11 (Oust XP® + Velpar L®), but these treatments were not significantly different from the flazasulfuron tank mixes.

Forb pressure on this site was strong (table 3). Again, the flazasulfuron applied alone did not provide desirable levels of control (treatments 2, 3, and 4). Oust XP® applied alone provided the best control (treatment 5). This treatment and the Arsenal AC® + Oust XP® (treatment 10) provided the best forb control.

On the Mississippi site with surfactant added, there was very little grass coverage (table 4) which is attributed to the chemical site preparation. With this lack of competition, there were no significant differences in any of the treatments. Forb coverage on this site varied through the growing season as winter annuals gave way to warm season species (table 5). Overall, the high rate of flazasulfuron applied alone and the mixes with both Arsenal AC® and Velpar L® provided comparable control to the "standards" of Arsenal AC® + Oust XP® or Oust XP® + Velpar L®.

Texas--Texas endured the worst drought in their history of recorded weather conditions during 2011. In essence, the lack of moisture compromised the response to treatments. There was an effect of treatments (tables 6, 7, 8, and 9) evidenced by a comparison of untreated

Table 2--Average percent grass coverage In Mississippi flazasulfuron field trials without surfactant. Values are averages of all replications[a]

Treatment number	Days after treatment 30	60	90	120	150	180
	percent					
1	80.0b	71.3c	72.5d	45.0c	38.8b	37.3b
2	13.0a	26.3b	55.0c	38.8b	36.3b	37.3b
3	12.5a	26.3b	51.3c	31.3b	28.8b	27.8b
4	7.3a	15.0ab	36.3bc	30.0b	31.3b	33.3b
5	6.0a	15.0ab	26.3b	37.5b	35.0b	33.3b
6	6.5a	7.0a	36.3bc	25.0ab	25.0ab	25.0ab
7	14.8a	5.0a	16.8ab	23.8ab	25.0ab	25.0ab
8	6.8a	11.3a	23.8b	26.3ab	26.3ab	25.0ab
9	6.3a	9.3a	26.8b	18.8a	21.3a	20.0a
10	10.8a	1.8a	6.8a	11.3a	12.5a	13.3a
11	7.0a	4.3a	8.0a	16.3a	16.3a	16.3a
12	4.5a	8.0a	20.0b	41.3bc	38.8b	37.3b
13	11.3a	23.8b	60.0c	51.3c	47.5c	45.0c

[a]Values in a column followed by the same letter do not differ at α = 0.05.

Table 3--Average percent broadleaf coverage in Mississippi flazasulfuron field trials without surfactant. Values are averages of all replications[a]

Treatment number	Days after treatment 30	60	90	120	150	180
	percent					
1	12.0b	38.0c	45.0b	62.5d	70.0e	75.0d
2	2.0a	20.0b	35.0b	48.8c	63.8d	65.8d
3	2.0a	12.5ab	27.5ab	30.0ab	41.3bc	45.0bc
4	1.5a	6.3a	26.3ab	35.0b	48.8c	50.8c
5	1.3a	6.3a	43.8b	35.0b	12.5a	18.8a
6	1.0a	4.5a	30.0ab	35.0b	46.3c	50.0c
7	0.8a	3.5a	22.5a	23.8a	32.5b	35.0b
8	2.5a	8.8a	22.5a	28.8a	40.0bc	41.3b
9	1.0a	4.5a	26.3ab	35.0b	42.5bc	42.5b
10	1.0a	1.5a	15.0a	21.3a	28.8b	33.3ab
11	1.3a	3.3a	40.0b	51.3c	61.3d	70.0d
12	0.5a	3.5a	30.0ab	42.5c	50.0c	56.8c
13	1.3a	3.3a	18.8a	32.5b	31.5b	37.8b

[a]Values in a column followed by the same letter do not differ at α = 0.05.

Table 4--Average percent grass coverage in Mississippi flazasulfuron field trials with surfactant. Values are averages of all replications[a]

Treatment number	Days after treatment 30	60	90	120	150	180
	percent					
1	0.0a	0.8a	3.8b	2.0ab	4.0b	7.5c
2	0.0a	0.3a	0.3a	0.5a	1.8a	3.3ab
3	0.0a	0.0a	0.5a	0.3a	1.3a	3.0ab
4	0.0a	0.3a	0.5a	0.3a	0.3a	3.0ab
5	0.0a	0.0a	0.3a	0.0a	0.5a	1.0a
6	0.0a	0.0a	0.0a	0.0a	0.5a	1.0a
7	0.0a	0.0a	0.0a	0.0a	0.8a	1.0a
8	0.0a	0.5a	0.5a	0.0a	0.3a	0.5a
9	0.0a	0.3a	0.3a	0.3a	0.5a	0.5a
10	0.0a	0.0a	0.0a	0.0a	0.5a	0.5a
11	0.0a	0.3a	0.3a	0.3a	0.5a	1.0a
12	0.0a	0.3a	0.0a	0.0a	0.0a	1.0a
13	0.0a	0.0a	0.3a	0.8a	1.8a	3.0ab

[a]Values in a column followed by the same letter do not differ at α = 0.05.

Table 5--Average percent broadleaf coverage in Mississippi flazasulfuron field trials with surfactant. Values are averages of all replications[a]

Treatment number	Days after treatment 30	60	90	120	150	180
	percent					
1	45.0c	41.3d	41.3b	57.5c	66.3c	70.0c
2	20.0b	28.8c	6.8a	27.5b	33.8b	35.3b
3	10.3b	12.8b	8.0a	25.0b	32.5b	37.3b
4	6.3a	33.8c	6.3a	16.3a	22.5ab	25.0a
5	9.3ab	19.0bc	5.0a	23.8ab	32.5b	34.3b
6	4.5a	17.0b	4.5a	20.0ab	25.0ab	30.0ab
7	9.5ab	19.0bc	4.0a	14.3a	18.8a	21.3a
8	3.0a	1.5a	3.5a	10.0a	17.5a	20.0a
9	3.0a	5.5a	3.0a	10.0a	15.5a	18.3a
10	8.8ab	2.0a	3.3a	12.5a	17.5a	20.3a
11	2.3a	3.3a	2.8a	9.5a	16.3a	18.8a
12	3.0a	12.8b	2.0a	10.0a	15.0a	18.8a
13	4.3a	12.8b	5.0a	15.0a	21.3a	25.0a

[a]Values in a column followed by the same letter do not differ at α = 0.05.

Table 6--Average percent grass coverage in Texas flazasulfuron field trials with surfactant. Values are averages of all replications[a]

Treatment number	Days after treatment 30	60	90	120	150
	percent				
1	9.0a	15.0a	32.5a	37.5a	37.5a
2	6.0ab	8.3b	25.0ab	22.5b	20.0bc
3	2.3b	4.8bc	18.3bc	20.8b	23.3b
4	1.5b	2.0c	10.3cde	13.3bcd	13.3cd
5	2.5b	7.3b	17.5bcd	20.0bc	27.5b
6	2.0b	4.3bc	10.8cde	6.5d	9.0d
7	2.0b	1.5c	6.3e	3.8d	3.8d
8	1.3b	2.0c	5.8e	5.8d	10.0cd
9	1.7b	4.3bc	8.5de	11.0cd	6.0d
10	3.3b	1.5c	2.5e	4.3d	6.0d
11	0.8b	1.8c	4.0e	4.0d	3.5d
12	1.7b	2.0c	8.5de	13.3bcd	12.5cd
13	3.3b	1.5c	6.0e	6.0d	8.5d

[a]Values in a column followed by the same letter do not differ at α = 0.05.

Table 7--Average percent broadleaf coverage in Texas flazasulfuron field trials with surfactant. Values are averages of all replications[a]

Treatment number	Days after treatment 30	60	90	120	150
	percent				
1	1.5a	9.5a	17.5a	27.5a	27.5a
2	1.0ab	2.0b	9.0b	11.5bc	15.8b
3	0.3b	1.0b	5.3bcd	7.8cd	7.8bc
4	0.5ab	1.0b	3.8bcd	4.8cd	4.8cd
5	1.0ab	1.5b	8.3bc	17.5b	15.0b
6	0.8ab	0.8b	1.3cd	1.8d	2.0c
7	0.5ab	1.0b	1.5cd	1.0d	2.0c
8	0.3b	0.5b	1.3cd	1.8d	1.8c
9	0.0b	0.3b	0.5d	1.5d	0.8c
10	0.5ab	0.8b	1.0d	2.0d	2.5c
11	0.3b	0.8b	1.3cd	1.3d	1.8c
12	0.8ab	0.8b	2.0cd	4.3cd	4.8c
13	0.5ab	1.0b	1.5cd	2.0d	2.0c

[a]Values in a column followed by the same letter do not differ at α = 0.05.

Table 8--Average percent grass coverage in Texas flazasulfuron field trials without surfactant. Values are averages of all replications[a]

Treatment number	Days after treatment 30	60	90	120	150
	percent				
1	1.0a	4.3a	8.3a	25.0abc	37.5ab
2	1.5a	4.3a	15.0a	27.5ab	37.5ab
3	1.5a	4.3a	7.3a	25.0abc	35.0ab
4	1.0a	4.3a	12.5a	22.5abc	37.5ab
5	1.5a	4.3a	7.3a	22.5abc	32.5ab
6	1.0a	2.5a	4.3a	15.0bc	27.5ab
7	1.0a	1.0a	26.0a	17.5bc	27.5ab
8	1.5a	2.0a	4.8a	12.5c	25.0b
9	1.5a	3.8a	6.8a	17.5bc	27.5ab
10	1.0a	1.5a	6.0a	15.0bc	30.0ab
11	1.5a	4.3a	10.8a	32.5a	45.0a
12	1.0a	2.0a	4.3a	17.5bc	30.0ab
13	1.3a	4.8a	6.5a	17.8bc	40.0ab

[a]Values in a column followed by the same letter do not differ at $\alpha = 0.05$.

Table 9--Average percent broadleaf coverage in Texas flazasulfuron field trials without surfactant. Values are averages of all replications[a]

Treatment number	Days after treatment 30	60	90	120	150
	percent				
1	1.0a	2.5a	4.8ab	20.0ab	30.0ab
2	1.5a	3.8a	10.8a	20.0ab	35.0ab
3	1.5a	3.8a	7.3ab	20.0ab	32.5ab
4	1.0a	2.0a	10.0ab	20.0ab	35.0ab
5	1.5a	3.8a	7.3ab	15.0ab	30.0ab
6	0.8a	1.0a	2.5b	9.0b	20.0b
7	0.8a	1.0a	2.5b	13.3ab	25.0ab
8	1.0a	1.0a	4.3ab	12.5ab	20.0b
9	1.5a	3.8a	6.8ab	15.8ab	27.5ab
10	1.0a	1.5a	4.3ab	12.5ab	27.5ab
11	1.5a	4.3a	8.5ab	25.0a	42.5a
12	1.0a	1.0a	3.8ab	15.0ab	30.0ab
13	1.3a	4.5a	6.5ab	17.8ab	42.5a

[a]Values in a column followed by the same letter do not differ at $\alpha = 0.05$.

areas (treatment 1) to treated plots (treatments 2 through13). However, the ability to separate treatments at either site was greatly impacted by the harsh weather conditions. Differences in weed pressure between the two sites are attributed to a difference in the species complex and the ability to endure the drought.

SUMMARY

The treatments in this study did not control *Andropogon* spp., johnsongrass [*Sorghum halpense* (L.) Pers.], or bermudagrass [*Cynodon dactylon* (L.) Pers.]. That is not unexpected as very few HWC treatments control these species, and the treatments commonly used in pine plantations are not expected to control these species. Flazasulfuron applied alone did not provide desirable levels of grass control on the sites used in this study. The mixtures with Arsenal AC® and Velpar L® worked well and compared favorably to the operational applications currently used most widely in the South.

In forb control, the treatments did not control wooly croton (*Croton capitatus* Michx.), Virginia buttonweed (*Diodia virginiana* L.), horse nettle (*Solanum carolinense* L.), or purple cudweed [*Gamochaeta purpurea* (L.) Cabrera] very well. Of this group, only the wooly croton poses a serious threat to loblolly pine seedlings.

Flazasulfuron applied alone did not provide overall desirable levels of forb control on these sites. Again, mixtures with Arsenal AC® or Velpar L® did provide control of the forbs at levels comparable to HWC operational applications currently used in the South.

Overall, flazasulfuron has potential for use in forestry HWC applications. Based on results of this study, this material would not be applied alone, but mixtures with Arsenal AC® or Velpar L® could be effective.

FIRST- AND FIFTH-YEAR RESULTS OF A CHOPPER GEN2 SITE PREPARATION TIMING TRIAL

Jason Grogan, Andrew Ezell, Jimmie Yeiser, and Dwight Lauer[1]

Forest site preparation using imazapyr products such as Arsenal AC® and Chopper® are common in the southern United States. In 2006, Chopper Gen2® was a new formulation of imazapyr that differed from previous Chopper EC® or Arsenal AC®. Short-term site preparation tests showed positive results. Long-term effects of site preparation, as well as effects of application timing with the new formulation, were unknown, and the information was needed for operational decisions.

The objectives of this study were to: (1) evaluate the effect of application timing on efficacy of Chopper Gen2 for competition control; (2) evaluate loblolly pine (*Pinus taeda* L.) seedling survival and first-year growth response to different application timings; and (3) evaluate long-term growth and survival response of loblolly pine to different application timings. A randomized complete block design with four blocks was implemented on three sites, one each at Allen, LA; Starkville, MS; and Appomattox, VA. Plots were each approximately 0.19 acre in size. Each site had varied amounts of tree, shrub and other vegetative competition and received various levels of pre-application mechanical site preparation. The Louisiana site was bedded prior to application and had light hardwood competition with American beautyberry (*Calicarpa americana* L.), sumac (*Rhus* spp.), and blackberry (*Rubus* spp.) as the major competitors. The Mississippi site was harvested over 1 year prior to herbicide application, received no mechanical site preparation, and had heavy hardwood competition consisting of red oaks (*Quercus* spp.), post oak (*Q. stellata* Wangenh.), sweetgum (*Liquidambar styraciflua* L.), black tupelo (*Nyssa sylvatica* Marsh.), red maple (*Acer rubrum* L.), shagbark hickory [*Carya ovata* (Mill.) K. Koch], and blackberry. The Virginia site received no mechanical site preparation, and herbicide was applied soon after harvest. Competing vegetation was moderate with dominate competitors consisting of white oak (*Q. alba* L.), scarlet oak (*Q. coccinea* Münchh.), red maple, yellow poplar (*Liriodendron tulipifera* L.), black tupelo, black cherry (*Prunus serotina* Ehrh.), hickory (*Carya* spp.), and blueberry (*Vaccinium* spp.).

The herbicide treatments were applied at three different timings; July 1, August 15, and September 30, 2006. All treatments, except for the control, received Chopper Gen2® at a rate of 32 ounces per acre plus 1.0 percent v/v methylated seed oil. Site preparation was followed by hand planting with 1-0 bareroot loblolly pine seedlings in December 2006 and an herbaceous weed control treatment of 4 ounces Arsenal AC® plus 2 ounces Oust® per acre in March 2007 in all treated plots. Control plots did not receive herbaceous weed control treatment.

Control of competing vegetation was assessed in June and August 2007. Year 1 results indicate statistically less competing vegetation in herbicide-treated plots, for all timings, than in the control. Significant reduction in competition between timings only occurred for the late timing (September 30) in Mississippi (table 1). Year 1 pine survival was not different from the control for any timing. The only differences in survival among timings occurred for the August 15 timing in Mississippi, which was significantly lower than the other timings. Height and diameter growth in Louisiana were significantly greater than the control in all treatments, with no differences among timings. In Mississippi, height growth was greater than the control only in the September timing, which also showed significantly greater diameter growth than the control and other timings. Both other timings also exhibited greater diameter growth than the control; however, they were less than the September 30 timing. In Virginia, no differences were detected in height growth, and diameters

[1]Research Specialist, Stephen F. Austin State University, Arthur Temple College of Forestry, Nacogdoches, TX 75962; Professor, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; Provost/Vice Chancellor of Academic Affairs, University of Arkansas at Monticello, Monticello, AR 71656; and Owner, Silvics Analytic, Wingate, NC 28174.

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Table 1--Year 1 and year 5 results for loblolly pine seedling survival, groundline diameter, height, and volume index for three Chopper Gen2 herbicide application timings on three sites in Louisiana, Mississippi, and Virginia

	--------------Louisiana----------------				----------------Mississippi-------------				----------------Virginia--------------------			
Year/ timing	Survival	Diam.[a]	Height	Volume index	Survival	Diam.	Height	Volume index	Survival	Diam.	Height	Volume index
	%	*inches*	*feet*	*feet*^{3}	*%*	*inches*	*feet*	*feet*^{3}	*%*	*Inches*	*feet*	*feet*^{3}
Year 1												
July 1	86a[a]	0.77a	2.8a	n/a	86a	0.28b	1.6ab	n/a	89a	0.42a	1.3a	n/a
Aug. 15	86a	0.67a	2.6a		63b	0.28b	1.5b		85a	0.37a	1.2a	
Sept. 30	90a	0.68a	2.5a		89a	0.36a	1.9a		86a	0.40a	1.3a	
Control	77a	0.29b	1.7b		76ab	0.17c	1.3b		82a	0.21b	1.1a	
Year 5												
July 1	77a	4.2a	15.6a	58.3a	83a	3.9a	17.4a	54.4a	n/a	n/a	n/a	n/a
Aug. 15	84a	4.7a	16.2a	63.3a	58b	4.0a	17.0a	34.1ab				
Sept. 30	83a	4.6a	15.3a	59.3a	84a	4.1a	17.6a	56.0a				
Control	65a	2.6b	10.3b	20.3b	69ab	1.9b	10.8b	8.0b				

[a]Values within a column and year, sharing the same letter, are not significantly different at α = 0.05, as determined by Duncan's New Multiple Range Test. Diam. is diameter.

were significantly greater than the control for all timings. Year 1 differences between timings were only evident in Mississippi which exhibited greater pine height, diameter, and volume index growth for the September 30 timing. The Mississippi site had the most advanced competing vegetation of the three sites; improved performance from the late timing is likely a result of improved competition control.

Year 5 data were only collected for the Mississippi and Louisiana sites. Year 5 results indicated significantly greater height and diameter growth for all herbicide treatments than the untreated control. However there were no statistical differences among herbicide timings (table 1). The same results were found for volume index with the exception of the August 15 timing in Mississippi, which was not statistically greater than the control nor statistically less than the other timings. Differences in volume index growth for the Mississippi-August 15 timing resulted from year-1 survival rather than differences in growth parameters.

Results indicate Chopper Gen2® herbicide site preparation application, followed by herbaceous weed control, significantly increased long-term loblolly pine height, diameter, and volume index growth over untreated plantations. Timing of herbicide application had little long-term impact on loblolly pine growth or volume index.

ASSESSING TOLERANCE OF LONGLEAF PINE UNDERSTORY HERBACEOUS PLANTS TO HERBICIDE APPLICATIONS IN A CONTAINER NURSERY

D. Paul Jackson, Scott A. Enebak, James West, and Drew Hinnant[1]

Abstract--Renewed efforts in longleaf pine (*Pinus palustris* Mill.) ecosystem restoration has increased interest in the commercial production of understory herbaceous species. Successful establishment of understory herbaceous species is enhanced when using quality nursery-grown plants that have a better chance of survival after outplanting. Nursery growing practices have not been clearly identified for propagating longleaf pine understory herbaceous species, especially for controlling weeds that can reduce plant growth in containers. To identify potential herbicides that could safely be applied postemergence to herbaceous plants in containers, trials were installed at the North Carolina Forest Service Nursery in Goldsboro, NC. Different container sets of wiregrass (*Aristida stricta* Michx.) and muhly grass [*Muhlenbergia expansa* (Poir.) Trin.] seedlings were treated at two growth stages with pendimethalin, oxyfluorfen, lactofen, or oxadiazon. One-year-old wiregrass was also treated 6 months after being outplanted at the nursery with the same treatments except that clopyralid was tested instead of oxadiazon. Oxyfluorfen killed all seedlings tested within 2 weeks of treatment with the exception of 7-week-old container-grown muhly grass. Lactofen and oxadiazon applications at 2 and 7 weeks post-sowing killed both grass species and caused moderate injury, respectively. Lactofen caused foliar damage, while pendimethalin and clopyralid did not affect outplanted wiregrass. Pendimethalin applied 7 weeks post-sowing did not injure muhly grass seedlings but did cause more severe injury when applied 2 weeks post-sowing. Results of the trials indicate that pendimethalin may be safe to use over-the-top of 7-week-old container-grown muhly grass and 1-year-old outplanted wiregrass as a preemergence application to targeted weeds.

INTRODUCTION

In the last several years, attention has focused on an ecosystem approach to longleaf pine (*Pinus palustris* Mill.) restoration rather than only on establishment and management of longleaf pine as a timber resource. Indicative of this interest, a Range-Wide Conservation Plan for Longleaf Pine was published in 2009 by the Regional Working Group for America's Longleaf (Lopez and others 2009). The 15-year goal of the conservation plan is to increase longleaf pine ecosystem acreage from 3.4 to 8 million acres. To achieve this goal, the plan calls for initiatives to: (1) maintain existing longleaf ecosystems in good condition, (2) improve areas classified as "longleaf forest types" that are missing key components of the understory necessary for ecosystem sustainability, and (3) restore longleaf pine ecosystems that are currently other forest types.

Of the six strategies presented in the conservation plan (Lopez and others 2009), one focused on understory regeneration and highlighted a key action that specifically states the need to

> "…develop the seed and plant production technologies, standards, and guidance needed to produce understory plant materials and identify species important in the ground-layer of the longleaf pine communities throughout the range with the goal to help development efforts for commercial production."

Developing sound and practical methods for growing understory plant species is necessary in order to make seedlings available to private landowners, industry, and government agencies for outplanting in longleaf pine ecosystem restoration projects. The longleaf pine ecosystem is second only to tropical rain forest systems in biological diversity with hundreds of endemic plant species found across its range from eastern Texas to southern Virginia (Jose and others 2006). Many of the plants serve as fuel to help carry low-intensity surface fires (Brockway and Lewis 1997) that help perpetuate pine canopy growth, maintain species richness in the ground-layer, and provide habitat for many indigenous animal species (Jose and others 2006).

Some longleaf pine understory herbaceous plant species are currently being grown commercially in southeastern nurseries. For instance, the

[1]Assistant Professor, Louisiana Tech University, Department of Agricultural Sciences, Ruston, LA 71270; Professor, Auburn University, School of Forestry and Wildlife Sciences, Auburn University, AL 36849; and Nursery/Tree Improvement Program Head and Nursery Technician, respectively, North Carolina Forest Service, Claridge Nursery, Goldsboro, NC 27530.

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North Carolina Forest Service's Claridge Nursery, Goldsboro, NC, produces species such as wiregrass (*Aristida stricta* Michx.), muhly grass [*Muhlenbergia expansa* (Poir.) Trin.], little bluestem [*Schizachyrium scoparium* (Michx.) Nash], and Indian grass [*Sorghastrum nutans* (L.) Nash]. The cultural practices required for growing these plant species have not been clearly identified. Nursery managers are currently using a "figure it out as you go" approach for applying herbicides to understory herbaceous plants at various growth stages. Controlling weeds during production is important to prevent plants from being stunted or smothered due to increased competition for nutrients and water. At the Claridge Nursery, some of the more troublesome weeds include annual sedge (*Cyperus compressus* L.), false pimpernel [*Lindernia dubia* (L.) Pennell], Florida pusley (*Richardia scabra* L.), prostrate spurge [*Chamaesyce maculata* (L.) Small] and smooth pigweed (*Amaranthus hybridus* L.). The weeds can be a problem in the container production area as well as in nearby nursery beds that contain outplanted 1-year-old wiregrass seedlings from which seeds are harvested each year. The objective of the following trials was to determine the level of tolerance for container-grown wiregrass and muhly grass seedlings at various growth stages and outplanted 1-year-old wiregrass seedlings to a range of herbicides that target troublesome weeds encountered at the Claridge Nursery.

METHODS

Container-Grown Seedlings

The trials took place at the North Carolina Forest Service's Claridge Nursery. Wiregrass and muhly grass seeds were collected at Holly Shelter Gamelands Nature Reserve in Pender County, NC and sown on April 30, 2012 into Forestry Tray (FT-135) containers placed on T-rail benches in full sun. Each container consisted of 135 cavities (581/m^2) with a volume of 113 cm^3 and depth of 13 cm. Container cavities were filled with growing media that consisted of 85 percent peat moss, 10 percent perlite and 5 percent vermiculite.

Wiregrass and muhly grass seedlings were challenged with four postemergence herbicide treatments: lactofen (Cobra®) at 1.16 L/ha [0.28 kg active ingredient (a.i.)/ha]; oxyfluorfen (Goal® 2XL) at 2.33 L/ha (0.56 kg a.i./ha); pendimethalin (Pendulum® AquaCap™) at 2.48 L/ha (1.12 kg a.i./ha); and oxadiazon (Ronstar Flo®) at 2.92 L/ha (1.12 kg a.i./ha). A fifth treatment received no herbicide (non-treated control). All herbicide treatments were applied using a CO_2 hand-sprayer calibrated at 205 L/ha. Herbicide applications occurred at two different growth stages to both plant species: at 2 weeks post-sowing when seedlings were about 1 to 2 inches in height and at 7 weeks post-sowing when seedlings were about 4 to 6 inches in height. There were five replications (reps) of each treatment for wiregrass seedlings, with a total of 25 containers and 3,375 seedlings compromised at each application period. There were three reps of each treatment for muhly grass seedlings, with a total of 15 containers and 2,025 seedlings compromised at each application period. A rep was considered one container of seedlings or one experimental unit, and containers were arranged in a randomized complete block design.

Wiregrass and muhly grass tolerance to the herbicide treatments was evaluated by rating the amount of plant injury using a scale from 1 to 10 (1 = no injury; 10= dead plant or no green foliage). Ratings of 2 through 9 reflected a progressive increase in the amount of either chlorotic or brown foliage. Injury ratings were recorded at 8, 13, 16, and 25 days after treatment (DAT) for seedlings treated 2 weeks post-sowing and at 9, 23, 33, 44, and 51 DAT for seedlings treated 7 weeks post-sowing. Foliage fresh weights were also recorded at 51 DAT for seedlings treated 7 weeks post-sowing by clipping and weighing the foliage (dead or alive) from each container (experimental unit). Foliage fresh weights per seedling were calculated by dividing the total amount of foliage from a container of seedlings by the number of seedlings sampled from that container.

Outplanted Seedlings

Wiregrass seedlings grown during the 2011 growing season were outplanted the first week of December 2011 in nursery beds at the Claridge Nursery. The seedlings are maintained as a seed source for successive crops grown each growing season at the nursery. These 1-year-old seedlings were treated on May 16, 2012 with the same herbicide treatments as described for the container-grown wiregrass and muhly grass trials with one exception: instead of oxadiazon, the herbicide clopyralid (Stinger®) was tested at 0.29 L/ha (0.1 kg a.i./ha). There were five reps of each treatment with 10 seedlings in each rep (plot) set up in a randomized complete block design. Each plot represented an experimental unit. A total of 250 wiregrass seedlings were compromised in the trial. Herbicide tolerance was recorded using the same injury rating scale as previously described for container-grown seedlings at 8, 13, 16, 25, and 33 DAT.

Data Analysis

Data were analyzed using a generalized linear model (GLM) in SAS (9.3 ed., SAS Institute, Cary, NC). Means of each dependent variable for each experimental unit were analyzed using analysis of variance (ANOVA) and mean separation tests were performed using Duncan's Multiple Range Test. Duncan groupings and least significant differences (LSD) are only reported for wiregrass and muhly grass data collected the final evaluation day for the 7-week post-sowing trial. Injury rating data in the 2-week post-sowing trial lacked variation, and thus, those statistics are not being reported.

RESULTS AND DISCUSSION

Container-Grown Seedlings

Wiregrass--All seedlings treated with oxadiazon, lactofen, and oxyfluorfen were dead at or before13 DAT when the herbicides were applied 2 weeks post-sowing (table 1). Pendimethalin applications resulted in severe seedling damage when applied at 2 weeks post-sowing, and the damage was more gradual compared to the other herbicide treatments.

Wiregrass seedlings treated at 7 weeks post-sowing were killed by oxyfluorfen 9 DAT (table 1). Oxadiazon and lactofen applications caused seedling injury by 9 DAT, but the herbicidal effects began to slightly improve by 44 DAT. Similar to the 2 weeks post-sowing treatments, foliar injury caused by pendimathalin applications at 7 weeks post-sowing was delayed, but the injury was less severe (table 1). All four herbicide applications resulted in less foliage fresh weight per seedling compared to the non-treated control with oxyflourfen having significantly less foliage than all treatments at 7 weeks post-sowing (fig. 1).

Muhly grass--Herbicide injury ratings for muhly grass followed a similar trend to those reported for wiregrass seedlings when treated 2 weeks post-sowing (table 2). When treated 7 weeks post-sowing, pendimethalin applications did not injure muhly grass seedlings (table 2). Treatments of oxadiazon and lactofen at the older stage of growth resulted in half the level of damage as seedlings treated 2 weeks post-sowing, and seedlings recovered from the damage to a level “2” rating by 44 DAT. Similarly, oxyfluorfen caused severe damage to seedlings by 8 DAT, but the plants recovered to a rating of “4” by 51 DAT (table 2). Foliage fresh weights were similar among all treatments, with seedlings treated with oxadiazon and pendimethalin having numerically more foliar biomass than non-treated control seedlings (fig. 2).

Outplanted Wiregrass

Applications of oxyfluorfen killed the 1-year-old outplanted wiregrass seedlings by 13 DAT (table 3). Lactofen applications resulted in moderate seedling injury, while clopyralid injury was rated as a “2” from 8 to 25 DAT before being upgraded to a “1” by 33 DAT (table 3). Along with clopyralid, pendimethalin applications did not affect wirergrass.

DISCUSSION

Applying pendimethalin, oxadiazon, lactofen, and oxyfluorfen at the rates used in these trials 2 weeks post-sowing may cause severe damage and death to both wiregrass and muhly grass seedlings. Two-week old seedlings are more susceptible to herbicidal injury because they have not developed much beyond the germinant (embryonic) phase of growth (about 1 to 2 inches tall). Another study reported that treatments of atrazine at 6.35 L/ha 30 days after wiregrass seedling emergence caused severe

Table 1--Mean injury ratings (1 = no injury; 10 = dead) for container-grown wiregrass seedlings on certain days after treatment (DAT) with herbicides at 2 and 7 weeks post-sowing. Means with the same letter within the "51 DAT" column in the 7-week post-sowing trial are not significantly different based on Duncan's Multiple Range Test. Due to a lack of variation, statistics for the 2-week post-sowing trial are not being reported

	-------------2 weeks post-sowing--------------				---------------------7 weeks post-sowing---------------------				
Herbicide	8 DAT	13 DAT	16 DAT	25 DAT	9 DAT	23 DAT	33 DAT	44 DAT	51 DAT
Non-treated	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0 d
Pendimethalin	2.4	5.4	7.6	9.0	1.0	5.8	5.8	5.8	6.2 b
Oxadiazon	9.0	10.0	10.0	10.0	7.8	7.8	7.8	6.0	5.4 bc
Lactofen	9.4	10.0	10.0	10.0	7.2	7.4	7.4	6.0	5.2 c
Oxyfluorfen	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0 a
LSD[a]									0.86

[a]LSD = least significant difference.

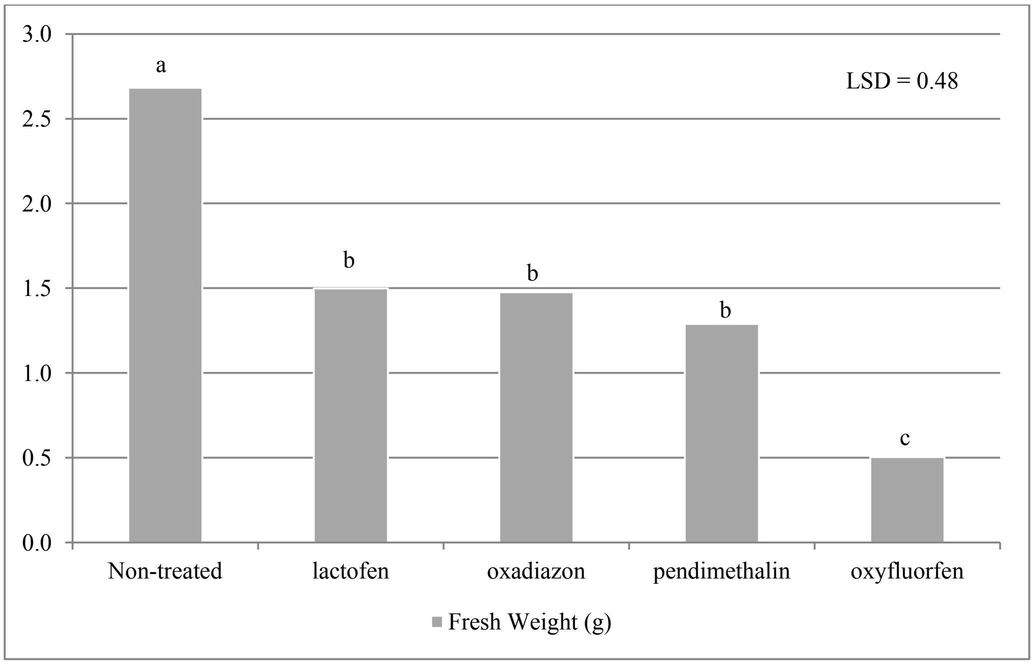

Figure 1--Mean seedling foliage fresh weights (live and dead) for container-grown wiregrass seedlings 51 days after treatment with herbicides at 7 weeks post-sowing. Means with the same letter are not significantly different based on Duncan's Multiple Range Test.

Table 2--Mean injury ratings (1 = no injury; 10 = dead) for container-grown muhly grass seedlings on certain days after treatment (DAT) with herbicides at 2 and 7 weeks post-sowing. Means with the same letter within the "51 DAT" column in the 7-week post-sowing trial are not significantly different based on Duncan's Multiple Range Test. Due to a lack of variation, statistics for the 2-week post-sowing trial are not being reported

	--------------2 weeks post-sowing------------				--------------------7 weeks post-sowing--------------------				
Herbicide	8 DAT	13 DAT	16 DAT	25 DAT	9 DAT	23 DAT	33 DAT	44 DAT	51 DAT
Non-treated	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0 b
Pendimethalin	3.0	7.0	9.0	9.0	1.0	1.0	1.0	1.0	1.0 b
Oxadiazon	9.0	10.0	10.0	10.0	5.0	2.3	1.7	2.0	2.0 b
Lactofen	9.7	10.0	10.0	10.0	5.7	5.0	4.3	2.0	2.0 b
Oxyfluorfen	10.0	10.0	10.0	10.0	8.0	8.0	7.7	6.0	4.3 a
LSD[a]									0.97

[a]LSD = least significant difference.

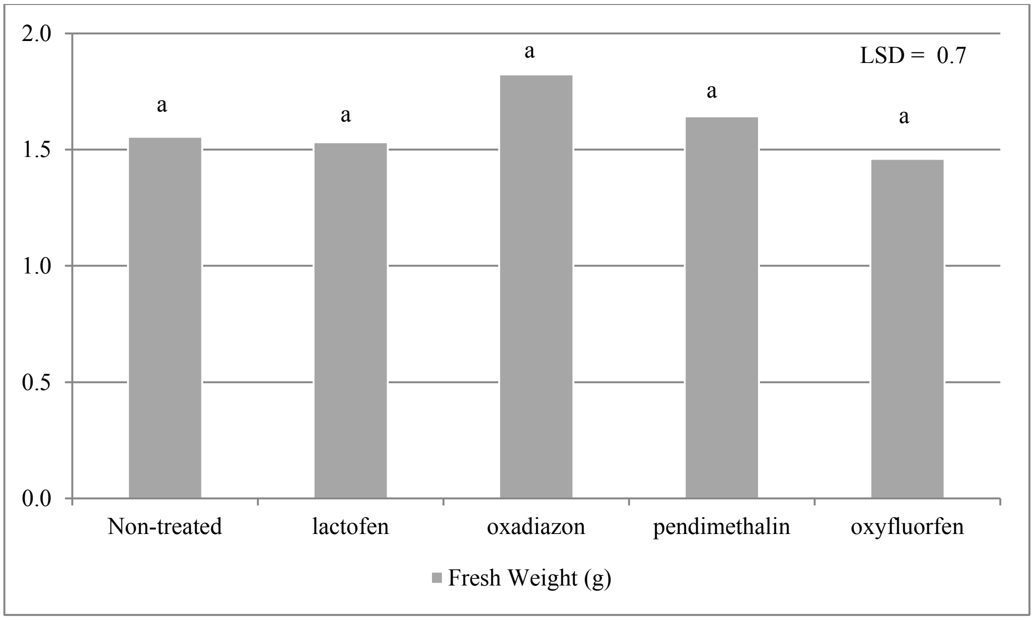

Figure 2--Mean seedling foliage fresh weights (live and dead) for container-grown muhly seedlings 51 days after treatment with herbicides at 7 weeks post-sowing. Means with the same letter are not significantly different based on Duncan's Multiple Range Test.

Table 3--Mean injury ratings (1 = no injury; 10 = dead) for 1-year-old outplanted wiregrass seedlings on certain days after treatment (DAT) with herbicides. Herbicides were applied 6 months after outplanting. Means with the same letter within the "33 DAT" column are not significantly different based on Duncan's Multiple Range Test

Herbicide	8 DAT	13 DAT	16 DAT	25 DAT	33 DAT
Non-treated	1.0	1.0	1.0	1.0	1.0 c
Pendimethalin	1.0	1.0	1.0	1.0	1.0 c
Clopyralid	1.6	1.6	1.6	1.6	1.0 c
Lactofen	3.6	5.8	5.8	5.8	3.6 b
Oxyfluorfen	9.2	10.0	10.0	10.0	10.0 a
LSD[a]					0.32

[a]LSD = least significant difference.

foliar damage and even seedling death, while applications 60 days after emergence caused either no injury or slight injury with the capacity to recover (Kaeser and Kirkman 2010). In that study, wiregrass seedlings also exhibited severe damage to death when treated with aminopyralid (Milestone®), imazapic (Plateau®), imazapyr (Arsenal®), hexazinone (Velpar®), butyric acid (2,4 DB), fluazifop-p-butyl (Fusilade®), and triclopyr (Garlon® 3A) at either 30 or 60 days after emergence (Kaeser and Kirkman 2010).

Oxyfluorfen applications quickly killed the container-grown wiregrass at 2 and 7 weeks post-sowing, muhly grass at 2 weeks post-sowing, and the 1-year-old outplanted wiregrass. Oxyfluorfen has preemergence and postemergence activity on weeds with postemergence applications being the most successful on juvenile plants not exceeding the 2-leaf stage and that are < 4 inches in height. Container seedling applications of oxyfluorfen occurred when plants were < 6 inches in height and more susceptible to herbicide injury. The killing of the 1-year-old outplanted wiregrass confirms that using 32 ounces per acre of the herbicide may be excessive. Lowering the rate of oxyfluorfen in future trials may result in an increased tolerance for wiregrass seedlings.

Muhly grass treated 7 weeks post-sowing were able to tolerate lactofen, oxadiazon, and oxyfluorfen but only after incurring foliar injury by 9 DAT from which they recovered later in the growing season. The ability of muhly grass to tolerate applications of the herbicides better than wiregrass at 7 weeks post-sowing may be attributed to differences in genetics between the two species. This fact highlights the need to develop specific herbicide regimes that can be implemented in nurseries for growing the different species required for longleaf pine ecosystem restoration.

Pendimethalin applications 7 weeks after sowing moderately injured wiregrass and did not affect muhly grass or outplanted wiregrass. Pendimethalin is a preemergence herbicide that interferes with cell division and elongation of emerging shoots and roots during the germination phase and has been known to cause non-target plant injury when applied postemergence to cotton (*Gossypium hirsutum* L.) (Miller and Carter 1980) and loblolly pine (*Pinus taeda* L.) (South and Hill 2009). The increased tolerance of muhly grass to pendimethalin at 7 weeks post-sowing compared to at 2 weeks post-sowing may be explained by the plants having time to develop a more extensive root system and surpass the stage of susceptibility to the pendimethalin. Due to their size, outplanted wiregrass seedlings were not affected by pendimethalin, and at this point it is unclear how early pendimethalin can be safely used over-the-top of container-grown wiregrass during propagation.

Outplanted wiregrass was also injured by lactofen applications but recovered slightly. Thus, these herbicides are not suited for use on either grass species at any growth stage with the rates used in these trials. Oxadiazon and lactofen have preemergence activity during the germination phase and on tender juvenile tissues. This may explain (like with pendimethalin) why seedlings treated 2 weeks post-sowing were killed, and seedlings treated 7 weeks post-sowing were initially affected but then recovered as they grew older and more developed.

Clopyralid performed well over-the-top of outplanted wiregrass. This herbicide is used to control broadleaf weeds in areas where grasses are being grown for seed harvesting, which is the purpose of the outplanted wiregrass at the Claridge Nursery. Clopyralid can be used over-the-top of conifer seedlings in some southern states to control sicklepod [*Senna obtusifolia* (L.) Irwin and Barneby] in bareroot nursery beds (South 2000). Extending its use over-the-top of the outplanted wiregrass would be beneficial in reducing broadleaf weed competition and hopefully increasing annual seed harvests for sowing future wiregrass crops at the nursery.

CONCLUSIONS

These results indicate that, of the applications of oxyfluorfen, lactofen, oxadiazon, and pendimethalin at 2 and 7 weeks post-sowing over-the-top of container-grown wiregrass and muhly grass seedlings, pendimethalin applied at 2.48 L/ha was the only herbicide that did not injure muhly grass seedlings at 7 weeks post-sowing. For outplanted 1-year-old wiregrass seedlings, pendimethalin at 2.48 L/ha and clopyralid at 0.29 L/ha performed well and showed promise as an option to combat weeds that may compete with and reduce seed yields from the wiregrass beds. These results give nursery managers a start on developing an herbicide program against troublesome weeds in the herbaceous understory plant nursery.

Future trials could test some of these same herbicides over-the-top of wiregrass, muhly grass, and other herbaceous understory species such as little bluestem or Indian grass but at lower rates and different growth stages. Other herbicides should be identified and tested that target problematic weeds. For instance, halosulfuron (Sedgehammer®) has activity on sedges and oxyfluorfen (Goal Tender®) is one that may offer more flexibility when applied over younger herbaceous seedlings.

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LITERATURE CITED

Brockway, D.G.; Lewis, C.E. 1997. Long-term effects of dormant-season prescribed fire on plant community diversity, structure and productivity in a longleaf pine wiregrass ecosystem. Forest Ecology and Management. 96(1-2): 167-183.

Jose, S.; Jokela, E.J.; Miller, D.L., eds. 2006. The longleaf pine ecosystem: ecology, silviculture, and restoration. New York: Springer. 438 p.

Kaeser, M.J.; Kirkman, L.K. 2010. The effects of pre- and post-emergent herbicides on non-target native plant species of the longleaf pine ecosystem. Journal of the Torrey Botanical Society. 137(4): 420-430.

Lopez, R.; Burns, N.; Gjerstad, D. [and others]. . 2009. Range-wide conservation plan for longleaf pine. http://www.americaslongleaf.org/media/86/conservation_plan.pdf. [Date accessed: July 15, 2012].

Miller, J.H.; Carter, C.H. 1980. Performance of six substituted dinitrobenzamine herbicides applied at layby of cotton (*Gossypium hirsutum*). Weed Science. 28(2): 212-215.

South, D.B. 2000. Tolerance of southern pine seedlings to clopyralid. Southern Journal of Applied Forestry. 24(1): 51-56.

South, D.B.; Hill, T.E. 2009. Results from six *Pinus taeda* nursery trials with the herbicide pendimethalin in the USA. Southern Forests. 71(3): 179-185.

THE EFFECT OF HERBACEOUS WEED CONTROL ON PLANTED LOBLOLLY PINE DURING A DROUGHT

John D. Kushla[1]

Abstract-- Seedling survival in loblolly pine (*Pinus taeda* L.) plantation establishment is often mentioned as a justification for herbaceous weed control (HWC). However, the effects of HWC treatment during drought have been difficult to find. Sometimes this research was proprietary in nature. Also, since weather patterns vary from year to year, drought may not have coincided with a research study.In the spring of 2007, a demonstration for a HWC comparison was installed on retired pasture at the North Branch Station near Holly Springs, MS. Second-generation loblolly seedlings from Weyerhaeuser were planted using a 10- by 10-foot spacing on March 27. Approximately half the area received HWC, and the other half did not (control). Broadcast HWC was simulated with a two-pass operation approximately 1 month after planting. The first pass was a 4-ounces Arsenal AC® plus 2-ounces Oust® mix sprayed at 15 gallons per acre (GPA) in a 5-foot band over the seedlings. The second pass was 32 ounces glyphosate in 15 GPA sprayed between the rows. Three measurement plots of one-tenth acre were randomly located on each treatment: control (no treatment) versus broadcast HWC. Initial measurements were taken August 14, 2007. Average survival on control plots was 84.4 percent, and 83.9 percent on treated plots. There was a late summer drought that growing season, so the study was measured again in mid-February 2008. Average stocking on control plots was 230 trees per acre, and survival was 37.1 percent. Treated plots, on the other hand, had an average 433 trees per acre and 74.7 percent survival. Broadcast HWC can affect survival of young pine plantations during drought years.

INTRODUCTION

The literature on vegetation management in loblolly pine (*Pinus taeda* L.) plantations is quite extensive. The Competition Omission Monitoring Project (COMP) evaluated different levels of weed and brush control during plantation development across the southeastern United States (Miller and others 1991; Miller and others 1995a, 1995b; Shiver and others 1991; Zutter and Miller 1998). This region-wide study evaluated the effects of herbaceous, woody, and complete versus no vegetation control on the development of loblolly pine plantations.

The advantages of HWC during plantation establishment included improved growth. However, documentation of the effects of HWC on pine seedling survival is rare. HWC treatment was shown to improve survival, but its effects varied by site, treatment area, and weather conditions (Dougherty and Lowery 1991, Miller and others 1991). The coincidence of drought during study establishment, and the proprietary nature of herbicide research partly explain this gap in the literature. This study compared survival effects of HWC treatment on loblolly plantation establishment during an actual drought in northern Mississippi.

METHODS

The study site was on the North Branch Station of the Mississippi Agriculture and Forestry Experiment Stations near Holly Springs, which has been described in detail by Kushla (2009). The site was retired Bermuda-grass bermudagrass [*Cynodon dactylon* (L.) Pers.] and tall fescue [*Schedonorus arudinaceus* (Schreb.) Dumort., nom. cons.] pasture from a former dairy operation. The fields were gently undulating hills (2 to 5 percent slope) consisting predominantly of Loring silt loam. The area comprised approximately 30 acres with a small drainage bisecting the site.

Planting was completed March 21, 2007 on a 10- by 10-foot spacing to demonstrate wildlife habitat early in plantation establishment. Broadcast HWC was added as a further comparison. The southern side of the drainage was sprayed, and the northern side was not. Since this was a demonstration, no statistical design was used.

Broadcast HWC treatment was simulated with a two-pass operation. Approximately 1 month after planting, a tank mix of 4-ounces Arsenal AC® plus 2-ounces Oust® mix was sprayed at 15 gallons-per-acre (GPA) over the top of planted seedlings in a 5-foot band over the seedlings. Afterward, 32 ounces glyphosate was sprayed in 15 GPA between rows.

On August 14, 2007, initial measurements were taken. Three one-tenth-acre measurement plots

[1]Associate Professor, Mississippi State University, North Mississippi Research and Extension Center, Verona, MS 38879.

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were randomly located in each treatment area (broadcast HWC or control). Initial measurements were for survival and stocking.

Subsequent measurements on the same plots were taken in mid-February 2008. These measurements included groundline diameter (GLD), total seedling height (HT), stocking, and survival. Weather data including temperature, precipitation, and growing season evaporation were collected on a daily basis according to standard procedures by an on-site National Weather Service/ National Oceanic and Atmospheric Administration weather station.

RESULTS

Initial stocking and survival measurements appear in table 1. While 10- by 10-foot spacing was the target, average actual stocking was a little higher on both treatment areas. As shown, average stocking and survival were comparable on both treatment areas in August 2007.

Table 1--Average initial loblolly stocking and survival measurements taken August 14, 2007

Treatment	Average stocking	Average survival
	trees/acre	*percent*
Broadcast HWC	486	83.9
Control	523	84.4

Average air and soil temperatures are presented in figure 1, which included actual temperatures with their historical 40-year average. Average monthly air temperatures tracked very closely to their respective 40-year averages. However, average monthly soil temperatures were well above their 40-year average. Through examination of the average maximum air temperatures, an explanation was found. Daytime maximum air temperatures during the summer of 2007 were considerably hotter than their 40-year average, despite the fact that average air temperatures appeared normal.

Monthly precipitation and evaporation are presented in figure 2, including the historical 80-year average monthly precipitation. For the most part, the spring and early summer were dry by historical standards, with the exception of July. The station experienced two significant rain events on July 11 and 13, totaling just over 5 inches of precipitation. After July 13, however, drought conditions were encountered. For 10 weeks thereafter, weekly rainfall was below 1 inch. In fact, there was no recordable rainfall during 4 of those weeks. Total rainfall for July through September was 10.01 inches, while evaporation that same period was -22.3 inches. Consequently, the drought coincided with a time of unseasonably high maximum temperatures on an historical basis.

First-year seedling GLD, HT, stocking, and survival measurements were taken during mid-February 2008 and are summarized in table 2. There were notable differences in average seedling GLD, HT, stocking, and survival between treatment areas. Seedlings receiving broadcast HWC on average were bigger and survived better than control seedlings. Although average survival was only 75 percent in the treated area, the average stocking of 433 trees per acre was very close to the original target stocking for a 10-by 10-foot spacing. Conversely, seedlings without HWC treatment were on average smaller and survived drought poorly (only 37 percent survival). The average stocking of 230 trees per acre would have required re-planting.

DISCUSSION AND MANAGEMENT IMPLICATIONS

Although just an operational comparison, broadcast HWC appeared effective in this study. Treated loblolly seedlings were larger and survived a severe drought. As the COMP and other studies showed, grass is very competitive to newly planted loblolly seedlings (Dougherty and Lowery 1991; Miller and others 1991; Miller and others 1995a , 1995b; Zutter and Miller 1998).

The difference in survival and stocking between treated and non-treated seedlings meant the difference between plantation establishment and failure. Despite the lack of statistical design, the evidence is very compelling that broadcast HWC is effective to assure plantation survival during drought.

With increasing concern for the effects of climate change on forested ecosystems, there are opportunities to continue vegetation control research in southern pine plantation establishment. Increasing maximum temperatures and occurrence of localized

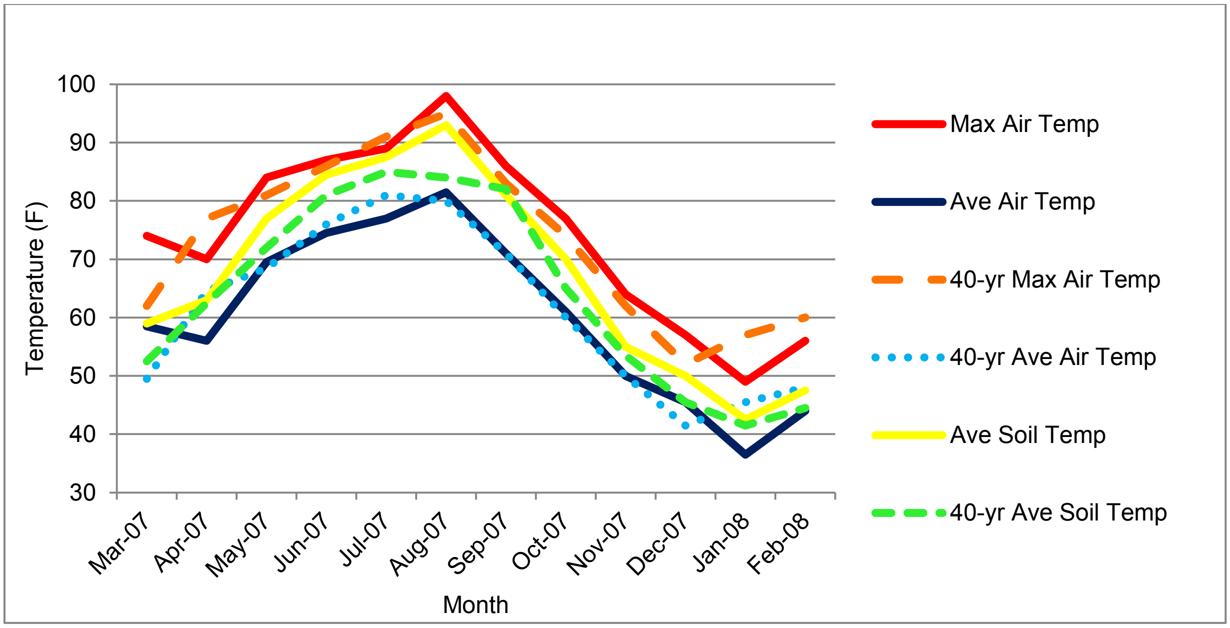

Figure 1--Monthly average temperatures March 2007 through February 2008, Holly Springs, MS.

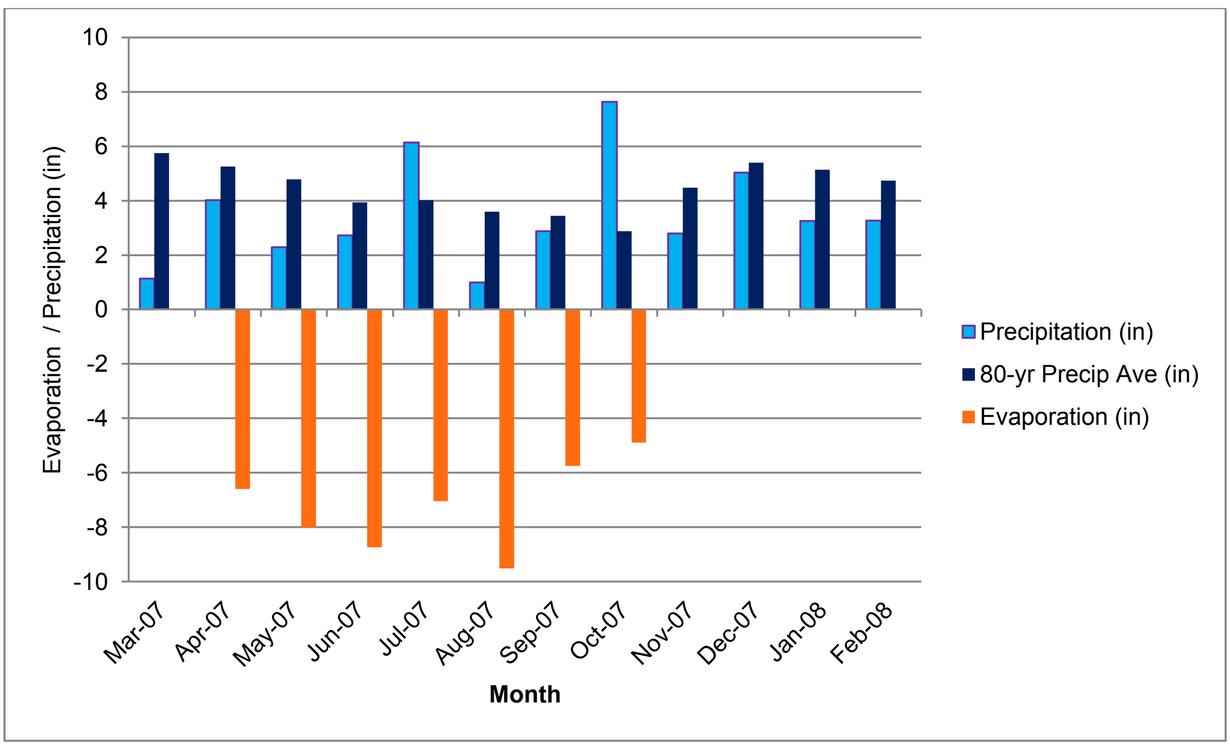

Figure 2--Monthly evaporation and precipitation March 2007 through February 2008, Holly Springs, MS.

Table 2--First-year loblolly seedling average measurements

Treatment	Average GLD	Average Height	Average Stocking	Average Survival
	inches (mm)	*inches (cm)*	*trees/acre*	*percent*
Broadcast HWC	0.347 (8.82)	18.6 (47.2)	433	74.7
Control	0.232 (5.89)	11.5 (29.3)	230	37.1

droughts could have profound effects on plantation establishment in retired pastures.

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LITERATURE CITED

Dougherty, P.M.; Lowery, R.F. 1991. Spot-size of herbaceous control impacts loblolly pine seedling survival and growth. Southern Journal of Applied Forestry. 15(4): 193-199.

Kushla, J.D. 2009. Afforestation in north Mississippi on retired farmland using *Pinus echinata*: first-year results. Southern Journal of Applied Forestry. 33(3): 142-144.

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 COMProject. 1995b. Gen. Tech. Rep. SO-117. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 52 p.

Shiver, B.D.; Knowe, S.A.; Edwards, M.B.; Kline, W.N. 1991. Comparison of herbicide treatments for controlling common coastal plain flatwoods species. Southern Journal of Applied Forestry. 15(4): 187-193.

Zutter, B.R.; Miller, J.H. 1998. Eleventh-year response of loblolly pine and competing vegetation to woody and herbaceous plant control on a Georgia flatwoods site. Southern Journal of Applied Forestry. 22(2): 88-95.

CHOPPER GEN2 + GLYPHOSATE EFFICACY FOR HEIGHT CLASSES OF HARDWOOD SPROUTS RECOLONIZING SIX CLEARCUT PINE SITES

Jimmie Yeiser and Andrew Ezell[1]

The purpose of this study was to assess sprout size as a determinant of subsequent control by a standard, single rate of imazapyr + glyphosate applied during site preparation. All study sites were in the hilly upper coastal plain of Mississippi (Winston or Oktibbeha Counties) or Louisiana (Sabine or Winn Parishes) and supported loblolly pine (*Pinus taeda* L.) plantations prior to installation of this project. In total, six sites, three in Mississippi and three in Louisiana, were tested. In Mississippi in 2007, two sites were tested, one with a harvest date before August 1 and one with a harvest date after August 1. In 2008, one test site was selected with indifference to harvesting date. The same design was followed in Louisiana. The Mississippi pre-August 1 site was lost to operational overspray.

An untreated check and four herbicide treatments (five total treatments) were installed at each site. Herbicide was applied for site preparation at four defined stages in hardwood sprout development. Treatments were: (1) untreated check (no herbicide); and (2) 0-foot (was applied to bare ground prior to hardwood sprouting). Subsequent treatment timings were applied when hardwood sprouts reached the following height classes: (3) 0.5- to < 1-foot; (4) 1- to < 2-feet; and (5) 3- to 4-feet. Sites harvested prior to August 1 received all 5 treatments. Sites harvested after August 1 received only treatments 1, 2, and 3. Herbicide treatments were as follows: 40 ounces of Chopper GEN2 only on treatment 2 and 40 ounces of Chopper GEN2 plus 2 quarts Accord plus 1 percent NIS per acre on all other treatments.

Treatment plots were 30- by 100-feet; measurement plots were 10- by 80-feet and centered in treatment plots. Herbicides were applied with a backpack pole sprayer equipped with a KLC-9 flood nozzle in a total volume of 15 gallons per acre. Herbicides were mixed and sprayed immediately.

Sweetgum (*Liquidambar styraciflua* L.) and mixed oaks (*Quercus* spp.) were present at all test sites. Oak species varied by site but were commonly water (*Q. nigra* L.), willow (*Q. phellos* L.), southern red (*Q. falcata* Michx.), white (*Q. alba* L.), and post (*Q. stellata* Wangenh.) oaks. Non-sweetgum and non-oak species were lumped into a category named 'other'.

Mississippi sites had three replications in a completely randomized design. In Louisiana, sites had four blocks in a completely randomized block design. Efficacy of treatments was evaluated based on the total cumulative linear height of all hardwood sprouts evaluated 0, 1, and 2 years after treatment. Year 2 results are presented here. Ratios were computed using the cumulative linear heights for the untreated check in the numerator and the treatment of interest in the denominator.

The cumulative linear height classes for sprouts are presented in table 1. Two years after treatment, the cumulative linear heights of sweetgum, oak, and other species were reduced by treatments. Values in table 1 may be visually separated into three groups: (1) the untreated check; (2) bare ground and sprout height class 0.5- to < 1-foot; and (3) sprout height classes 1- to < 2-feet and 3- to 4-feet.

Growth ratios show the ability of treatments to reduce and stunt sprout growth (table 1). Using other species as an example, cumulative linear sprout height ratios for untreated checks and

[1]Provost/Vice Chancellor for Academic Affairs, University of Arkansas at Monticello, Monticello, AR 71656; and Professor, Mississippi State University, Department of Forestry, Mississippi State, MS 39762.

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Table 1--Total cumulative heights and ratios for sprout classes receiving a single application of 40 ounces of Chopper GEN2 + 2 quarts Accord + 1 percent NIS per acre and totaled for 3 sites in Louisiana and 2 sites in Mississippi two growing seasons after treatment

Sprout height classes	------Cumulative heights--------			---------------Ratios---------------		
	Oak	Sweetgum	Other	Oak	Sweetgum	Other
	---------------*feet*-----------------					
Untreated check	162.2	166.6	790.4			
0 (bare ground)	39.9	31.0	171.7	4.1	5.4	4.6
0.5 to < 1.0	31.9	41.8	132.2	5.1	4.0	6.0
1.0 to < 2.0	3.7	6.6	29.9	43.8	25.2	26.4
3 to 4	4.8	3.4	40.9	33.8	49.0	19.3

treatments with sprouts < 1.0-foot tall (treatments 1 and 2) averaged 5.2 (e.g. 790.4/152.0 = 5.2) (table 1). The same ratio between untreated checks and sprouts > 1.0-foot tall is 22.3 (e.g. 790.4/35.4 = 22.3). Values of 5.2 and 22.3 are in sharp contrast and illustrate the potential growth surge of unwanted woody sprouts < 1.0-foot tall not appropriately treated with this one rate of imazapyr + glyphosate. That is, for the same cost, by waiting until sprouts are > 1.0-foot tall, managers can significantly reduce hardwood competition beyond that achieved by spraying bare ground or sprouts < 1.0-foot tall.

Total hardwood linear height (feet) for all species and sites was 3,952; 859; 661; 150; and 205 feet for treatments 1 through 5, respectively. As these values show, an herbicide application to bare ground (treatment 2) was better than doing nothing (treatment 1); however, control of unwanted hardwoods increased as sprout height at application increased (treatments 3, 4, and 5). Results suggest there is substantial benefit to allowing hardwood competition to sprout prior to herbicide application. Allowing sprouts to exceed 1 foot in height appears to be the threshold necessary to optimize hardwood control, without increasing herbicide rate or cost.

Forest Threats

Moderator:

Kristina Connor
USDA Forest Service
Southern Research Station

EVALUATING GROWTH EFFECTS FROM AN IMIDACLOPRID TREATMENT IN BLACK WILLOW AND EASTERN COTTONWOOD CUTTINGS

Luciano de Sene Fernandes, Ray A. Souter, and Theodor D. Leininger[1]

Black willow (*Salix nigra* Marsh.) and eastern cottonwood (*Populus deltoides* Bartram ex Marsh.), two species native in the Lower Mississippi Alluvial Valley, have importance in short rotation woody crop (SRWC) systems for biomass production (Ruark 2006). For these tree species, the cottonwood leaf beetle (*Chrysomela scripta* Fabricius) is one of the most serious pests of young trees in nurseries and plantations (Anonymous1989). Defoliation during infestations by this beetle is documented to impact height growth of cottonwood in newly established plantings (Coyle and others 2002). Nebeker and others (2006) reported significant reductions in a measure of volume production of approximately 50 percent after two growing seasons occurring in untreated trees as compared to trees protected from insect damage by using chemical control. Losses of this magnitude indicate a risk to the economic viability of SRWC, and the use of a treatment to control potential loses may be advisable. Applied as a safeguard against possible insect infestations, the establishment practice of treating cuttings prior to planting with the systemic insecticide, imidacloprid, is recommended (Bayer CropScience 2009). Besides the benefits of insect control, there are reports of enhanced growth rates in material treated with imidacloprid, with effects extending into a second growing season (Robison and Rousseau 2007). Growth enhancement may be due to modified physiological processes expressed in both stressed and optimal nutrient and water regimes (Chiriboga 2009). Enhanced growth of cuttings treated with imidacloprid could offset the cost of this treatment and enhance the economic viability of SRWC production.

In spring 2010, an operational experiment designed to examine SRWC production systems and harvest methods was initiated in Ouachita Parish, Louisiana (USDA FS-SRS-4155-2011 study plan on file. Center for Bottomland Hardwoods Research, 432 Stoneville Road, Stoneville, MS 38776). That experiment included both black willow and eastern cottonwood cuttings. A standard treatment with imidacloprid was included in the protocol as a common practice for preparation of all cuttings prior to planting in order to provide protection from defoliation by cottonwood leaf beetle. Approximately 40,000 cuttings of each tree species were treated according to protocol. Black willow cuttings, collected in the winter and stored under water for several weeks, were produced from whips of naturally regenerated willow collected adjacent to the Holt Collier National Wildlife Refuge near Hollandale, MS. Eastern cottonwood cuttings were produced immediately from freshly collected whips of Texas clone S7C8 grown by Big River Nursery, Winnsboro, LA. After production, all cuttings were kept refrigerated until needed for planting. Shortly before planting, cuttings were treated with a particular formulation of imidacloprid applied according to the label instructions as a 24 hour soak of 16 fluid ounces per 100 gallon solution for the partially hydrated black willow cuttings or a 24 hour soak of 8 fluid ounces per 100 gallon solution for the unhydrated eastern cottonwood cuttings (Bayer CropScience 2009).

In order to examine the magnitude of enhanced growth rates that may derive from this imidacloprid treatment of cuttings, a study was initiated in Stoneville, MS. This study, ancillary to the operational experiment, examined the height growth and basal area production of black willow and eastern cottonwood plantations established in May 2010. A small, random selection of cuttings was obtained from those produced

[1]International Forestry Research Intern, Research Forester, and Supervisory Research Plant Pathologist, respectively, USDA Forest Service, Southern Research Station, Stoneville, MS, 38776.

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Table 1—Growth analysis following second-year growing season for plot-level average tree height (H) and plot-level total basal area production (B) comparing cuttings treated prior to planting with a 24 hour soak in an imidacloprid solution versus untreated controls. Average tree height was calculated for surviving trees only, and total basal area production is summed over surviving trees effectively assigning zero growth to those that died during the study. Note that total basal area is not expressed per unit area. A positive treatment difference indicates that the treatment mean is greater than the control

Species	Analysis variable	Overall mean	Treatment difference	Treatment MS	Error MS	F-test significance level
Willow	H (*feet*)	13.89	-0.57	0.49874	0.00537	0.0108
	B ($feet^2$)	0.81	-0.10	0.01460	0.01580	0.4378
Cottonwood	H (*feet*)	17.70	0.32	0.15681	0.22032	0.4877
	B ($feet^2$)	1.15	0.15	0.03302	0.06326	0.5451

according to protocol for the operational experiment, as well as a selection of cuttings that were not treated and held in reserve to be used as controls (300 cuttings for each species for each treatment). With each species examined in independent trials, the experimental design was a randomized complete block with two treatments (control and imidacloprid treated) and three replications. Each experimental unit was composed of 100 cuttings, planted on a 20 by 5 rectangular pattern with 5 feet between planting spaces in both dimensions. Following each growing season, all tree heights and diameters at breast height (4.5 feet) were obtained. Following the second growing season, insect damage ratings were assessed by recording the presence/absence of shoot tip damage and dieback of the season's terminal growth.

The analyses of the measurements obtained in October 2011 following the second growing season are reported here. The incidence of insect damage was unremarkable in the first growing season based on casual assessments throughout the season. During the second growing season, cottonwood leaf beetle was absent or present only at very low levels. For the insect damage assessment, in both species there were no differences among treatments. Populations of defoliating insects have a high level of variation in spatial and temporal occurrence, and at this location during the conduct of this study, populations were very low. This condition should provide for an unconfounded comparison of enhanced growth due to the physiological impacts of the imidacloprid treatment, in the absence of any growth impacts due to defoliation. The only statistically significant result (at $\alpha = 0.05$) was obtained for the average height of black willow trees, with the untreated control plots having a 6.84 inches taller height than the average in treatment plots (table 1). This increase is not considered to be of a practical magnitude. The basic conclusion is that using imidacloprid under operational constraints may not provide enhanced growth performance, though trees were not considered to be stressed at any time during the two growing seasons and defoliation was not problematic.

LITERATURE CITED

Anonymous. 1989. Insects and diseases of trees in the South. 1997 reprint. Protection Rep. R8-PR16. Atlanta: U.S. Department of Agriculture Forest Service, Southern Region. 98 p.

Bayer CropScience. 2009. Poplar/cottonwood cuttings/whip application recommendations. ADMIRE® PRO systemic protectant label: 62.

Chiriboga, A. 2009. Physiological responses of woody plants to imidacloprid formulations. Columbus, OH: The Ohio State University. 130 p. M.S. thesis.

Coyle, D.R.; McMillin, J.D.; Hall, R.B.; Hart, E.R. 2002. Cottonwood leaf beetle (Coleoptera: *Chrysomelidae*) defoliation impact on *Populus* growth and above-ground volume in a short-rotation woody crop. Agricultural and Forest Entomology. 4: 293-300.

Nebeker, T.E.; Stone, W.D.; Beatty, T.K. 2006. Impact of herbivory by cottonwood leaf beetle on three selected cottonwood clones: year 2 results. 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: 420-423.

Robison,T.L.; Rousseau, R.J. 2007. Cottonwood leaf beetle control with imidacloprid soaked cuttings. In: Buckley, D.S.; Clatterbuck,W.K., eds. Proceedings, 15th central hardwood forest conference. e-Gen. Tech. Rep. SRS-101. [CD-ROM]. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 198-205. http://www.treesearch.fs.fed.us/pubs/27398. [Date accessed: December 5, 2011].

Ruark, G. 2006. Woodys have biomass potential. Inside Agroforestry. 15(3). Lincoln, NE: U.S. Department of Agriculture, National Agroforestry Center: 3, 11.

INVASIBILITY OF MAJOR FOREST TYPES BY NON-NATIVE CHINESE TALLOW IN EAST TEXAS

Zhaofei Fan[1]

Abstract--Non-native invasive Chinese tallow trees [*Triadica sebifera* (L.) Small,formerly *Sapium sebiferum* (L.) Roxb.] are rapidly spreading into natural ecosystems such as forests in the southeastern United States. Using the 2001-2010 USDA Forest Service's Forest Inventory and Analysis (FIA) data and forest land cover data, we estimated the regional invasibility of major forest types (groups), loblolly/shortleaf pine forests and oak/gum/cypress forests, by using Geographic Information Systems (GIS) and geostatistical tools. We defined the regional invasibility of a forest ecosystem as its susceptibility to the colonization and establishment of Chinese tallow, measured by a function of tallow presence and cover percent. The invasibility of these two major forest types to tallow has been estimated and potential management applications discussed.

INTRODUCTION

The rapid spread of Chinese tallow [*Triadica sebifera* (L.) Small,formerly *Sapium sebiferum* (L.) Roxb.] is a significant concern in the southeastern United States, for it displaces native species, changes natural soil conditions, and creates a transformation of community structure from grassland to woodland (Battaglia and others 2009, Bruce and others 1995). Wang and others (2011) utilized Forest Inventory and Analysis (FIA) data from the USDA Forest Service to simulate the expansion of Chinese tallow (hereafter referred to as tallow) in Texas and Louisiana. They estimated that over 1.5 million ha of forest lands could be occupied by tallow by the year 2023. Further, their study estimated that tallow could migrate over 300 km into areas as far north as 34 °N latitude within the next 115 years (100 years if temperatures increase by 2 °C).

For invasive species research and management, it is a challenge to understand the invasion process and associated driving factors and to design effective methods for controlling the spread of tallow and reducing ecological and economic loss. The objective of this study is to quantify the invasibility of major forest communities by tallow. The information will be useful to control and mitigate the spread of tallow trees in southern forested lands.

METHODS

The study area is confined to east Texas where tallow has a relatively longer invasion history and is more prevalent in the target forests (fig. 1). From 2001 to 2008, a total of 2,426 FIA plots were measured in east Texas among which 454 plots were found to be infested by tallow with an overall probability of presence of 19 percent. Of the 2,426 FIA plots, there were 2,224 that were measured in both consecutive cycles (1999-2004 and 2005-2008), and 190 of the plots were found to be newly infested by tallow in the 2005-2008 period. This represented an overall annual probability of spread of 1.8 percent. Classification and regression tree (CART) was used to classify the study region into a set of homogenous, stationary sub-regions of similar spread patterns of tallow. Within each homogeneous sub-region, both the newly infested and all infested FIA plots were extracted, separately by forest type group and the spread rate (change of presence probability). The cover change of tallow in different forest types/communities was calculated from this information. To assess the invasibility of a forest community, we calculated the means of presence probability and cover percent by CART-identified sub-region. A simple linear regression was used to analyze the relationship between cover percent (response variable) and presence probability (independent variable). The slope (increase of cover percent per unit of change of presence probability) of the regression was used to measure the "relative invasibility" of a forest community.

[1]Associate Professor, Mississippi State University, College of Forest Resources, Department of Forestry, Mississippi State, MS 39762.

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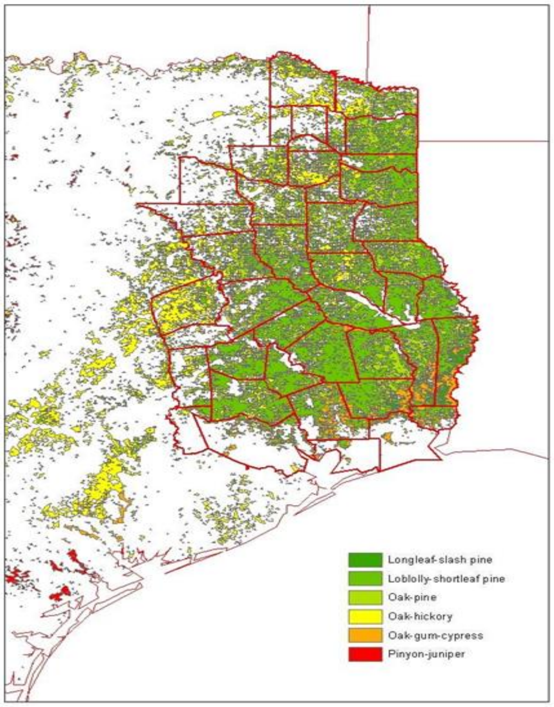

Figure 1--Distribution of forest covers/communities in east Texas

RESULTS AND DISCUSSION

As reported by Fan and others (2012), east Texas was classified into four relatively homogeneous spread regions: (1) region I, latitude ≤ 30.4˚ and longitude ≤ -94.9˚ (most prevalent); (2) region II, latitude ≤ 30.4˚ and longitude > -94.9˚ (highly prevalent); (3) region III, latitude > 30.4˚ and longitude > -94.9˚ (moderately prevalent); and (4) region IV, latitude > 30.4˚ and longitude ≤ -94.9˚ (less prevalent). This spread pattern was attributed to climate (e.g., low temperature), site slope, distance to roads and infested plots, forest type, stocking, physiographic condition, human activity, and natural disturbances such as wind and fire. However, the impact of these factors varied spatially. Among the identified contributing factors, forest type proved to affect tallow spread regionally (Fan and others 2013). Within the two dominant forest type (groups), the oak/gum/cypress forest (annual infestation probability = 3.5 percent) was more sensitive to invasion than the loblolly/shortleaf pine forest (annual infestation probability = 1.8 percent). Due to many confounding factors, the cover change of tallow *per se* is not appropriate to quantify the invasibility of different forest communities (types). Instead, the slope of the regression model of tallow's cover change against its presence probability change from each homogeneous spread region was used. The slope of the regression model for the oak/gum/cypress forests was nine times larger than that of the loblolly/shortleaf pine forest, indicating the former was highly susceptible to tallow compared to the latter (fig. 2). The slope coefficient considered both the invasion (introduction and spread) and growth (colonization and establishment) processes and balanced the potential confounding factors in the regression model. Therefore, it is more reliable to evaluate the general invasibility of a forest community, and useful comparisons of invasibility (or susceptibility) among different forest communities and environments can be made.

CONCLUSION

The invasibility (susceptibility) of major forest types/groups to invasive species varies by a number of contributing factors that affect the invasion and growth process. But the average invasibility can be estimated by a relative measure, the slope coefficient of the regression lines of cover (dominance) change versus presence (abundance) data. By the slope coefficient, the oak/gum/cypress forest is more invasive or susceptible to tallow than the loblolly/shortleaf pine forest.

LITERATURE CITED

Battaglia, L.L.; Denslow, J.S.; Inczauskis, J.R.; Baer, S.G. 2009. Effects of native vegetation on invasion success of Chinese tallow in a floating marsh ecosystem. Journal of Ecology. 97(2): 239-246.

Bruce, K.A.; Cameron, G.N.; Harcombe, P.A. 1995. Initiation of a new woodland type on the Texas coastal prairie by the Chinese tallow tree [*Sapium sebiferum* (L.) *Roxb*.]. Bulletin of the Torrey Botanical Club. 122: 215-225.

Fan, Z.; Tan, T.; Crosby, M.K. 2012. Factors associated with the spread of Chinese tallow in east Texas forestlands. Open Journal of Ecology. 2(3): 121-130.

Fan, Z.; Moser, W.K.; Hansen, M.H.; Nelson, M.D. 2013. Regional patterns of major non-native invasive plants and associated factors in upper midwest. Forest Science. 59 (1): 38-49.

Wang, H.H.; Grant, W.E.; Swannack, T.M. [and others]. 2011. Predicted range expansion of Chinese tallow tree (*Triadica sebifera*) in forestlands of the southern United States. Diversity and Distributions. 17: 552-565.

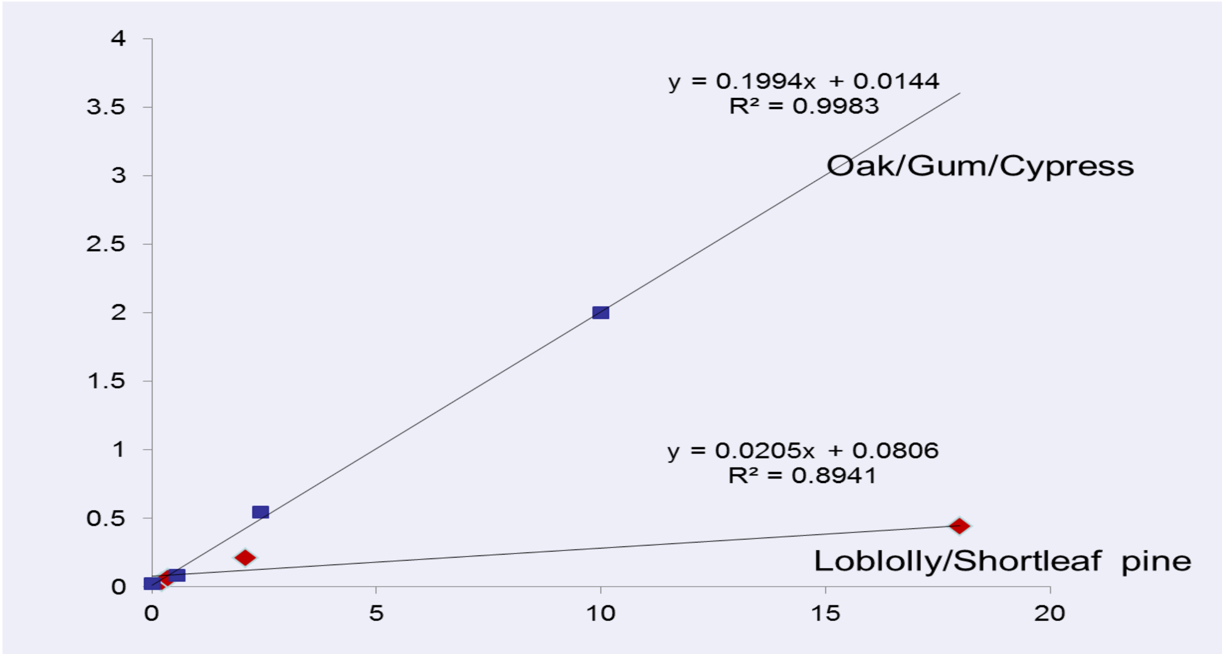

Figure 2--Regression lines of annual increase of cover (%, Y-axis) vs. presence probability (%, X-axis) of tallow by forest type (group).

MISSISSIPPI'S COMPREHENSIVE PROGRAM FOR SOUTHERN PINE BEETLE PREVENTION: EXTENSION FORESTRY'S ROLE AND THE ECONOMIC CONTRIBUTION

John D. Kushla, Stephen G. Dicke, Jason S. Gordon, James E. Henderson, and Andrew J. Londo[1]

The southern pine beetle (SPB), *Dendroctonus frontalis* Zimmermann, has a history as an extremely damaging pest to southern pine forests (Coulson and Klepzig 2011). Extension Forestry at Mississippi State University (MSU) initiated the SPB Prevention Program in collaboration with the Mississippi Forestry Commission to reduce the threat of SPB outbreak through education and thinning activities. Funding came from the USDA Forest Service for the years 2006-2009 and 2011, and the American Recovery and Reinvestment Act (ARRA) for 2010.

As background, Extension Forestry at MSU has a long history of conducting educational outreach activities to landowners, foresters, and loggers. There is a statewide network of 68 county forestry associations. We have had great success using the tax roll to advertise short courses and workshops (Londo and others 2008). Educational programming for SPB Prevention included evening presentations, short courses, workshops, radio shows, and site visits.

For the SPB Prevention Project, we conducted 260 educational programs to 12,457 participants and distributed 134,277 copies of publications. In addition, we made 359 site visits. Expenditures for educational outreach 2006-2011 (table 1) were nearly $1.54 million. This included funding for three staff positions, travel, meals at programs, and mensuration equipment for county forestry associations. Topics presented under the project included SPB biology and prevention, pine plantation thinning, forest health, species selection, prescribed burning, thinning cost share, and site visits.

In 2008, the SPB Prevention Project expanded to include a thinning cost-share program. That year the cost shares were available for north Mississippi and went statewide in 2009. The first thinning could be pre-commercial or commercial. The thinning cost share targeted pine stands with medium to high beetle-hazard rating. Most approved stands were commercially thinned, and about 5 percent were pre-commercially thinned.

From table 1, total expenditures for the thinning cost share was over $1.81 million, paid to landowners, loggers, and foresters. For the years 2008, 2009, and 2011, thinning was completed on 17,755 acres of pine stands for SPB prevention through this project. In these years, $1,283,968 cost shares were paid to forest landowners. However, since 2011, the project scope has been limited to south Mississippi.

For the year 2010 only, American Recovery & Reinvestment Act (ARRA) funds provided cost sharing to foresters and loggers for job retention but not to landowners. Already-scheduled stands for first thinning were used pay cost shares to these professionals. Pine stands approved for thinning cost share required a Mississippi Forest Stewardship plan, which was subsidized with ARRA funds. Stewardship plans on 91,932 acres were completed with $229,830 paid in cost shares to foresters, retaining 57 jobs. That year, first thinning was done on 3,709 acres paying $296,738 to loggers and retaining 232 jobs.

Meanwhile, input-output analysis using the IMPLAN software (Minnesota IMPLAN Group

[1]Associate Professor, Professor, Assistant Professor, and Assistant Professor, respectively, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; and Assistant Director of Agriculture and Natural Resources, Ohio State University Extension, College of Food, Agriculture, & Environmental Sciences, Columbus, OH 43210.

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Table 1--Summary statistics for the SPB Prevention Project

Year	Educational outreach	Thinning cost share	Budgeted thinning
	----------------------------$--------------------------------		*acres*
2006	238,619	0	0
2007	304,792	0	0
2008	309,169	334,494	6,276
2009	299,108	300,892	6,110
2010	285,663	526,568	0
2011	101,418	648,582	5,369

Table 2--Input-output analysis for 2008-2009 thinning cost share

Impact type	Employment	Labor income	Value-added	Output
		--------------------------------$------------------------------		
Direct Effect	32.0	1,326,365	2,171,828	7,328,337
Indirect Effect	28.5	1,141,825	1,855,765	4,860,042
Induced Effect	20.8	633,763	1,254,354	2,195,103
Total	81.3	3,101,953	5,281,947	14,383,482

2010) was conducted to estimate the total economic activity resulting from direct, indirect, and induced impacts of thinning activities and cost-share payments. Direct effects resulted from demand changes within this sector. Indirect effects reflected response of other market sectors for goods and services to initial spending in the targeted sector. Induced effects occurred from consumer spending in the targeted sector for goods and services. Each effect was estimated for employment, wages, output, and value-added processing.

The total cost for the SPB Prevention Project in 2008 and 2009 was $1,243,663 (table 1). The economic impact of the SPB project on the Mississippi forest economy was $14,383,482 in employment, wages, value added processing, and sector output for each impact of direct, indirect, and induced effects (table 2). The program had a benefit/cost ratio of 11.6, which was very favorable. Overall, the SPB Prevention Project has been highly beneficial to Mississippi's forest health as well as its forest economy.

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LITERATURE CITED

Coulson, R.N.; Klepzig, K.D., eds. 2011. Southern pine beetle II. Gen. Tech. Rep. SRS-140. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 512 p.

Londo, A.J.; Kushla, J.D.; Smallidge, P. 2008. Use of county tax rolls for the creation of mailing lists for extension programming. Article #6FEA6. Journal of Extension. 46(6). http://www.joe.org/joe/2008december/a6.php. [Date accessed: January 7, 2009].

Minnesota IMPLAN Group, Inc. 2010. IMPLAN v 3.0 user's guide. Stillwater, MN: Minnesota IMPLAN Group Inc. 152 p.

PHYSIOLOGICAL DECLINE AND RECOVERY OF EASTERN HEMLOCK TO HEMLOCK WOOLLY ADELGID

Kelly McDonald, John Seiler, Scott Salom, and Rusty Rhea[1]

Eastern hemlock [*Tsuga canadensis* (L.) Carr.] is a foundation species that occupies a unique niche in forest ecosystems and which often forms pure stands throughout the eastern United States. Throughout the last half century, widespread mortality of *T. canadensis* has occurred with the introduction of an invasive pest, the hemlock woolly adelgid (*Adelges tsugae* Annand) (HWA). HWA now threatens to destroy millions of ha of hemlock-dominated forests and to disrupt the associated ecosystems (Salom and others 2008). By quantifying physiological responses of *T. canadensis* to HWA, better insights can be made into control methods for HWA to slow and potentially stop its spread, thus protecting remaining hemlock stands.

In order to determine how HWA impacts hemlock physiology, threes sites with varying degrees of infestation were chosen, and half of the trees at each site were treated with imidacloprid (Merit® 2 F, Bayer, Kansas City, MO) while the rest were left untreated. Trees were assessed monthly over the course of a complete growing season using a LI-COR 6400 portable open path gas exchange system (LI-COR Inc, Lincoln NE) to determine rate of photosynthesis, stomatal conductance, and other physiological parameters. Chlorophyll fluorescence and bud break were also characterized for all trees at each site. At the end of one complete growing season, statistical comparisons were made between treated and untreated trees. Tree size and available environmental factors were tested as covariates. Models for gas exchange were also developed using environmental variables and a stepwise regression procedure.

Over the course of the measurement period, photosynthetic rates were generally similar across treated and untreated trees, as well as across sites (fig. 1). More often, treated trees had higher rates of photosynthesis, particularly at the Fishburn and Twin Falls sites. There were six sampling dates with significant differences, and treated trees had higher photosynthetic rates on 5 out of the 6 dates. In a regression analysis in which photosynthesis was modeled using environmental parameters, there was a significant difference between the intercepts of treated (3.4146) and untreated (3.2509) trees, suggesting that throughout the measurement period, treated trees had slightly higher photosynthetic rates compared to untreated trees. Chlorophyll fluorescence was also similar between treated and untreated trees, suggesting no difference in photosynthetic capacity. Bud break did differ significantly between treatments, with higher bud break for treated trees after a year of treatment (table 1). These results suggest that HWA is causing tree mortality largely through a reduction of leaf area and not a reduction in photosynthetic capacity.

LITERATURE CITED

Salom, S.M.; Kok, L.T.; Lamb, C. [and others]. 2008. Biological control of hemlock woolly adelgid: what is it going to take to make it work? In: Onken, B; Reardon, R., comps. Fourth symposium on hemlock woolly adelgid in the eastern United States. FHTET-2001-1. Washington, DC: U.S. Department of Agriculture Forest Service, Forest Health Technology Team. http://na.fs.fed.us/fhp/hwa/pubs/proceedings/2008_proceedings/fhtet_2008.pdf. [Date accessed: March 1, 2013].

[1]Graduate Student and Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061; Professor, Virginia Polytechnic Institute and State University, Department of Entomology, Blacksburg, VA 24061; and Entomologist, USDA Forest Service, Southern Research Station, Asheville, NC 28804.

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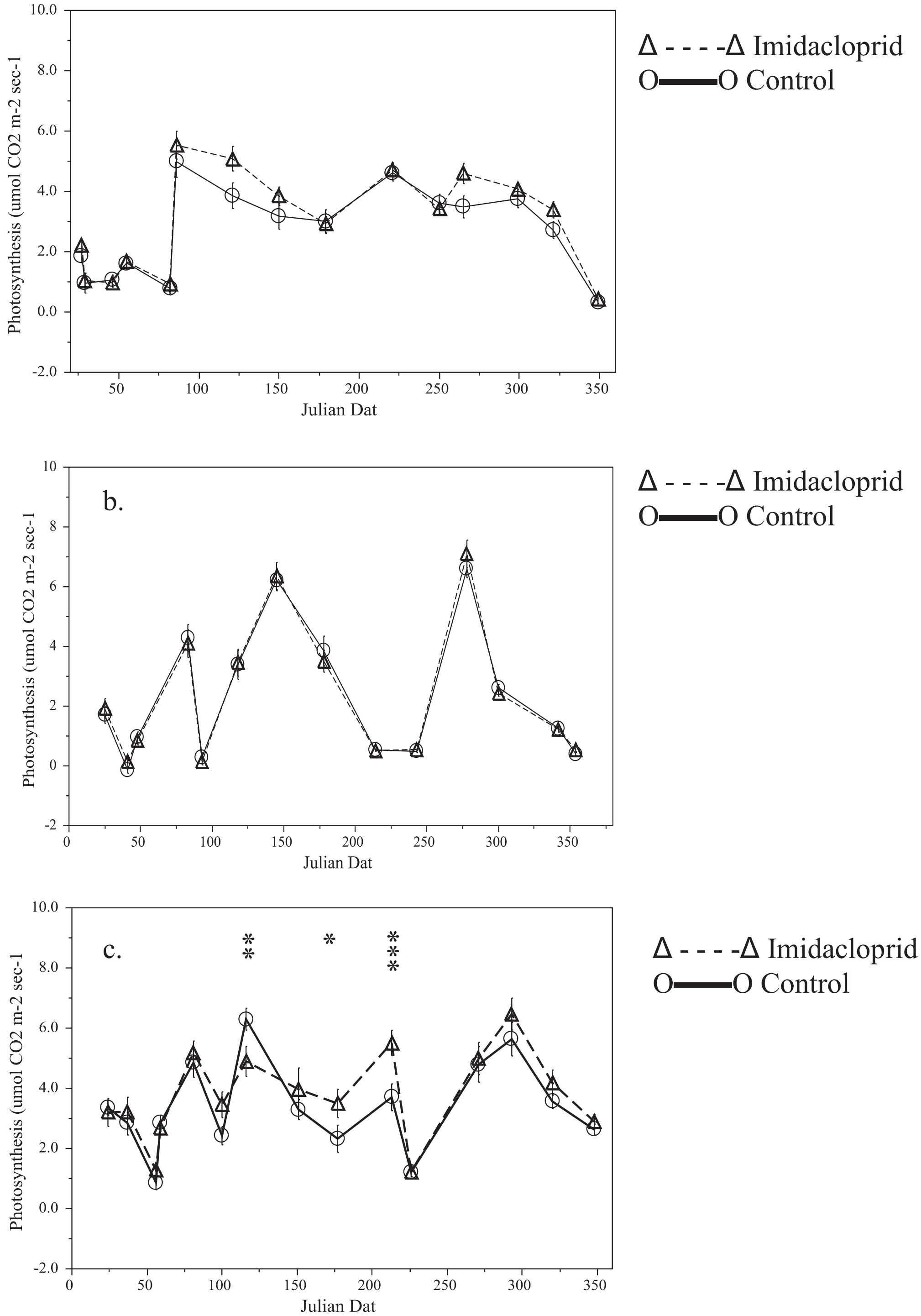

Figure 1--Eastern hemlock photosynthesis rates comparison between imidacloprid treated and untreated trees at (A) Fishburn, (b) Mountain Lake, and (c) Twin Falls from 2012 to 2013. Points show mean ± SE. Single asterisk denotes difference at α = 0.10, double asterisk denotes a significant difference at α = 0.05, and a triple asterisk denotes a difference at α = 0.01.

Table 1—Hemlock bud break at Fishburn and Mountain Lake in 2012 and 2013

	----------Fishburn----------		------Mountain Lake------	
Year	Treated	Untreated	Treated	Untreated
2012	13.4	5.78	46.9	46.6
2013	88.6	22.5	58.9	24.5

SOURCES OF AUTUMN OLIVE INVASION FOLLOWING SILVICULTURAL TREATMENTS

Matthew R. Moore and David S. Buckley[1]

Exotic, invasive plants are a significant concern for forest managers and conservation-minded private landowners. Autumn olive (*Elaeagnus umbellate* Thunb.), originally introduced for wildlife forage and for the stabilization of mine soils in the 1920s (Fowler and Fowler 1987), has become ubiquitous along many roads. Autumn olive is shade tolerant (Yates and others 2004), salt tolerant, drought tolerant and thrives across a wide range of soil pH (Dirr 1998). Of particular concern is the potential for this shade-tolerant exotic, invasive plant to spread from roadside areas into the understories of adjacent forests and displace native flora.

Results of previous research have linked soil disturbance (Bergelson and others 1993), light levels (Winter and others 1982), and the frequency of human traffic (Lundgren and others 2004) with successful exotic plant invasions. All three of these factors are often more abundant along roadways, which tend to have greater exotic, invasive plant presence than other areas (Trombulak and Frissell 2000). In the present study, relationships between several factors and the success of autumn olive were examined along forest-road edges. Specific objectives were: (1) to investigate the effect of forest-road edge aspect on autumn olive patch depth, abundance, and height; (2) to determine the relationships between the height and abundance of autumn olive and those of woody native plants and other woody, exotic, invasive plants; and (3) to record slope, elevation, canopy cover, and basal area at points of invasion to investigate factors potentially related to patch depth, abundance, and height of autumn olive.

To achieve these objectives, autumn olive patch depth and the abundance of autumn olive in five height class categories (0-0.50 m, 0.51-1.00 m, 1.01-2.00 m, 2.01-3.00 m, and > 3.00 m) were measured in 5- by 10-m rectangular sample plots. Indices of average autumn olive height were calculated using the midpoints for each height class. Plots were established on forest-road edges having aspects within ± 10° of cardinal north, east, south, and west, and the short axis of each plot was centered on the boles of the trees along the forest edge. Fifteen plots were established for each aspect. Slope toward the road, slope toward the forest interior, elevation, canopy cover, and basal area were measured at plot center, and the numbers of native, woody species and exotic, invasive, woody species in each of the size classes developed for autumn olive were tallied. Differences between forest-road edge aspects were analyzed with an analysis of variance (ANOVA). Relationships between autumn olive and other plant species were evaluated using linear regression. The range of values for each site factor measured was subdivided into logical categories and analyzed with ANOVA to determine effects of these variables on autumn olive success.

Forest-road edges with northern, southern, and western aspects had the greatest autumn olive patch depths ($p = 0.0273$). Density of 2.00- to 3.01-m-tall autumn olive was greatest on south-facing forest-road edges and least on north-facing forest-road edges ($p = 0.0324$). All other measures of autumn olive success did not vary with aspect. Linear regression indicated that there was a positive relationship ($y = 0.54061 + 0.28205 * x$) between the average height of autumn olive and the average height of native tree species along south-facing forest-road edges (table 1). In addition, east-facing forest-road edges exhibited an increased density ($y = -46.79446 + 66.06644 * x$) of all species of woody, exotic, invasive plants with greater average autumn olive heights. On north-facing

[1]Graduate Teaching Assistant and Professor, respectively, University of Tennessee Institute of Agriculture, Department of Forestry, Wildlife and Fisheries, Knoxville, TN 37996.

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Table 1--Linear regression relationships between the density and height of autumn olive and the density and height of woody, native plants and other woody, exotic, invasive plants across all aspects and by individual aspect

Variables	All aspects	North	East	South	West
Average height of native trees[a]	ρ = 0.0498 R^2 = 0.0647	ρ = 0.6602 R^2=0.0153	ρ = 0.7240 R^2=0.0099	ρ = 0.0200 R^2=0.3509	ρ = 0.9596 R^2=0.0002
Density of all exotic, invasive plant species[a]	ρ = 0.8537 R^2 = 0.0006	ρ = 0.8946 R^2=0.0014	ρ = 0.0556 R^2=0.2537	ρ = 0.3796 R^2=0.0598	ρ = 0.4120 R^2=0.0524
Avg. height of exotic plants (all species)[b]	ρ = 0.1472 R^2=0.0359	ρ = 0.0271 R^2=0.3229	ρ = 0.3643 R^2=0.0637	ρ = 0.9231 R^2=0.0007	ρ = 0.6883 R^2=0.0128
Density of all exotic, invasive plant species[b]	ρ = 0.7596 R^2=0.0016	ρ = 0.2756 R^2=0.0906	ρ = 0.5416 R^2=0.0293	ρ = 0.0128 R^2=0.3902	ρ = 0.4688 R^2=0.0411

[a]Independent variables related to the average height of autumn olive.
[b]Independent variables related to the density of all height classes of autumn olive combined.

forest-road edge aspects, the average height of all other woody, exotic, invasive plants was positively related (y = 0.51011 + 0.00516 * x) to the density of all size classes of autumn olive combined. The density of autumn olive in all height classes combined was positively related (y = 25.42396 + 1.03242 * x) to the density of all other woody, exotic, invasive species in all height classes combined. When data for site factors were subdivided into categories, average height of autumn olive differed (fig. 1) with changes in slope toward the road (p = 0.0100) and slope toward the interior forest (p = 0.0188). Density of 0.51- to 1.00-m autumn olive differed with changes in slope toward the road (p = 0.0494), and 2.01- to 3.00-m autumn olive differed with changes in elevation (p = 0.0324).

These results suggest that edge aspect, slope characteristics, and elevation may be important in influencing autumn olive success. The positive relationships between the heights and densities of autumn olive and those of native woody plants and other invasive woody plants suggest that conditions that favor autumn olive may also favor other native and exotic species. Knowing which forest edges are particularly susceptible to shade-tolerant, exotic, invasive plants will increase the efficiency of managing existing populations and detecting new invasions. The factors identified in this study that appear to be linked to autumn olive success may prove useful in the development of GIS-based predictions of when and where autumn olive is likely to become a problem. If successful, this approach could be used to produce more comprehensive, multi-species prediction tools for managers.

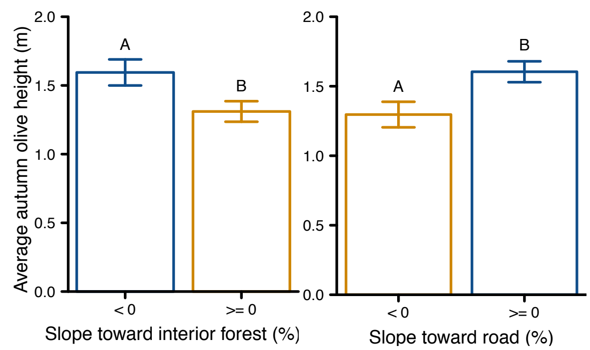

Figure 1--Average autumn olive height by upward or downward slope from plot center. Blue bars for slope toward the interior forest represent a downward slope from plot center and blue bars for slope toward road represent a downward slope from the road toward plot center. Groupings based on Tukey's method with α = 0.05. Error bars represent one standard error.

LITERATURE CITED

Bergelson, J.; Newman, J.A.; Floresroux, E.M. 1993. Rates of weed spread in spatially heterogeneous environments. Ecology. 74(4): 999-1011.

Dirr, M.A. 1998. Manual of woody landscape plants: their identification, ornamental characteristics, culture, propagation and uses. Champaign, IL: Stipes Publishing, LLC. 1187 p.

Fowler, L.J; Fowler, D.K. 1987. Stratification and temperature requirements for germination of autumn olive (*Elaeagnus umbellata*) seed. Tree Planter's Notes. Winter: 14-17.

Lundgren, M.R.; Small, C.J.; Dreyer, G.D. 2004. Influence of land use and site characteristics on invasive plant abundance in the Quinebaug Highlands of southern New England. Northeastern Naturalist. 11(3): 313-332.

Trombulak, S.C.; Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14(1): 18-30.

Winter, K.; Schmitt, M.R.; Edwards, G.E. 1982. *Microstegium vimineum*, a shade adapted C_4 grass. Plant Science Letters. 24: 311-318.

Yates, E.D.; Levia, D.F., Jr.; Williams, C.L. 2004. Recruitment of three non-native invasive plants into a fragmented forest in southern Illinois. Forest Ecology and Management. 190: 119-130.

DEVELOPMENT OF INTEGRATED MANAGEMENT PRACTICES FOR THE CONTROL OF CHINESE TALLOW ON PARRIS ISLAND MARINE CORPS RECRUIT DEPOT

Lauren S. Pile, G. Geoff Wang, and Patricia A. Layton[1]

Chinese tallow [*Triadica sebifera* (L.) Small] is an aggressive, fast-growing, highly adaptable invasive tree of the southeastern United States coastal region. Since its introduction in the early 1800s, Chinese tallow has become a serious threat to native grassland and forest communities from mid-coastal North Carolina to northern Florida and west to central Texas. Our study is located on Parris Island Marine Corps Recruit Depot (MCRD) in Beaufort County, SC. Parris Island MCRD consists of 3257 ha, with vegetation dominated by mixed maritime forest, pine forest (natural and plantation), and saltwater marsh.

Chinese tallow has been managed on Parris Island MCRD since 2001 through the use of herbicides, primarily with 'hack and squirt' methodology. In 2010, we surveyed for invasive species presence and abundance on Parris Island MCRD in order to monitor the Chinese tallow population and to assess the effectiveness of previous control efforts. Results from this survey suggested that there is the need for a more effective management approach because the Chinese tallow population in some areas had increased despite herbicide applications. Our study was designed to develop an effective approach for managing Chinese tallow while restoring the native forest ecosystem impacted by the invasion of Chinese tallow. Specifically, we tested four integrated treatments series including mechanical, herbicide and fire (MHF), mechanical and herbicide (MH), herbicide and fire (HF), and herbicide (H) only to determine their efficacy on Chinese tallow control as well as their potential adverse effects on native vegetation.

We implemented a randomized complete block design with four treatments and eight replicates. Treatment areas were ≥ 0.4 ha, with a sample area of 0.08 ha randomly nested within each treatment area to monitor treatment response of Chinese tallow as well as the resident plant community. In 2012, we quantified woody plant abundance by measuring the diameter at breast height (d.b.h.) for each individual in the sampling plots of each experimental unit, and we used analysis of variance to determine any pre-treatment differences in mean basal area, d.b.h., and stems/ha by species. Pre-treatment results indicate that Chinese tallow is the second-most dominant tree species by basal area and has the second highest density across the 32 experimental units at (table 1). There were no significant differences in tree and shrub species composition among treatments (significance level of $p = 0.05$).

The application of the mechanical mulching treatment (MHF and MH) will occur in the spring of 2013 when total non-structural carbohydrates are at their lowest levels in roots and are being actively transported to aboveground tissues that are associated with the metabolic costs of break in dormancy, bud break, and leaf development (Conway and others 1999). Mulching may also provide a dampening effect on diurnal soil temperature fluctuation and amplitude, resulting in reduced seed germination, even though the large size of Chinese tallow seeds may provide the nutritional resources for germination and emergence in deep mulch (Donahue and others 2004). The application of herbicide (all treatments) during fall 2013 will occur when total non-structural carbohydrates actively translocate downwards to the roots, which will be more effective for herbicide assimilation into perennating buds and organs (Conway and others 1999). The addition of a fire treatment (MHF and HF) may also negatively impact Chinese tallow forest dominance and stimulate native species abundance and diversity.

[1]Graduate Research Assistant and Professors, respectively, Clemson University, Department of Forestry and Natural Resources, Clemson, SC 29634.

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Table 1--Summary of basal area (m^2/ha), diameter at breast height (cm) and density (stems/ha) for all tree and shrub species measured in the 32 (0.08 ha) plots

Scientific Name	Common name[a]	Basal area	DBH	Density
		m^2/ha	*cm*	*stems/ha*
Pinus elliottii	slash pine*	18.93950	7.87	3891.02
Triadica sebifera	Chinese tallow*	3.04471	4.54	1883.98
Morella cerifera	wax myrtle	2.01761	2.97	2906.25
Ilex vomitoria	yaupon	1.87733	2.78	3086.33
Liquidambar styraciflua	sweetgum*	0.51409	4.53	319.53
Sabal palmetto	cabbage palmetto	0.46441	31.77	5.86
Fraxinus pennsylvanica	green ash*	0.28090	28.85	4.30
Quercus virginiana	live oak*	0.28029	10.89	30.08
Quercus nigra	water oak*	0.19744	7.20	48.44
Quercus hemisphaerica	Darlington oak*	0.11749	7.74	25.00
Nyssa sylvatica	blackgum*	0.04760	27.85	0.78
Acer rubrum	red maple*	0.03492	15.09	1.95
Juniperus virginiana var. silicicola	southern redcedar*	0.03423	6.20	11.33
Prunus serotina	black cherry*	0.03320	5.27	15.23
Celtis laevigata	sugarberry*	0.03234	16.23	1.56
Quercus pagoda	cherrybark oak*	0.02996	4.10	22.66
Carya ovalis	red hickory*	0.02689	29.60	0.39
Baccharis halimifolia	eastern baccharis	0.02674	2.52	53.52
Callicarpa americana	American beautyberry	0.02623	1.19	235.16
Melia azedarach	chinaberry*	0.01428	8.81	2.34
Rhus copallinum	winged sumac	0.00317	1.61	15.63
Ailanthus altissima	tree-of-heaven*	0.00291	5.62	1.17
Morus rubra	red mulberry*	0.00070	4.77	0.39
Diospyros virginiana	common persimmon*	0.00053	4.14	0.39
Lantana camara	lantana	0.00024	0.89	3.91

[a]An asterisk denotes tree life form.

Chinese tallow is a thin-barked species when young, which may result in increased mortality from fire. Prescribed burning has also been shown to reduce germination probability of Chinese tallow (Burns and others 2004). However, rapid leaf litter decomposition by this species may suppress the ability to carry a surface fire without a previous treatment such as mulching

We expect that the MHF treatment will have the greatest overall effect on the Chinese tallow population within the treatment areas. The application of the mechanical treatment in the spring will reduce carbohydrate stocks in the roots and will be followed by foliar herbicide treatment in the fall of the same year to target advanced regeneration and newly established seedlings. The prescribed fire treatment will be carried by the fuels provided by the mulching treatment 2 years prior, which will also help to reduce vigor of any additional regeneration. The use of fire is also intended to re-establish the historical fire regime, to prevent the future invasion of Chinese tallow and to promote native plant diversity in herbaceous layer.

LITERATURE CITED

Burns, J.H.; Miller, T.E. 2004. Invasion of Chinese tallow *(Sapium sebiferum)* in the Lake Jackson area, northern Florida. American Midland Naturalist. 152: 410-417.

Conway, W.C.; Smith, L.M.; Sosebee, R.E.; Bergan, J.F. 1999. Total nonstructural carbohydrate trends in Chinese tallow roots. Journal of Range Management. 52: 539-542.

Donahue, C.; Rogers, W.E.; Siemann, E. 2004. Effects of temperature and mulch depth on Chinese tallow tree *(Sapium sebiferum)* seed germination. Texas Journal of Science. 56: 347-356.

MEADOW VOLE-INDUCED MORTALITY OF OAK SEEDLINGS IN A FORMER AGRICULTURAL FIELD PLANTING

Andrew B. Self, Andrew W. Ezell, Dennis Rowe, Emily B. Schultz, and John D. Hodges[1]

Abstract--Seedling mortality due to meadow vole herbivory is an often acknowledged but relatively unstudied aspect of hardwood afforestation. Vole-induced mortality is not typically a major item of concern in afforestation attempts. However, damage has been extreme in some plantings. A total of 4,320 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 three Mississippi sites. All sites were of comparable soils and received above average precipitation throughout the 3-year duration of the study. One half of seedling plots were treated with a 1-year post-plant application of Oust XP®. The other half was treated for 2 years. May 2008 seedling survival was excellent at 99.7 percent. However, 76.9 percent of observed seedling mortality was directly attributable to vole damage. End-of-growing-season survival for the first three growing seasons was 98.9 percent, 97.1 percent, and 94.1 percent, respectively. While these survival rates would be considered excellent in most afforestation attempts, vole herbivory accounted for 57.6 percent of observed seedling mortality over the course of the study. If the assumption is made that vole-induced mortality had been nonexistent, third-year survival would have been approximately 97.5 percent. Fourth and fifth growing season mortality accounted for an additional 7.8 percent reduction in overall survival at one site. If vole-related seedling mortality were nonexistent, fifth-year survival would have been approximately 98.9 percent. Seedling mortality levels of this magnitude are significant and may deserve consideration in planting efforts.

INTRODUCTION

A problem that occurs infrequently in plantings across the eastern United States is seedling mortality due to the meadow vole, also known as the pine vole (*Microtus pinetorum* LeConte) (Ostfeld and Canham, 1993). Meadow voles are semifossorial, arvicoline rodents found in woodlands and other habitats across the eastern United States. Meadow vole herbivory is usually subterranean in nature, resulting in seedling taproots eaten below the root collar (Schreiber and Swihart 2009). Several studies have indicated 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. 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 the protection from predation provided by the greater levels of herbaceous vegetation typically found on these sites (Buell and others 1971, Gill and Marks 1991, Ostfeld and Canham 1993). While typically not a major concern in afforestation attempts, vole herbivory can reach levels with substantial impact to planting success.

OBJECTIVE

The objective of this study was to determine the effect of mechanical site preparation, species, and herbaceous weed control on seedling mortality attributable to meadow vole herbivory.

MATERIALS AND METHODS

Site Description

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

[1]Assistant Extension Professor, Mississippi State University, Department of Forestry, Grenada, MS 38901; Professor, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; Professor, Mississippi State University, Experimental Statistics Unit, Mississippi State, MS 39762; Professor, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; and Professor Emeritus, Ashland, MS, 39576.

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Malmaison WMA

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

At initiation of this project, dominant onsite herbaceous species were ryegrass (*Lolium* spp.), bermudagrass (*Cynodon dactylon* L.), Brazilian vervain (*Verbena brasiliensis* Vell.), and Carolina horsenettle *(Solanum carolinense* L). Forty other herbaceous species occurred in small quantities. Cumulative herbaceous coverage by all species was 100 percent.

Copiah County WMA

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

At study initiation, dominant onsite herbaceous species were crimson clover (*Trifolium incarnatum* L.), ladino clover (*Trifolium repens* L.), and ryegrass (*Lolium multiflorum* Lam.). Twenty-five other herbaceous species occurred in small quantities. Cumulative herbaceous coverage by all species was 100 percent.

Arkabutla Lake

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

Dominant herbaceous species on site at study initiation were Brazil vervain, poorjoe (*Diodia teres* Walt.), and thorny amaranth (*Amaranthus spinosus* L.). Twenty-one other herbaceous species were observed in small quantities across the site. Cumulative herbaceous coverage of all species was approximately 5 percent.

Experimental Design

The study was completely replicated at all three sites. Each site had its own unique installment of randomized treatment combinations. 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 pre-emergent competition control. The experimental unit was a plot with its unique combination of site preparation treatment, species, and pre-emergent chemical treatment. The response variable was seedling mortality directly resulting from vole herbivory.

There were three blocks at each site that contained all possible site preparation treatment/species/pre-emergent chemical treatment combinations. 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. Each plot received different pre-emergent chemical treatments.

Mechanical Site Preparation and Herbaceous Weed Control (HWC) Treatments

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 1-year application and a 2-year application of Oust XP®. Both treatments were applied in 5-foot-wide bands using a rate of 2 ounces of product per acre and were applied over the top of seedlings prior to budbreak. The 1-year Oust XP® application was applied during March 2008. The 2-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 (GPA).

Seedling Establishment

Nuttall oak, Shumard oak, and swamp chestnut oak were chosen for use in this study. Seedlings were purchased from Joshua Timberlands Elberta Nursery in Elberta, AL and were lifted mid-January 2008. 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).

A total of 4,320 seedlings were planted. Across the three sites, 1,440 seedlings of each species were planted. At each site, 480 seedlings of each species were planted. Seedlings were planted at root collar depth during February 2008 by university personnel using a10-foot spacing.

Survival Measurements

Seedling survival and cause of death were determined by ocular evaluation and recorded October 2008, October 2009, and October 2010 for all sites. Seedling bases were examined to determine if vole herbivory was the causal agent for mortality. If a seedling were observed as a resprout in later observations, it was back-modified. Seedlings dying from vole herbivory were recorded as such. For logistical reasons, the Malmaison WMA and Copiah County WMA sites were not resampled in years 4 and 5. Fourth and fifth-year survival was recorded using the same measuring procedures for the Arkabutla Lake site during October 2011 and January 2013.

Data Analysis

All statistical analyses were performed using Statistical Analysis System version 9.2 (Cary, NC). 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. Repeated Measures Analysis of Variance was applied on transformed survival data using Proc Mixed and Proc Glimmix. Best covariance structure for repeated measures compared CS, AR(1), and TOEP(2) alternatives using BIC criteria. The best covariance structure was found to be TOEP(2). Full model was fit; however, no differences were detected among species, mechanical site preparation, HWC treatments, or their interactions for stems killed by voles. As such, overall survival averages are used in this paper. While transformed survival data were used for analyses, untransformed means are presented for interpretation.

RESULTS AND DISCUSSION

Overall First, Second, and Third-Year Survival

Overall seedling survival was excellent during the May 2008 evaluations (99.67 percent over all combinations) ranging from 97.78 to 100.0 percent for individual treatment combinations (table 1). Survival levels this high indicate that there was little mortality due to planting shock. Most observed initial seedling mortality resulted from vole herbivory. Of the 13 seedlings that died by the May 2008 observations, 10 died from clipping. Vole-induced mortality accounted for 76.92 percent of the total mortality for initial evaluations. Assuming that vole-killed seedlings had lived and not died from some other cause, overall survival for the May 2008 observations would have been approximately 99.9 percent.

Table 1—Yearly survival of oak seedlings on all sites by treatment

		Species					
		Nuttall oak		Shumard oak		Swamp chestnut oak	
Mechanical	Timing	1 year OustXP	2 year OustXP	1 year OustXP	2 year OustXP	1 year OustXP	2 year OustXP
Control	5-2008	99.44	99.44	100.00	98.89	99.44	99.44
	10-2008	98.33	99.44	98.89	98.33	99.44	98.33
	10-2009	96.67	99.44	98.89	93.33	98.89	95.00
	10-2010	96.11	88.89	97.22	88.89	96.67	94.44
Subsoiled	5-2008	99.44	100.00	100.00	100.00	100.00	99.44
	10-2008	99.44	99.16	97.22	98.33	99.44	97.22
	10-2009	98.89	99.16	93.89	96.67	98.89	92.22
	10-2010	98.33	93.04	92.22	93.83	98.27	91.11
Bedded	5-2008	97.78	100.00	100.00	100.00	100.00	100.00
	10-2008	97.78	98.89	99.44	99.44	100.00	99.44
	10-2009	97.22	95.56	99.44	97.22	97.78	97.22
	10-2010	96.11	92.78	98.30	93.33	95.00	95.56
Combination plowed	5-2008	99.44	100.00	100.00	100.00	100.00	99.38
	10-2008	99.44	100.00	98.33	99.44	99.44	98.27
	10-2009	98.33	96.67	96.11	99.44	98.89	95.49
	10-2010	88.01	95.56	93.89	89.44	98.33	93.14

Overall, October 2008 seedling survival was 98.9 percent (table 2) and ranged between 97.22 and 100.0 percent for individual treatment combinations (table 1). Meadow vole herbivory continued to be the primary source of seedling mortality throughout the 2008 growing season. Meadow vole herbivory accounted for the death of an additional 25 seedlings during the growing season. By the time October 2008 evaluations were performed, 35 of 45 seedlings that died throughout the growing season were killed by meadow vole herbivory. Vole-induced mortality accounted for 77.78 percent of the total mortality for 2008 evaluations. Assuming that vole-killed seedlings had survived and not died from some other cause, overall survival for the October 2008 observations would have been approximately 99.7 percent.

The greatest change in seedling survival between May and October 2008 was observed in Shumard oak seedlings planted in subsoiled areas treated with a 1-year Oust XP® application (table 1). May 2008 survival was 100.0 percent for this combination, and by October, survival had dropped to 97.22 percent. This 2.78 percent difference was heavily influenced by meadow vole-induced seedling mortality. Of the five trees that died by the October 2008 evaluations, four were the result of vole herbivory. Assuming that vole-killed seedlings had not died and did not die from some other cause, overall survival for this combination would have been approximately 99.44 percent.

Table 2--Overall survival of oak seedlings on all sites by year

Year	Survival
	percent
2008	98.90
2009	97.14
2010	94.11

Overall, survival for 2009 was 97.1 percent (table 2). This 1.8 percent drop in survival between October 2008 and October 2009 was driven primarily by leaf scorch in swamp chestnut oak seedlings. Swamp chestnut oak is classed as shade intolerant (Burns and Honkala 1990) but scorches relatively easily when fully exposed to full sunlight such as that encountered in HWC treated areas. Vole herbivory was almost non-existent for the year.

Overall seedling survival was 94.1 percent ranging between 88.01 and 98.33 percent for individual treatment combinations in 2010 (table 2). Lower survival was observed for all combinations in October 2010 evaluations. While most combinations did not exhibit large differences between the October 2009 and October 2010 growing seasons, some of the differences were substantial. Meadow vole herbivory was the largest contributor to seedling mortality between the 2009 and 2010 growing seasons. Of the 129 seedlings that died between October 2009 and October 2010, 97 seedlings (75.19 percent) were killed by voles. Assuming that seedlings killed by voles had not died from some other cause, overall survival for the October 2010 observations would have been approximately 96.35 percent.

Meadow vole damage was the main factor in lower survival noted in the four combinations with the lowest survival at the end of the 2010 growing season (table 1). A drop in survival of 10.55 percent between 2009 and 2010 was observed for Nuttall oak in subsoiled areas treated with 2-year Oust XP® applications. Vole herbivory accounted for 13 (68.42 percent) of the 19 seedlings killed during 2010 for that combination. Similar survival reductions were noted in Nuttall oak planted in combination plowed areas receiving Oust XP® for 1 year, Shumard oak planted in combination plowed areas that received the 2-year Oust XP® treatment, and Nuttall oak planted in subsoiled areas treated with 2 years of Oust XP® (10.32 percent, 10.0 percent, and 6.12 percent reductions, respectively). Vole herbivory accounted for a large portion of this mortality (10, 15, and 8 seedlings, respectively). This accounted for a respective 52.63, 83.33, and 72.73 percent of the mortality observed in these treatment combinations.

Of the 255 seedlings that died over the course of the 3-year span of this study, vole herbivory accounted for 147 seedlings (57.64 percent). If the assumption is made that vole-induced mortality had been nonexistent, overall third year survival would have been approximately 97.50 percent. A 3.40 percent reduction in survival typically does not result in a change in management strategy. Seedlings should be of a size by year 3 or 4 that would serve to inhibit substantial levels of continued herbivory in the future. However, more severe vole herbivory has been noted in some studies. Rathfon and others (2008) found that vole herbivory resulted in mortality levels as high as 19 percent in southern Indiana for three species of oak seedlings under mature, closed-canopy, oak-dominated forests. The highest levels of meadow vole-induced seedling mortality were observed in areas where midstory removals had been performed. The authors surmised that increased mortality in these areas was due to the added protection from predation that increased herbaceous coverage provided.

Although statistical differences were not detected, vole herbivory was more prevalent in areas undergoing 2 years of Oust XP® treatment (table 1). Slightly lower overall seedling survival levels were observed starting with first growing season evaluations. At this point, all plots had received identical HWC treatment, and seedlings were not exhibiting any phytotoxic effects. Second year difference among treatments were similar with the gap between HWC treatments opening slightly in swamp chestnut oak for reasons previously discussed. Third year survival gaps increased among HWC treatments in all three oak species. While not significantly different, vole herbivory in areas treated with 2 years of Oust XP® treatment was noticeably greater in year 3. The likely cause for increased vole herbivory in these areas can be found in the form of increased herbaceous vegetation coverage compared to areas receiving only 1 year of HWC treatment. In the third growing season, areas treated with 1 year of Oust XP® were in their second year of herbaceous vegetation growth. Herbaceous vegetation was at a more mature growth stage compared with the vegetation observed in areas treated with 2 years of HWC. The 1-year-old vegetation in 2-year HWC areas was denser and provided better meadow vole cover. Other studies have noted that afforestation attempts on retired agriculture fields might be hindered due to protection from predation resulting from the greater herbaceous vegetation levels inherent to these sites (Buell and others 1971, Gill and Marks 1991, Ostfeld and Canham 1993).

Arkabutla Lake Fourth and Fifth-Year Survival

For logistical reasons, seedling survival of only the Arkabutla Lake site was resampled in years 4 and 5. Overall, 2010 seedling survival was 95.6 percent (table 3). At this point, meadow vole herbivory was responsible for 75 percent of overall seedling mortality on this site. Assuming that vole-killed seedlings had survived and not died from some other cause, overall survival for 2010 observations would have been approximately 99.7 percent.

Table 3--Overall third-, fourth-, and fifth-year survival of oak seedlings on the Arkabutla Lake site

Year	Survival
	percent
2008	95.6
2009	91.7
2010	87.8

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 more traditional trend at the Arkabutla Lake site. Overall, 2011 seedling survival was 91.7 percent (table 3). Meadow vole herbivory was responsible for 100 percent of seedling mortality observed between the 2010 and 2011 observations (3.9 percent reduction). Overall, 2012 seedling survival was 87.8 percent with vole herbivory accounting for 100 percent of seedling mortality observed between the 2011 and 2012 observations (3.9 percent reduction).

Of the 176 seedlings that died over the 5-year course of the study at the Arkabutla Lake site, vole herbivory accounted for 159 seedlings (90.3 percent). If the assumption is made that vole-induced mortality had been nonexistent, overall fifth- year survival would have been approximately 98.8 percent. Vole herbivory was directly responsible for an 11 percent reduction in oak survival on this site. A possible explanation for the increasing vole damage over time may be the amount of vegetative coverage within the plantation.

CONCLUSIONS

Meadow vole-induced mortality of planted seedlings is a known but ordinarily minor factor in afforestation attempts. We found vole herbivory can reach levels with substantial impact to planting success. Additionally, vole herbivory is typically thought to be nonexistent after the second or third growing season. Oak mortality from vole damage reached levels of concern for this study both in frequency and seedling age. Most land managers evaluate planting success during the first 3 years after plantation establishment. In some situations it is possible that the greatest levels of vole damage do not manifest until after this point. It is possible that vole herbivory plays a larger role in oak survival rates in plantation settings than is currently assumed.

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: Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture. 877 p. Vol. 2.

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]. 2011. Precipitation totals. Washington, DC: U.S. Department of Commerce National Climatic Data Center web. http://www.noaa.gov. [Date accessed: February 18, 2011].

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: 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.

TREE-RING RECORD OF DROUGHTS AND SEVERE WINTER STORMS IN THE OUACHITA MOUNTAINS SINCE 1745

Douglas J. Stevenson, Thomas B. Lynch, and James M. Guldin[1]

Severe winter storms cause serious damage to trees, timber, power lines, and transportation systems each year. In the Ouachita Mountains, historical records of these storms extend back only 117 years, and many of them are of low-quality or have missing data. Dendrochronology helps fill in and correct the historical record by providing a severe winter storm signal in *Pinus echinata* Mill. that extends back to 1745. Drought may be associated with the occurrence of severe winter storms, creating similarities in tree-ring patterns. Using the Palmer Drought Severity Index (PDSI) to de-trend tree ring data may remove the severe storm signal along with the drought signal. The winter storm signal is consistent with injury to the tree by trunk breakage, branch loss, and bending. Except for storm year, trees that break have wider growth rings than those that don't, suggesting greater exposure to ice accumulation. Missing rings on high-quality sites occur only in years with severe storms. The winter storm signal is:

$$R_i = (Y_i + Y_{i+1}) / (Y_{i+2} + Y_{i+3}) \quad (1)$$

where: R_i is the ratio of ring width in the 2 years after the storm to that of the succeeding 2 years, and Y_i is the ring width of the year of the storm.

Values of R_i usually run between 0.700 and 0.900. The proportion of R_i values that exceed an index value, usually between 0.100 and 0.300, determines whether Year *i* had a severe storm. The Index value is chosen to maximize the number of correct predictions of historical storms. Major winter storms occurred once in 16 to 20 years; two out of three known ice storm years produced trunk breakage, giving a probability of 0.042 that an ice storm will cause damage to Ouachita shortleaf pines in any given year. Tree rings permit estimation of annual precipitation and low temperatures and identification of years with severe winter storms. Future research might make it possible to distinguish between wind storms, ice storms, and other severe storms and estimate precipitation by season. Tree-ring chronologies are a powerful tool for weather and climate studies at a finer scale than is possible with any other method or proxy.

[1]Senior Research Specialist and Professor, respectively, Oklahoma State University, Department of Natural Resource Ecology and Management, Stillwater, OK 74078; and Supervisory Ecologist, USDA Forest Service, Southern Research Station, Hot Springs, AR 71902.

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ALLOMETRIC BIOMASS MODELS AND APPLICATIONS FOR BRANCHES OF CHINESE TALLOW IN A MISSISSIPPI BOTTOMLAND FOREST

Allison M. Stoklosa, Nana Tian, and Zhaofei Fan[1]

Abstract--Chinese tallow [*Triadica sebifera* (L.) Small, formerly *Sapium sebiferum* (L.) Roxb.] is a non-native invader moving through the southeastern United States. Tallow can invade a multitude of habitats, from coastal prairies to closed-canopy forests, forming monospecific stands and driving out native flora and fauna (Bruce and others 1997). Tallow has large impacts, both economically and ecologically, and is a growing concern among ecologists, landowners, and the public alike. An understanding of the ecology and biology of tallow, along with its effects on forest alteration, is necessary for any attempt at control to be successful. This paper aims to establish an accurate model for the branch biomass of Chinese tallow to be used in additive modeling and as a tool in future field research under the hypothesis that biomass increases with diameter and length.

INTRODUCTION

Tallow [*Triadica sebifera* (L.) Small, formerly *Sapium sebiferum* (L.) Roxb.], native to China and Japan and commonly referred to as the popcorn or chicken tree, is an aggressive invader in the southern United States (Bruce and others 1997). In the introduced range, tallow has the ability to expel native species and forms monospecific stands (Zou and others 2006). The Chinese tallow is versatile in its habitat range, able to tolerate both salt- and fresh-water flooding and shade (Barrilleaux and Grace 2000, Pattison and Mack 2009).

Tallow's rapid growth, high reproductive rates, competitive ability, and large range of tolerances make this species an incredible invader, and consequently, a substantial problem for native species, land owners, and managers. As with any invasive species, a comprehensive understanding of the biology and ecological interactions is a necessary foundation for any successful control or eradication program.

Comprehensive knowledge of the crown biomass is an important component in understanding the biology and ecological functioning of the species, specifically in the assessment of invasive and competitive abilities (Blossey and Nötzold 1995, Perry 1985, Weiner 2004). This study aims to create functional biomass models at the individual branch level to provide an understanding of the total crown biomass for Chinese tallow. Modeling at this small scale allows us to gain an accurate measure of the biomass in different components of the tree to be used in a wide array of comparisons and analyses. Branch models can be used to show the relationship between crown and stem biomass as an indication of competitive ability and become important in carbon storage and cycling dynamics, especially as displacement of native species persists (Blossey and Nötzold 1995, Brown 2002).

In this paper, working biomass models for individual branches of Chinese tallow were created under the hypothesis that biomass increases with length and diameter.

MATERIALS AND METHODS

Location

Tallow was identified and sampled from a site near Poplarville, MS on September 25, 2012. The study site is in Pearl River County, borders the Louisiana state line, and can be classified as a bottomland forest.

Sample Collection and Preparation

Five trees were felled and measured for diameter at breast height (d.b.h.) and total height. Branches were then cut from the bole and saved for measurement. Branches were further cut at any adjoining division that was over 1cm in diameter. Measurements were then taken for branch basal diameter and branch length. Two hundred branches were taken back to the lab for subsample measurements.

Subsample Analysis

The branch subsamples were oven dried at 38.9 °C for 24 hours, until a constant weight was

[1]Master of Science Candidate, Master of Science Candidate, and Associate Professor, respectively, Mississippi State University, College of Forest Resources, Department of Forestry, Mississippi State, MS 39762.

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reached. Using dry weight as an indicator of biomass, the samples were then weighed and recorded. This information was used in the creation of biomass models and was applied to branches from all five sampled trees.

Statistical Modeling

Biomass, length (L), and basal diameter (D) for each branch were entered into a data set in SAS (SAS Institute 2008) and fitted using regression analysis assuming that biomass was a function of D and/or L. Eight models were tested using varying relationships of L and D and histograms, probability plots of residuals, Akaike's Information Criteria (AIC), R^2, and RMSE values were evaluated for goodness of fit.

RESULTS

From the eight models tested we concluded that biomass is exponentially related to diameter and length. Between the exponential models, only slight differences emerged in goodness of fit between models that included basal diameter only ($R^2 = 0.71$), length only ($R^2 = 0.70$), and models including both diameter and length ($R^2 = 0.71$). Based on our fit statistics, parameter estimations from the best fitting model were used to construct our final branch biomass model:

$$Biomass(g) = 9.2813 + \exp(0.8342 * D) \qquad (1)$$

where *D* is the branch basal diameter in cm and *biomass* is measured as dry weight in g.

When graphed with measured data, this model proves to be an accurate depiction of branch biomass (fig. 1).

This biomass equation was then used to generate biomass estimations for all branches from the five sampled trees. These branch biomass numbers were then grouped by tree and added to obtain a total crown biomass per tree. A preliminary model of crown biomass versus tree d.b.h is shown in figure 2. This relationship appears to be exponential from our data but with the inclusion of a larger data set may in fact be sigmoidal, with a leveling off of crown biomass as tree d.b.h. increases.

DISCUSSION

Biomass models can be accurately constructed through a thorough statistical analysis and prove to be a useful tool in estimating the biomass contained within a tree. This is especially useful when quantifying the impact that an invasive species may have on carbon pools as it invades and displaces native species or when quantifying invasive and competitive abilities (Blossey and Nötzold 1995, Brown 2002, Perry 1985, Weiner 2004). Through our analysis we have found that branch biomass is exponentially related to branch basal diameter and an accurate model to calculate biomass is: Biomass(g)=9.2813+exp(0.8342*Diameter).

We additively calculated the total crown biomass for each tree sampled and compared them to our measured tree d.b.h. In doing so, we provided a first step that will be useful in determining a stem-to-crown ratio for each tree in future research. A tree with a higher crown biomass to stem biomass will demonstrate a higher competitive ability and will be a successful invader (Blossey and Nötzold 1995, Perry 1985, Weiner 2004). It is also important to note that the location in which a tree is found (forest edge versus interior) can have a significant impact on this ratio. In future research, we will show a comparison of tallow crown to stem ratios to those of native species in an attempt to understand and predict tallow's competitive ability in a quantifiable approach.

For field research it will prove useful to have a model in place to easily and accurately evaluate crown biomass from tree d.b.h. In our study, we showed a preliminary relationship between our calculated crown biomass and tree d.b.h. and plan to build upon this model in the future.

Accurate biomass predictions give great insight into a tree's competitive ability and also its role in carbon cycling within a habitat as an indication of stored forest carbon (Brown 2002). As invasive species are becoming an ever-increasing concern, it is critical that these components are understood to comprehend the full impacts of the non-native organisms and as a baseline for any successful control program.

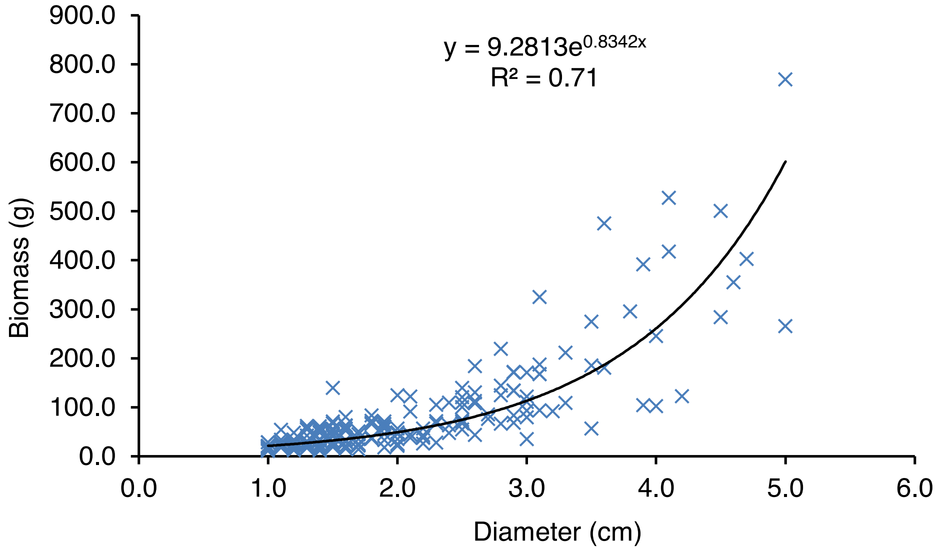

Figure 1--Relationship between Chinese tallow branch basal diameter and branch biomass for 200 samples with model and R^2 value.

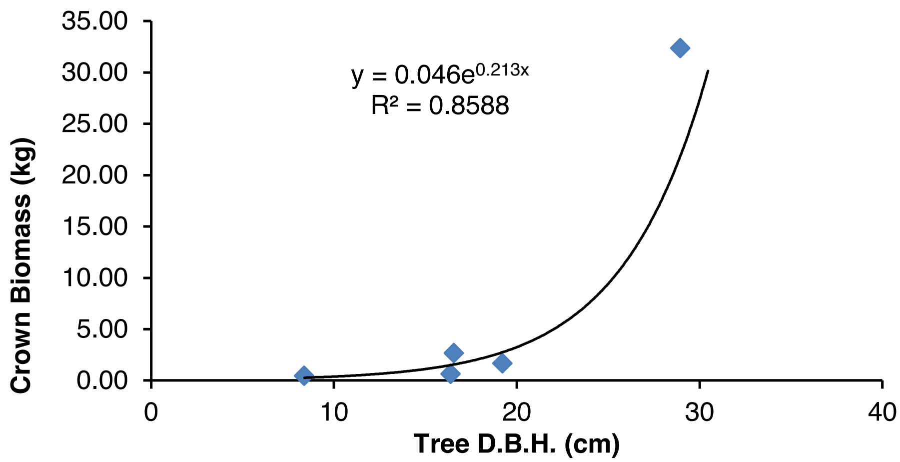

Figure 2--Relationship between Chinese tallow d.b.h. and crown total biomass with model and R^2 value.

ACKNOWLEDGMENTS

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LITERATURE CITED

Barrilleaux, T.C.; Grace, J.B. 2000. Growth and invasive potential of *Sapium sebiferum* (Euphorbiaceae) within the coastal prairie region: the effects of soil and moisture regime. American Journal of Botany. 87: 1099-1106.

Blossey, B.; Nötzold, R. 1995. Evolution of increased competitive ability in invasive nonindigenous plants: a hypothesis. Journal of Ecology. 83: 887-889.

Brown, S. 2002. Measuring carbon in forests: current status and future challenges. Environmental Pollution. 116: 363-372.

Bruce, K.A.; Cameron, G.N.; Harcombe, P.A.; Jubinsky, G. 1997. Introduction, impact on native habitats, and management of a woody invader, the Chinese tallow tree, S*apium sebiferum*. Natural Areas Journal. 17: 255-260.

Pattison, R.R.; Mack, R.N. 2009. Environmental constraints on the invasion of *Triadica sebifera* in the eastern United States: an experimental field assessment. Oecologia. 158: 591-602.

Perry, D.A. 1985. The competition process in forest stands. In: Cannell, M.G.R.; Jackson, J.E., eds. Attributes of trees as crop plants. Abbotts Ripton, UK: Institute of Terrestrial Ecology: 481-502.

Weiner, J. 2004. Allocation, plasticity and allometry in plants. Perspectives in Plant Ecology. 6: 207-215.

Zou, J.; Rogers, W.E.; DeWalt, S.J.; Siemann, E. 2006. The effect of Chinese tallow tree (S*apium sebiferum*) ecotype on soil-plant system carbon and nitrogen processes. Ecosystems Ecology. 150:272-281.

GROWTH OF CHINESE TALLOW IN A BOTTOMLAND FOREST IN SOUTHERN MISSISSIPPI

Nana Tian and Zhaofei Fan[1]

Abstract--Chinese tallow tree [*Triadica sebifera* (L.) Small, formerly *Sapium sebiferum* (L.) Roxb.] is a monoecious and deciduous tree, native to central and southern China. As a nonnative invasive tree species, it has aggressively invaded forestlands in southeastern United States, particularly the low- and bottom-land forests along the coastal region of the Gulf of Mexico. This study, on the basis of a destructive sample of 11 tallow trees collected from a bottomland oak-gum-cypress forest in the southern Mississippi, developed a group of individual tree level models to reflect the growth of diameter, total height, and standing volume with age for Chinese tallow. Moreover, we used the destructive sample data to explore allometric relationships between diameter and total height of Chinese tallow. These results are useful to estimate and compare competition potential and establishment rate of tallow trees relative to other native trees threatened by its invasion and rapid establishment. Resource managers could use this information to design efficient treatments to control and mitigate further invasion of tallow.

INTRODUCTION

The Chinese tallow tree [*Triadica sebifera* (L.) Small] has become an invasive species in the United States (Bruce 1993) since its introduction as an ornamental and potential oil species in the 1770s. Bruce and others (1997) reported that coastal ecosystems from Texas to North Carolina were severely invaded by tallow tree. In recent years it has expanded to non-coastal areas and regions further inland (e.g., northward), but has not become abundant in these areas. Studies show that tallow will invade new regions beyond current range (Pattison and Mack 2009, Wang and others 2011). Zou and others (2008) reported that tallow reached maturity after 3 years. Fast growth is an important factor for invasive species rapid colonization and establishment in the affected regions.

Avery and Burkhart (1983) regard growth modeling for a tree as the incremental increase in diameter, height, and volume during a certain period. In growth and yield modeling, diameter at breast height (d.b.h.) is the most used and easily obtained measure of tree size (Avery and Burkhart 1983). In order to estimate the volume of an individual tree or forests, height is another traditionally measured tree attribute (Hann and Larsen 1991). Furthermore, a basal area equation based on d.b.h. was more linear related to volume growth (Hökkä and Groot 1999). Modeling the increment of individual tree diameter or basal area is usually accomplished through a composite or a modifier model (Zhang and others 2004). In a composite model, independent variables usually refer to tree characteristics (such as tree size, crown ratio, vigor) and stand conditions including stand age, site index, and stand density. Response variables are either diameter or basal area increment (Vanclay 1994, Zhang and others 2004). By contrast, a modifier model represents a potential maximum attainable growth for a tree and is able to explain tree growth biologically (Zhang and others 1997). In recent studies (Budhathoki and others 2008, Zhang and others 2004) potential growth is also modeled using more flexible functions (e.x. Chapman-Richards and logistic model) which are physiologically based.

If non-homogeneity of variance exists, best linear unbiased estimate of the parameters is not realized. Therefore, a modified modeling approach using weighted least square regression is used. Meng and Tsai (1986) built a volume growth model for loblolly pine (*Pinus taeda* L.) and compared the weight value of diameter-squared*height (D^2H) and diameter-squared (D^2) in the model. Diéguez and others (2006) developed an individual tree volume model for Scots pine (*Pinus sylvestris* L.) using a modified second-order continuous autoregressive method which was mainly to resolve the high autocorrelation of the data.

[1]Master's Candidate and Assistant Professor, respectively, Mississippi State University, College of Forest Resources, Department of Forestry, Mississippi State, MS 39762.

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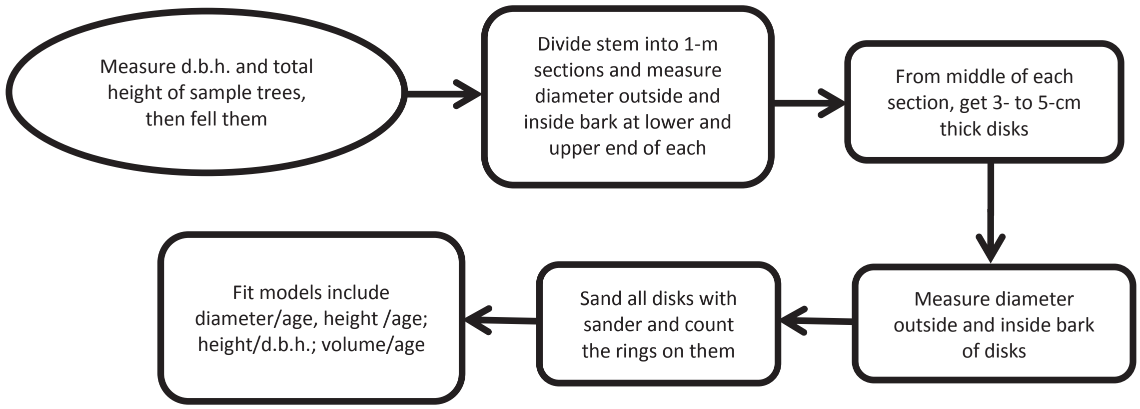

Figure 1--Details of field research and lab work.

Besides the non-constancy of variance during field data collection, spatial correlation still exists leading to the development of distance-dependent models (Aguirre and others 2003, Porté and Bartelink 2002, Zhang and others 2004). However, the difficulty of getting the specific spatial location of each tree makes it impossible under some situations. As a result, a radical intermediary growth model between spatial and non-spatial models appeared. Perot and others (2010) used this method to build a growth model for sessile oak [*Quercus petraea* (Mattuschka) Liebl.] and Scots pine in north-central France, and the results indicated that the radical intermediary growth model had similar behavior with the distance-dependent model.

To obtain the growth rate of tallow in bottomland oak-gum-cypress in southern Mississippi, a group of growth models were constructed to quantify the relationships among diameter, height, volume, and tree age at individual tree level. Models included height, d.b.h., and volume growth with age, as well as allometric models between d.b.h. and height.

DATA COLLECTION

Destructive sampling was used to obtain the tallow tree age and profile data. Before felling sample trees in the field, total height and d.b.h. were 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 to 5 cm thickness were then extracted from the midpoint of each section. Inside bark and outside bark diameters were obtained at the upper end of each section. Diameter at selected height positions (0.8 m and 5.3 m) was also recorded. Disks were transported to the U. S. Forest Service laboratory in Starkville, MS and sanded for an accurate ring count to determine tree age. Details of the field work and lab work were shown in figure 1. To accurately calculate the stem volume, volume of each section in m^3 was firstly computed using equation 1.

$$\mathrm{v} = \frac{\pi}{8}(D^2 + d^2) * \mathrm{l} \qquad (1)$$

where: l was the length of each section (1 m); and *D* and *d* were diameters at the upper and lower end of each section. By adding up the volume of all sections from one tree, the whole stem volume of individual trees was determined (Vol_i) which was used in volume modeling.

METHODS

Models in this study were: d.b.h. increased with age, height grew with age and d.b.h., and volume grew with age. Previous studies showed that there were several models to describe tree height and diameter growth process. Here, we mainly try the Schumacher model, Chapman-Richards model, logistic model, and Mitscherlich model to quantify tallow tree growth (table 1). In all these models, *A*, *r*, and *c* are the parameters to be estimated; *y* is the annual increment of diameter/height/volume and *t* indicates tree age. Based on the collected field data, these four models [equation (1), (2), (3), (4) in table 1] were fitted using SAS statistical software (SAS Version 9.2, SAS Inc., Campus Drive, Cary, NC) and from them we chose the best fitted model.

Table 1--Available growth models to be fitted for Chinese tallow tree

Model name	Model form[a]
Schumacher	$y = a * e^{-\frac{b}{t}}$ (1)
Chapman-Richards	$y = A(1 - e^{-rt})^c$ (2)
Logistic model	$y = \frac{A}{a + me^{-rt}}$ $A, m, r > 0$ (3)
Mitscherlich	$y = A(1 - e^{-rt})$ $A, r > 0$ (4)

[a]In all models, *A, r*, and *c* are the parameters to be estimated. *y* is the annual increment of diameter/height/volume and t indicates tree age.

RESULTS

All fitted models showed that: the DBH-age, volume-age and total height-DBH relationships were quantified by exponential models, while the total height-age relationship was expressed by Logistic regression model (fig. 2). Based on fitted regression models, a 16-year-old tallow tree, on average, can reach up to 25 cm (10 inches) in d.b.h., 18 m (63 feet) in total height, and 0.3 m^3 (10 cubic feet) in stem volume. This d.b.h. and height growth rate is comparable to American sycamore (*Platanus occidentalis* L.), a commonly found bottomland species. Tallow's growth rate is an important parameter to estimate its colonizing and establishing potential in different forest types.

DISCUSSION AND CONCLUSIONS

Tallow has severely invaded coastal forest lands, and its rapid growth makes it an aggressive competitor to many native tree species. To accurately reflect growth rate of tallow in bottomland oak-gum-cypress forests, additional samples are required. Moreover, to evaluate tallow's competitive ability and invasiveness of different forest communities, estimation of tallow's growth under various site and environmental conditions needs further research. Model results show that tallow trees in southern forest stands grow rapidly; meanwhile, these exotic tallow trees are still on the juvenile stage with diameter and height exponentially growing. Hence, it is the time for managers to make some efficient treatments to control further growth and invasion.

LITERATURE CITED

Aguirre, O.; Hui, G.; von Gadow, K.; Jiménez, J. 2003. An analysis of spatial forest structure using neighbourhood-based variables. Forest Ecology and Management. 183: 137-145.

Avery, T.B.; Burkhart, H.E. 1983. Forest measurements. In: Avery, T.B., ed. Forest measurements. New York: McGraw-Hill Book Company. 331 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.

Bruce, K.A.; Cameron, G.N.; Harcombe, P.A.; Jubinsky, G. 1997. Introduction, impact on native habitats, and management of a woody invader, the Chinese tallow tree, *Sapium sebiferum* (L.) Roxb. Natural Areas Journal. 17: 255–260.

Budhathoki, C.B.; Lynch,T.B.; Guldin, J.M. 2008. Nonlinear mixed modeling of basal area growth for shortleaf pine. Forest Ecology and Management. 255: 3440-3446.

Diéguez, A.U.; Dorado, F.C.; Álvarez, J.G.; Rojo, G.A. 2006. Compatible taper function for Scots pine plantations in northwestern Spain. Canadian Journal of Forest Research. 36: 1190-1205.

Hann, D.W.; Larsen, D.R. 1991. Diameter growth equations for fourteen tree species in Southwest Oregon. Res. Bull. 69. Corvallis, OR: Oregon State University, Forestry Research Laboratory. 18 p.

Hökkä, H.; Groot, A. 1999. An individual-tree basal area growth model for black spruce in second-growth peatland stands. Canadian Journal of Forest Research. 29: 621-629.

Meng, C.H.; Tsai, W.Y. 1986. Selection of weights for a weighted regression of tree volume. Canadian Journal of Forest Research. 16: 671-673.

Pattison, R.R.; Mack, R.N. 2009. Environmental constraints on the invasion of *Triadica sebifera* in the eastern United States: an experimental field assessment. Oecologia. 158: 591-602.

Perot, T.; Goreaud, F.; Ginisty, C.; Dhôte, J-F. 2010. A model bridging distance and distance-independent tree models to simulate the growth of mixed forests. Annals of Forest Science. 67: 502-513.

Porte´, A.; Bartelink, H.H. 2002. Modeling mixed forest growth: a review of models for forest measurement. Ecological Modeling. 150: 141-188.

Vanclay, J.K. 1994. Modeling forest growth and yield: application to mixed tropical forests. United Kingdom: CAB International. 312 p.

Wang, H.H.; Grant, W.E.; Swannack, T.M. [and others]. 2011. Predicted range expansion of Chinese tallow tree (*Triaduca sebifera*) in forestlands of the southern United States. Diversity and Distributions. 17: 552-565.

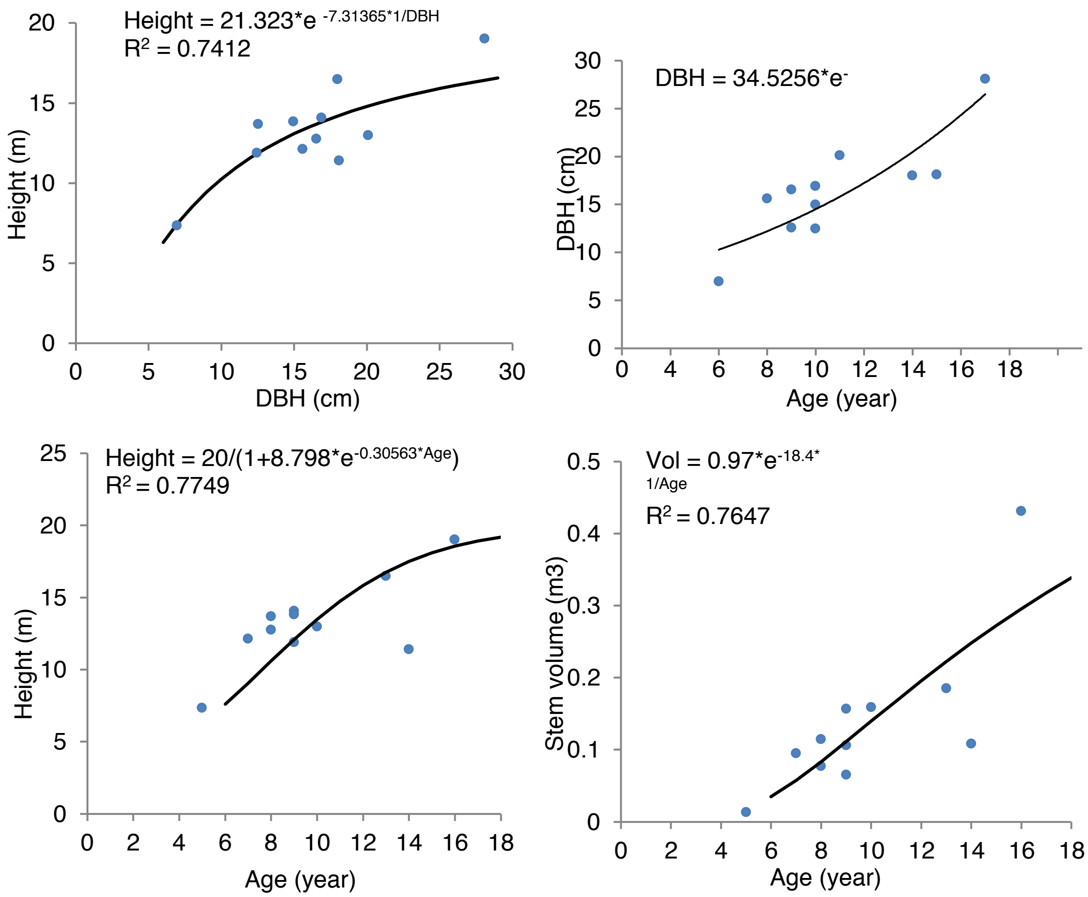

Figure 2--Fitted models of Height-DBH, DBH-Age, Height-Age, and Volume-Age of tallow trees, where DBH is diameter at breast height.

Zhang, S.; Amateis, R.L.; Burkhart, H.E. 1997. Constraining individual tree diameter increment and survival models for loblolly pine plantations. Forest Science. 43: 414-423.

Zhang, L.; Peng, C.; Dang, Q. 2004. Individual-tree basal area growth models for jack pine and black spruce in northern Ontario. The Forestry Chronicle. 80(3): 366-375.

Zou, J.; Rogers, W.E.; Siemann, E. 2008. Increased competitive ability and herbivory tolerance in the invasive plant *Sapium sebiferum*. Biological Invasions. 10: 291–302.

Biomass

Moderator:

Randall Rousseau
Mississippi State University
College of Forest Resources

ESTIMATION OF ABOVE GROUND BIOMASS FOR MULTI-STEMMED SHORT-ROTATION WOODY CROPS

Brian A. Byrd, Wilson G. Hood, Michael C. Tyree, and Dylan N. Dillaway[1]

With the increasing interest in short-rotation woody crop (SRWC) systems, an accurate yet quick, non-destructive means for determining aboveground biomass is necessary from both management and research perspectives. Equations determined using measures of stem volume (D^2H) have been used extensively to estimate plant biomass in single-stem plants (Crow 1978, Ter-Mikaelian and Korzukhin 1997, Tritton and Hombeck 1982). However, little data is available regarding this relationship in multiple-stemmed individuals (i.e. Zhou and others 2007), particularly those grown from cuttings. The objective of this study was to determine how accurately a simple volume equation (D^2H) can predict aboveground biomass (AG_{mass}) for two important short-rotation woody crop species,eastern cottonwood (*Populus deltoides* Bartram ex. Marsh.) and black willow (*Salix nigra* Marsh.). This research evaluated the use of total stem volume (ΣD^2H) to predict AG_{mass} both within and across a species and soil type, compared cottonwood plants regenerated from cutting with plants regenerated from seed, and evaluated the use of a universal curve across our full dataset. We hypothesized that a universal model can be developed that will accurately predict AG_{mass} across species, soil type, and regeneration method.

Over 320 observations were used from multiple datasets to construct volume equations (table 1). Sites utilized in this study covered a wide longitudinal range from northern Wisconsin to northern Louisiana. Contrasting soil types and textures, from very high clay percentages to fine sandy loam, were also represented. All data represents first year growth after regeneration from either seed or cutting. For each primary stem, plant heights were measured from the ground to the base of the terminal bud; basal diameters were measured as close to the node as possible to not account for artificial swell near the base (< 1 cm from the node).

Calculated volumes for each primary stem were summed to calculate a total plant stem volume (ΣD^2H). The relationship of ΣD^2H to AG_{mass} and goodness of fit were determined by linear regressions using the REG procedure in SAS (SAS Institute, Cary, NC). Comparison among treatments (i.e., species, soil texture, regeneration method) were made by analysis of covariance (ANCOVA; same-slopes analysis) using the GLM procedure in SAS 9.3.

The relationship between natural log transformed AGmass (*ln*AGmass) and natural log transformed D^2H (lnD^2H) accounted for a respective 84 and 90 percent of the variation in cottonwood (fig.1A; $r^2 = 0.84$, $p < 0.0001$) and willows (fig.1C; $r^2 = 0.90$ $p < 0.0001$) on loam sites (table 1). Applying this relationship to cottonwood and willow on clay sites described 93 percent (fig.1B; $r^2 = 0.93$, $p < 0.0001$) and 91 percent (fig.1D; $r^2 = 0.91$, $p < 0.0001$) of the variation in lnAGmass, respectively. The slope of cottonwood and willow were not significantly different on either loam ($p = 0.3427$) or clay ($p = 0.2811$) sites while intercept differed significantly for comparisons on loam ($p < 0.0001$) and clay ($p < 0.0001$). When cottonwoods on loam and clay were compared, significant difference emerged in both the slope ($p = 0.0004$) and the intercept ($p < 0.0001$). Significant differences were also found between soil textures for willow in both slope ($p = 0.0004$) and intercept ($p < 0.0001$).

When comparing cottonwoods regenerated from seed ($r^2 = 0.99$, $p < 0.0001$) with all cottonwoods from cuttings ($r^2 = 0.92$, $p < 0.0001$), significant

[1]Graduate Assistant, Graduate Student, and Assistant Professor, respectively, Louisiana Tech University, School of Forestry, Ruston, LA 71272; and Assistant Professor, Unity College, Center for Natural Resource Management and Protection, Unity, ME, 04988.

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Table 1--Parameter estimates for the relationship between the sum of D^2H and above ground mass across soil texture and regeneration treatments using natural log transformed dependent and independent variables

				---------------Parameter estimates---------------	
Regression	n	r^2	p-value	Slope	Intercept
Cottonwood clay	44	0.93	<0.0001	0.84352 ± 0.03482	-0.19723 ± 0.18080
Cottonwood loam	24	0.84	<0.0001	0.61248 ± 0.05639	1.46410 ± 0.38300
Willow clay	48	0.91	<0.0001	0.90110 ± 0.03981	-0.58668 ± 0.19752
Willow loam	24	0.90	<0.0001	0.68503 ± 0.04819	0.89860 ± 0.31981
Cottonwood seed	113	0.99	<0.0001	0.70935 ± 0.00713	-0.02231 ± 0.02027
Cottonwood cuttings	68	0.92	<0.0001	0.80657 ± 0.02860	0.05218 ± 0.16611

differences appeared both in slope ($p < 0.0001$) and in intercept ($p < 0.0001$). Finally, we calculated a single regression across all our data ($r^2 = 0.97$, $p < 0.0001$). In spite of this good fit, a comparison of observed and predicted mass of the median individual from each treatment revealed considerable (as high as 50 percent) error when a universal equation is used.

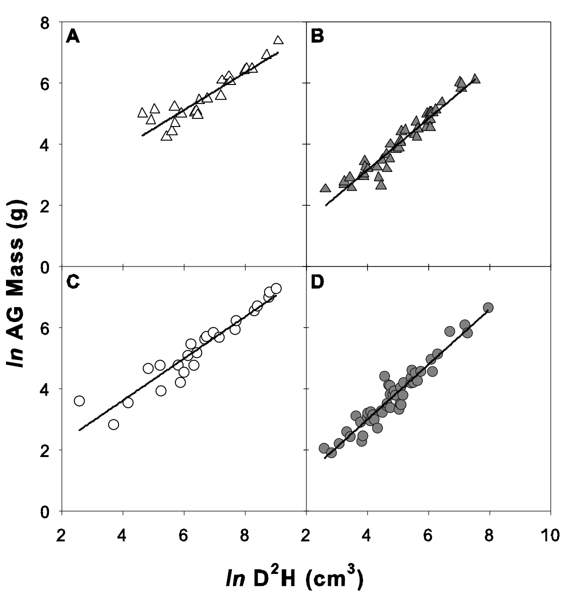

Figure 1--The relationship between natural log transformed above-ground mass and natural log transformed D^2H for cottonwoods grown in loam (A; $y = 0.61248x + 1.46410$, $p < 0.0001$) and clay soil textures (B; $y = 0.84352x - 0.19723$, $p < 0.0001$); and for willows grown in loam (C; $y = 0.68503x + 0.89860$, $p < 0.0001$) and clay soil textures (D; $y = 0.90110x - 0.58668$, $p < 0.0001$).

In conclusion, we find a very strong relationship between $ln AG_{mass}$ and D^2H within a species, soil type, or regeneration method. However, the differences between these conditions affect the output of the model greatly, which we feel diminishes the usefulness of the universal model from both a management and research standpoint.

LITERATURE CITED

Crow, T.R. 1978. Notes: common regressions to estimate tree biomass in tropical stands. Forest Science. 24:110-114.

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

Tritton, L.M.; Hombeck, J.W. 1982. Biomass equations for sixty-five North American tree species of the northeast. Gen. Tech. Rep. NE-69. Broomall, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 46 p.

Zhou, X.; Brandle, J.R.; Schoenberger, M.M.; Awada, T. 2007. Developing above-ground woody biomass equations for open-grown multiple-stemmed tree species: shelterbelt grown Russian-olive. Ecological Modeling. 202:311-323.

FINANCIAL RETURN FROM TRADITIONAL WOOD PRODUCTS, FEEDSTOCK, AND CARBON SEQUESTRATION IN LOBLOLLY PINE PLANTATIONS IN THE SOUTHERN U.S.

Umesh K. Chaudhari and Michael B. Kane[1]

We know that planting trees is a key approach for mitigating climate change; however, we are uncertain of what planting density per unit of land and what cultural regimes are needed to optimize traditional timber products, feedstock, and carbon sequestration. This study was undertaken to determine the financial return from the aforementioned timber commodities in planted loblolly pine (*Pinus taeda* L.) stands. This study uses 17 years of data from two Florida installations of the University of Georgia Plantation Management Research Cooperative Culture x Density study on loblolly pine in the Lower Coastal Plain to examine the role of site index (SI), planting density, and silvicultural treatment on financial returns. These planted loblolly pine sites were categorized into high site index (30.7 m) and low site index (19.2 m). Each installation received two silvicultural treatments: intensive and operational. Operational plantations received standard chemical site preparation and four fertilizations through age 17 whereas intensive plantations received additional herbicide treatments resulting in complete sustained control of competing vegetation throughout their rotation and eight fertilizations. Finally, each cultural regime was planted at four different densities: 1,483; 2,224; 2,965; and 3,707 trees per ha.

Tree biomass was measured using allometric equations from Baldwin (1987) and Pienaar and others (1987). Financial calculations were made for traditional wood products, feedstock, and carbon sequestration using the discounted cash flow technique: net present value (NPV). South-wide timber product prices of the first quarter of 2012 were used for analysis (Timber Mart-South 2012). The bioenergy price was assumed to be $3 per green ton (Personal communication. 2013. Dale Greene, Professor of Forest Operations, University of Georgia, 180 E Green St, Athens, GA 30602). Carbon dioxide (CO_2) price was assumed to be $5 per green metric ton. The average costs of site preparation, plantation, herbicide, and fertilizer were inflated to the 2012 prices. Financial calculations were made with a 6 percent discount rate, and final values were reported as before tax.

On the high SI site, the operational regime produced as much biomass as the intensive regime (fig. 1). In contrast, on the low SI site, the intensive regime enhanced biomass growth and had a higher economic return than the operational regime. The lower density stands (1,483 to 2,224 trees per ha) produced higher biomass and accrued greater total financial return in terms of traditional wood products, carbon sequestration, and bioenergy feedstock than the higher density stands, except on one low SI site under operational management. On this particular site, greater densities (i.e. 2,965 to 3,706 trees per ha) were needed to produce more biomass up through age 17 (table 1 and fig. 1).

There was little sawtimber production through age 17. Optimum sawtimber harvests are typically performed on stands older than those evaluated here, when sawtimber makes up a greater proportion of the harvested material. Subsequent analysis will use a growth and yield simulator to predict optimal rotations. Feedstock from tops and branches and trees of < 4 inches diameter at breast height accrued relatively little economic return after considering logging costs. If we assume that the only product is bioenergy feedstock, at least $7 per green metric ton stumpage is needed for profitability of the study stands. In that market condition, a rotation as short as 6 years for planted pine can be

[1]Graduate Research Assistant and Professor, respectively, University of Georgia, Warnell School of Forestry and Natural Resources, Athens, GA 30602.

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Table 1--Net present values for traditional products, feedstock, carbon, and combined products (i.e., sum of all the products) in the Lower Coastal Plain by site index, cultural regime, and planting density[a]

TPH[b]	Intensive plantation: Traditional	Feedstock	Carbon	Overall	Operational plantation: Traditional	Feedstock	Carbon	Overall
	dollars ($)							
High site index								
1483	1,168	58	1,285	2,511	2,040	63	1,512	3,615
2224	962	64	1,411	2,437	1,977	69	1,689	3,735
2965	508	51	1,269	1,828	1,217	71	1,495	2,783
3706	329	46	1,252	1,627	1,085	75	1,520	2,680
Low site index								
1483	1,192	64	1,451	2,707	-86	48	415	377
2224	605	65	1,432	2,102	-98	53	461	416
2965	194	73	1,337	1,604	92	65	816	973
3706	-263	71	1,144	952	93	96	950	1,139

[a]Values at optimum rotation of age 16 or 17 years. Analysis made from the actual stand data.
[b]TPH = trees per hectare.

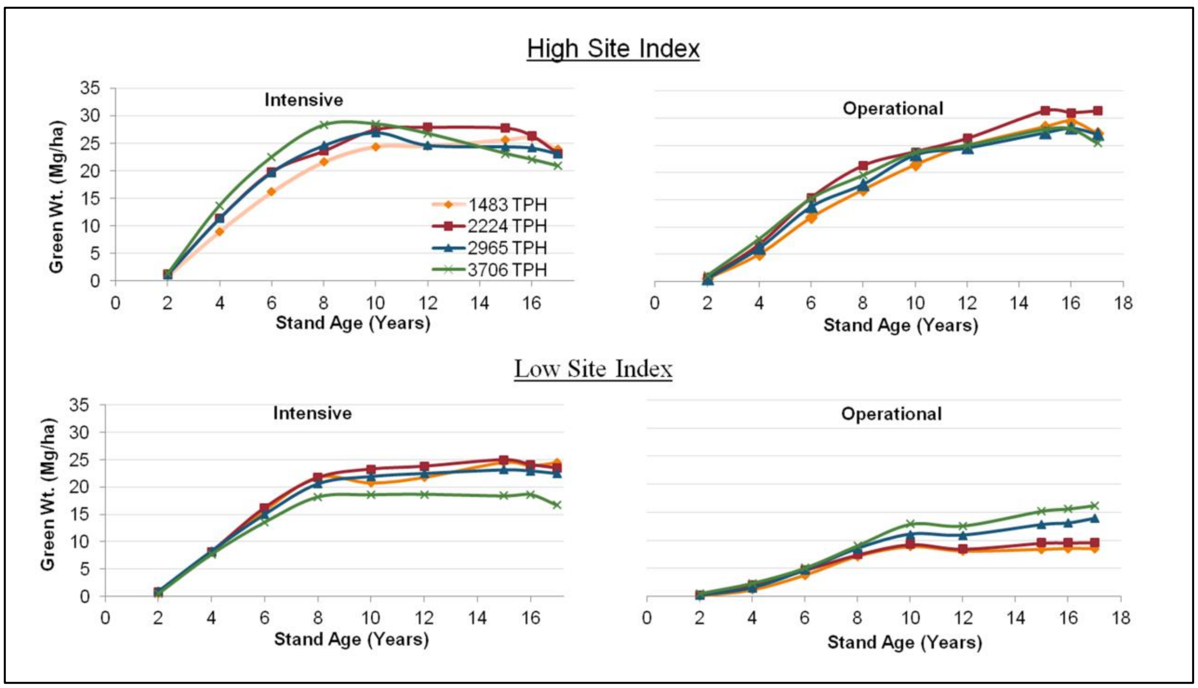

Figure 1--Mean annual increment (MAI) of planted loblolly pine in the Lower Coastal Plain by site index, cultural regime, and planting density.

anticipated in high SI operational stands. In the present bioenergy market, a modest CO_2 price of $5 per green metric ton can increase financial return by over $1,500 per ha for some sites (table 1).

LITERATURE CITED

Baldwin, V.C. 1987. Green and dry-weight equations for above-ground components of planted loblolly pine trees in the West Gulf region. Southern Journal of Applied Forestry. 11(4): 212-218.

Pienaar L.V.; Burgan, T.; Rheney, J.W. 1987. Stem volume, taper and weight equations for site-prepared loblolly pine plantations. Plantation Management Res. Coop. Tech. Rep.1987-1. Athens, GA: University of Georgia, School of Forest Resources. 13 p.

Timber Mart-South. 2012. Timber Mart-South market news quarterly. http://www.timbermart-south.com/pdf/1Q2012news.pdf. [Date accessed: June 19, 2013].

Mohr, C. 1896. The timber pines of the southern United States. Bull. 13. Washington, DC: U.S. Department of Agriculture Division of Forestry. 176 p.

Noss, R.F.; LaRoe, E.T.; Scott, J.M. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Biol. Rep. 28. Washington, DC: U.S. Department of the Interior Fish and Wildlife Service, National Biological Service. 58 p.

Outcalt, K.W. 2000. Occurrence of fire in longleaf pine stands in the southeastern United States. In: Proceedings, 21st Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station. 21: 178-182.

Sargent, C.S. 1884. Report on the forests of North America (exclusive of Mexico): 10th census report. Washington, DC: U.S. Department of the Interior, Census Office. Vol. 9. 612 p.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth, and management. 1st ed. Washington, DC: Charles Lathrop Pack Forestry Foundation. 429 p. In cooperation with: The U.S. Department of Agriculture Forest Service.

Walker, L.C.; Wiant, H.V., Jr. 1973. Silviculture of longleaf pine. Nacogdoches, TX: Stephen F. Austin State University, School of Forestry. 105 p.

MEASURING AND PARTITIONING SOIL RESPIRATION IN SHARKEY SHRINK-SWELL CLAYS UNDER PLANTATION GROWN SHORT-ROTATION WOODY CROPS

Wilson G. Hood, Michael C. Tyree, Dylan N. Dillaway, and Theodor D. Leininger[1]

The Lower Mississippi Alluvial Valley (LMAV) offers an ecological niche for short-rotation woody crop (SRWC) production by mating marginal agricultural land with optimal growing conditions. Approximately 1.7 million ha within the LMAV consist of Sharkey shrink-swell clays. They are considered marginal in terms of traditional agricultural productivity due to their hydric characteristics (Vepraskas and Richardson 1997). However, these soils are ideal for SRWC production using native, well-adapted species.

Total soil carbon dioxide (CO_2) efflux (F_S) is a composite measure of many processes. One major component is rhizosphere respiration (R_{Rhizo}), where CO_2 is derived through both growth and maintenance of plant roots and rhizosphere microorganisms. The second major component is heterotrophic respiration (R_H), which is CO_2 derived from soil macro- and microorganisms. Soil temperature and moisture are among the strongest environmental factors exhibiting control over F_S, (Bond-Lamberty and Thomson 2010, Boone and others 1998, Carlyle and Than 1988, Janssens and Pilegaard 2003, Raich and others 2002). To gain a better understanding of F_S, a separation of its components is essential since both R_{Rhizo} and R_H respond differently to temperature and moisture, and their relative contribution to F_S will be variable throughout the year (Boone and others 1998). The objectives of this work are to determine: (1) the suitability of the PVC root-exclusion method for measuring and partitioning F_S into its primary components, R_H and R_{Rhizo}, throughout the year of establishment; and (2) if F_S and its primary components are influenced by species and management intensity.
The work was conducted on an Army Corps of Engineers tract, which is 36.42 ha in size and located near Hollandale, MS (33˚ 09' 03.51" N, 90˚ 54' 17.65" W). This region has a mean annual temperature of 23.4 ˚C and receives on average 123.3 cm of precipitation annually (NOAA 2012). The soil is classified as Sharkey series (very-fine, smetitic, thermic, Chromic, Epiaquerts). At the end February 2012, the site was established using cuttings of native *Salix nigra* Marsh. and clonal *Populus deltoids* Bartram ex Marsh. This study is a randomized complete block (RCB) design replicated three times. Treatments were arranged as a 2 by 3 full factorial, where tree species was randomly assigned across three planting intensities (326; 1,006; and 1,659 trees ha^{-1}).

Soil gas exchange measurements were taken four times during the 2012 growing season, and one dormant season measurement was taken in February 2013. Gas exchange was measured *in-situ* using a LiCor 8100A infra-red gas analyzer (IRGA) with a 10-cm-diameter survey chamber (LiCor Biosciences Inc., Lincoln, NE). Volumetric soil moisture (SM; average across 0 to 6 cm depth) and soil temperature (TC; 5 cm depth) were measured concurrently with gas exchange measurements using a Theta Probe ML2x (LiCor 8100-204; Delta-T Devices Ltd., Cambridge, England) and an Omega soil temperature probe (LiCor 8100-201), respectively. Soil efflux collars were manufactured to LiCor specification using 10-cm polyvinyl chloride pipe (PVC, schedule SDR 35) cut to 5 cm in length with a 45º bevel on one end. Collars were installed the evening prior to taking measurements (LI-COR 2010) to a depth of approximately 3 cm. F_S was separated into its autotrophic and heterotrophic respiratory components using the root-exclusion method (Bond-Lamberty and others 2011). Briefly, root-exclusion collars were manufactured from 25.4-

[1]Graduate Research Assistant and Assistant Professor, respectively, Louisiana Tech University, School of Forestry, Ruston, LA 71270; Assistant Professor, Unity College, Center for Natural Resource Management and Protection, Unity, ME 04988; and Supervisory Research Plant Pathologist, USDA Forest Service, Southern Research Station, Stoneville, MS 38776-0227.

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cm-diameter PVC pipe (schedule 40) cut to 50-cm lengths. Root-exclusion collars were manually driven to a depth of 50 cm, where the top edge of the exclusion was flush with the surrounding soil to prevent altering the soil-water status within the root-exclusion zone. The distance between paired measurement collars for F_S and R_H did not exceed 30 cm. One measurement was taken inside the root-exclusion zone, and one measure was taken outside of the root-exclusion zone. Heterotrophic respiration (R_H) was estimated from the measurement taken within root-exclusion zone. All measurements were averaged across subsamples for each experimental unit (plot). Treatment effects were tested by analysis of variance (ANOVA) with repeated measures using the MIXED procedure in SAS 9.3.1.

The first field campaign took place 14 days following the complete installation of root-exclusion zones. We compare inside the root exclusion (R_H) to outside the root exclusion (F_S) to test for the presence of any pulse respiration that might have resulted from the potentially traumatic nature of installing the root-exclusion zones and found no difference ($p > 0.05$; $n = 96$). We repeated this test 54 days after root-exclusion installation (98 days after planting); the resulting data suggest a clear difference ($p < 0.05$; $n = 91$) between inside and outside the root-exclusion zone, indicating that the difference at 54 days following installation is driven by an emerging autotrophic efflux component. Not only have we established that our root-exclusion zones are not influenced by pulse respiration and are effectively excluding autotrophic sources, but we also demonstrated their viability throughout the first growing season (fig. 1). One can see that the root-exclusion zones not only maintain effectiveness throughout the growing season but also yield a seasonal trend where the relative contributions of R_{Rhizo} and R_H to F_S are consistent with those report in the literature (Bond-Lamberty and others 2004).

Installing a large segment of PVC in the soil to exclude roots spawned concern over whether or not soil temperature and moisture within the exclusion would be representative of the surrounding soil. The influence of abiotic properties were assessed utilizing paired measurements. A comparison of soil temperature yields a significant ($p < 0.01$) time by exclusion interaction; however, this interaction is largely driven by a large sample size. Repeating this assessment for soil moisture revealed no interaction ($p = 0.99$) for time by exclusion. Soil moisture within a campaign does exhibit limited variability during some campaigns, and these minor differences are likely due to a buffering effect of the PVC when going into, and emerging from, a drydown event. Implications of this work will provide future investigators with some insight into the suitability of root exclusion and the response of F_S and its components in shrink-swell clay soil.

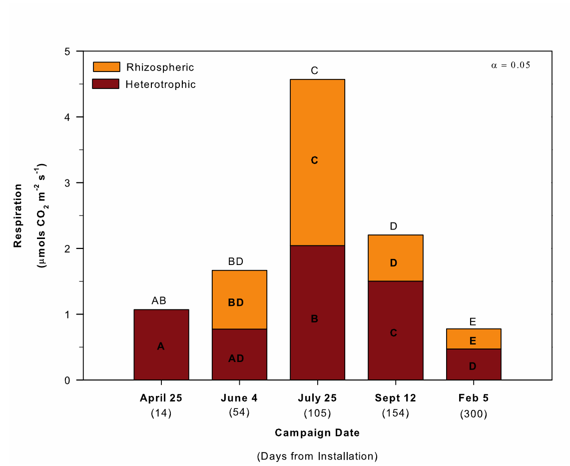

Figure 1—The relative contributions of R_{Rhizo} and R_{Hetero} to F_S. Letters within a color show significant differences, while letters atop bars indicate significant differences among F_S.

LITERATURE CITED

Bond-Lamberty, B.; Bronson, D; Bladyka, E.; Gower, S.T. 2001. A comparison of trenched plot techniques for partitioning soil respiration. Soil Biology and Biochemistry. 43: 2108-2114.

Bond-Lamberty, B.; Thomson, A. 2010. Temperature-associated increases in the global soil respiration record. Nature. 464: 579-582.

Bond-Lamberty, B.; Wang, C.; Gower, S.T. 2004. A global relationship between the heterotrophic and autotrophic components of soil respiration? Global Change Biology. 10: 1756-1766.

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.
Carlyle, J.; Than, U.B. 1988. Abiotic controls of soil respiration beneath an eighteen-year-old *Pinus radiata* stand in south-eastern Australia. Journal of Ecology. 76: 654-662.

Janssens, I.A.; Pilegaard, K. 2003. Large seasonal changes in Q_{10} of soil respiration in a beech forest. Global Change Biology. 9: 911-918.

LI-COR. 2010. Automated soil CO_2 flux system & LI-8150 multiplexer instruction manual. 1st ed. Lincoln, NE: LI-COR. 404 p.

NOAA [National Weather Service]. 2011. Climate data, Jackson, MS. http://www.nws.noaa.gov/climate/. [Date accessed: January 5, 2012].

Raich, J.W.; Potter, C.S.; Bhagawati, D. 2002. Interannual variability in global soil respiration, 1980-94. Global Change Biology. 8: 800-812.

Vepraskas, M.; Richardson, J. 1997. Review of the aquic conditions and hydric soils status of the Sharkey and related vertisols. U.S. Department of Agriculture NRCS Soils Division. http://www.ar.nrcs.usda.gov/soils/. [Date accessed: January 1, 2012].

WOODY BIOMASS AND VOLUME FOR FOUR TREE SPECIES IN MISSOURI FORESTS

Charles D. Keating and David R. Larsen[1]

Global energy supply concerns and increasing energy consumption are forcing our society to look into non-fossil-based energy sources and fuels to supplement humanity's ever growing energy requirements (Hahn 1984, Jenkins and others 2004, Smith 1985, Stortz 1975). Biofuels have received increased attention over recent years including biomass from plant and animal sources. This study examines woody biomass and volume available in four tree species commonly found in southeastern Missouri forests. Approximately 200 samples of black oak (*Quercus velutina* Lam.), white oak (*Q. alba* L.), post oak (*Q. stellata* Wangenh.), and hickories (*Carya* spp.) were selected for the study. Pre-harvest data collected from these samples included diameter, total height, and crown height.

The selected samples were then felled, and green weights of each tree were gathered using a Volvo front-end loader equipped with a grapple lift and a hydraulic load cell. Total tree weight with top was collected (fig. 1) before the tops were removed, allowing for merchantable weight data collection. Individual log weights were also collected from stems that were bucked into sawlogs. This method of collecting biomass data greatly speeds the process.

Sample sections were collect for each log to determine specific gravity and dry weight. We are currently in the process of fitting biomass equations and evaluating published equations for use in Missouri.

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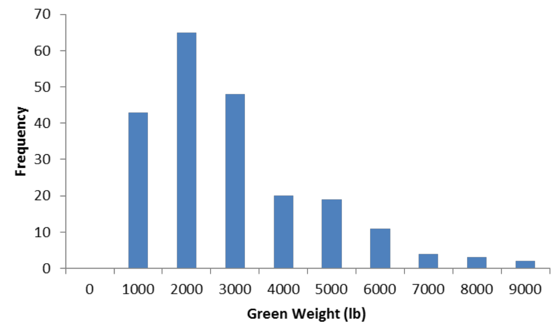

Figure 1--Observed green weights for the study (in pounds).

LITERATURE CITED

Hahn, J.T. 1984. Tree volume and biomass equations for the Lake States. Res. Pap NC-250. St Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 10 p.

Jenkins, J.C.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A. 2004. Comprehensive database of diameter-based biomass regressions for North American tree species. Gen. Tech. Rep. NE-319. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station. 45 p.

Smith, W.B. 1985. Factors and equations to estimate forest biomass in the North Central region. Res. Pap. NC-268. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 6p.

Stortz, L. 1975. Total tree volume and weight equations and tables for four upland hardwoods in southern Illinois. Carbondale, IL: Southern Illinois University. 96 p. M.S. thesis.

[1]Graduate Research Assistant and Professor, respectively, University of Missouri, The School of Natural Resources, Columbia, MO 65211.

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EARLY COMPETITIVE EFFECTS ON GROWTH OF LOBLOLLY PINE GROWN IN CO-CULTURE WITH SWITCHGRASS

Kurt J. Krapfl, Scott D. Roberts, Randall J. Rousseau, and Jeff A. Hatten[1]

Abstract--This study: (1) examined competitive interactions between switchgrass and loblolly pine grown in co-culture, and (2) assessed early growth rates of loblolly pine as affected by differing switchgrass competition treatments. Co-cultures were established and monitored on two Upper Coastal Plain sites for 2 years. The Pontotoc site has a history of agricultural use with some likely residual fertility, while the Starr site has a history of forest use, although it has been maintained for several decades as a mowed field. Five treatments were employed at each site: (1) switchgrass only (SG); (2) pine only (Pine); (3) pine planted into switchgrass with a 1.2-m competition-free zone on either side of the row of trees (PS-4); (4) pine in switchgrass with a 0.6-m competition-free zone (PS-2); (5) and pine planted into switchgrass with no-competition free zone (PS-0). Switchgrass yields at Starr were roughly one quarter of those at Pontotoc, which led to differences in competitive intensities between species at the two sites. Loblolly pine heights at Pontotoc and Starr averaged 70 and 63 cm following year 1, and 161 and 176 cm following year 2, respectively. Planting pines directly into switchgrass significantly reduced tree heights, but seedlings in PS-4 treatments experienced growth ≥ Pine treatments. Switchgrass yields did not significantly differ by distance from tree row for any treatment. Our results suggest switchgrass dominates resource competition in early phases of this production system. However, these competitive pressures can be managed to effectively co-produce loblolly pine and switchgrass on southern lands.

INTRODUCTION

Emerging markets for biomass-based fuel sources present an opportunity for establishing and managing fast-growing and high-yielding bioenergy feedstocks on marginal or degraded lands. One common associate of North American grassland communities, switchgrass (*Panicum virgatum* L.), has been specifically identified by the U.S. Department of Energy as a model bioenergy feedstock. Characteristics making the species highly desirable include sustained high productivity across a wide spectrum of sites, relatively low demand for soil moisture and nutrients, and positive environmental benefits such as wildlife habitat and carbon sequestration (Keshwani and Cheng 2009, McLaughlin and others 1999, McLaughlin and Walsh 1998, Parrish and Fike 2005, Sanderson and others 2006).

In the southeastern United States, loblolly pine (*Pinus taeda* L.) is commonly planted on many of the marginal or degraded lands considered target areas for switchgrass production. Loblolly pine plantation yields can exceed 400 cubic feet per acre per year (Fox and others 2007), and the species dominates approximately 13.4 million acres of southern lands (Baker and Langdon 1990, Schultz 1999). While loblolly pine has customarily and still is primarily managed for pulpwood and high-value sawtimber production, expanding bioenergy markets have also raised interest in managing loblolly pine plantations for bioenergy production (Scott and Tiarks 2008).

Many landowners may lack confidence investing in relatively unfamiliar bioenergy feedstock production systems such as those involving switchgrass. One possible remedy may be to combine a well-known production system (loblolly pine) with an emerging system (switchgrass). Such a system (referred to here as co-culture) would ideally diversify investment risks and lead to greater net productivity per land area, relative to monocultures of either species. However, incorporating two high-yielding species on the same land area may spur intense interspecific competitive interactions, occurring both above and belowground, thereby hindering productivity. A better understanding of these competitive interactions is needed before loblolly pine-switchgrass co-culture management may be considered economically viable.

Specific objectives of this study were to: (1) examine competitive interactions between switchgrass and loblolly pine grown in co-culture, and (2) assess early growth rates of loblolly pine as affected by different switchgrass competition treatments.

[1]Graduate Research Assistant, Professor, and Assistant Professor, respectively, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; and Assistant Professor, Oregon State University, College of Forestry, Corvallis, OR 97331.

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METHODS

Two study sites were established in northern Mississippi. The Pontotoc site was on a Mississippi State University (MSU) Agricultural Experiment Station site approximately 75 km north of Starkville, MS. The soil on the site is primarily an Atwood silt loam (NRCS 2013). The site has a history of agricultural production and likely possesses some residual fertility. The Starr site is on the MSU John Starr Memorial Forest located approximately 20 km south of Starkville, MS. The soil on the site is primarily an Ora fine sandy loam. The site has historically contained forest cover but has been maintained as a mowed pasture for the past several decades.

Alamo switchgrass was seeded at each site in spring of 2010. At Pontotoc, site preparation and seeding consisted of disking, drill seeding, and cultipacking. The site was seeded three times in 2010 to ensure full establishment. At Starr, site preparation and seeding consisted of prescribed burning, liming, and drill seeding. Containerized loblolly pine varietal seedlings were planted at both sites in March 2011. Pine seedlings were planted into subsoiled rows, spaced 9.0 m apart with 1.5 m within row tree spacing. Measurement plots were 15 m (i.e., 10 trees) by 9 m (i.e., 4.5 m on either side of the tree row).

The study consisted of five treatments: (1) switchgrass only (SG); (2) pine only (Pine); (3) pine planted in switchgrass with a 1.2-m competition-free zone on either side of the row of trees (PS-4); (4) pine in switchgrass with a 0.6-m competition-free zone (PS-2); and (5) pine planted into switchgrass with no-competition free zone (PS-0). Applications of 2 percent glyphosate were applied as needed to exclude unwanted vegetation from competition free zones. Treatments were replicated eight times at each site.

At Pontotoc, the 2011 growing season switchgrass was harvested between January 31 and February 9, 2012, and the 2012 growing season switchgrass was harvested on November 15 and November 20, 2012. At Starr, the 2011 switchgrass growing season harvest was initiated on January 6, 2012 but was not completed until March 14, 2012 due to complications with harvesting equipment. The 2012 switchgrass harvest occurred on November 1, 2012. At each site, switchgrass was harvested and yields were determined in four 1-m strips on either side of the tree rows. At both sites, subsamples were collected for each 1-m strip, weighed fresh, dried and reweighed, and total yields were converted to a dry weight basis. Per ha dry weight yields of switchgrass were evaluated relative to proximity to the tree row (0-1, 1-2, 2-3, and 3-4 m), and for the total plot. Year 1 pine seedling heights were measured in early February 2012 at both sites, and year 2 pine seedling heights were measured in January 2013.

Differences among treatments were evaluated using analysis of variance (ANOVA). If ANOVA revealed significant main effects or significant interactions, Tukey multiple comparison tests were used for post-hoc comparisons. Statistical significance was set at a critical value of α = 0.05, and all statistical analyses were performed using Statistical Analyses Software (SAS) version 9.3 (SAS, Cary, NC).

RESULTS

Switchgrass yields differed substantially between sites in both 2011 and 2012 (table 1). In 2011, Pontotoc per ha dry weight yields for all treatments other than Pine averaged nearly 7.1 Mg ha^{-1} and ranged as high as 12.7 Mg ha^{-1}. Switchgrass yields at Starr in 2011, for treatments other than Pine, averaged only 1.6 Mg ha^{-1}, and much of the yield consisted of herbaceous plant material other than switchgrass. Pontotoc switchgrass yields in 2012 averaged 9.74 Mg ha^{-1} and ranged as high as 13.6 Mg ha^{-1}. Starr switchgrass yields averaged 2.18 Mg ha^{-1} and were still largely a scattered mixture of herbaceous grasses in 2012. At both sites and for both years, yields per m^2 of established switchgrass did not differ by treatment or distance from tree row. However, plot averages of total switchgrass yields did differ by treatment, reflecting the exclusion of switchgrass within the competition-free zones.

Tree heights at Pontotoc after the first growing season (2011) were greatest in the Pine and PS-4 treatments, averaging 84 cm (fig. 1). Heights in the PS-0 and PS-2 treatments were significantly less, averaging 46 cm and 66 cm, respectively. The seedlings at Pontotoc put on

Table 1—Switchgrass harvest yields at Pontotoc and Starr, MS

Location	Treatment	---------Yield---------- 2011	2012
		------*Mg ha*$^{-1}$------	
Pontotoc	SG	7.9	10.7
	PS-0	8.1	10.3
	PS-2	6.4	9.5
	PS-4	5.9	8.4
	Pine	2.3	---
Starr	SG	1.8	2.4
	PS-0	1.7	1.6
	PS-2	1.7	2.7
	PS-4	1.3	2.1
	Pine	1.1	---

considerable growth the following growing season (2012), and the Pine and PS-4 treatments averaged 207 cm at the end of the growing season. Tree heights for PS-2 and PS-0 treatments at Pontotoc were 137 and 91 cm following the 2012 growing season, respectively. Starr tree heights following the initial growing season (2011) averaged 63 cm and ranged from 59 to 68 cm (fig. 2). Tree heights in year 2 averaged 176 cm and were greatest in the PS-4 treatment and smallest in the PS-0 treatment. Year 2 trees at Starr were larger in the PS-4 treatment than in the Pine treatment.

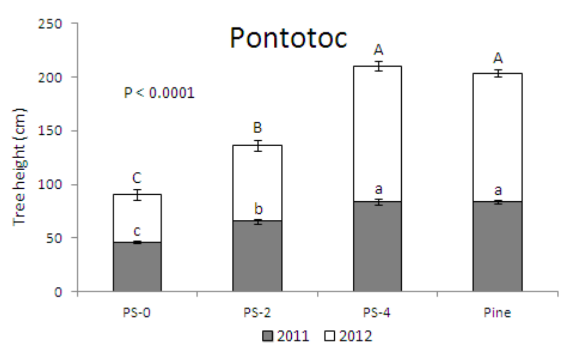

Figure 1--Tree heights at Pontotoc following the 2011 and 2012 growing seasons. Within years, differing letters indicate significant differences among treatments.

DISCUSSION

Switchgrass yields varied greatly between the two sites in northern Mississippi. At Pontotoc, switchgrass yields approached the 15 Mg ha^{-1} sustainable average reported by Parrish and Fike (2005), but Starr yields fell well below this mark. Yield discrepancies between sites were likely driven by differences in cultural techniques. Pontotoc received multiple herbicide applications in the years prior to seeding and had a previous agricultural history, which minimized weed pressure from residual seedbanks, allowing the switchgrass to rapidly establish a canopy and shade out emerging competitors. Fewer efforts to control weed pressures at Starr, combined with a well-established weed seedbank from years of pasture usage, hindered switchgrass establishment and its ability to form a dense canopy. The current lack of selective herbicides for controlling herbaceous competition without negatively impacting switchgrass presents an obstacle to the conversion of previous pasturelands to herbaceous bioenergy feedstocks (Buhler and others 1998).

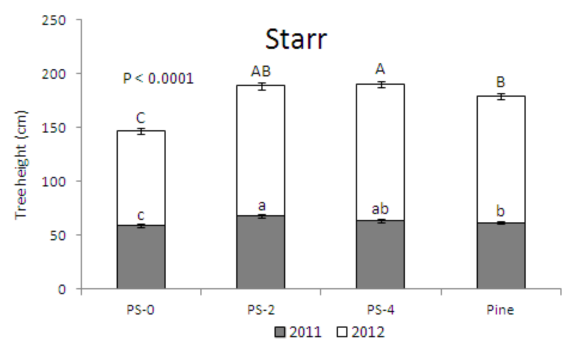

Figure 2--Tree heights at Starr following the 2011 and 2012 growing seasons. Within years, differing letters indicate significant differences among treatments.

It is well established that loblolly pine responds favorably to early competition control with increased growth (Cain 1991, Miller and others 1991, Morris and others 1993, Nilsson and Allen 2003, Zutter and Miller 1998). However, few studies have specifically examined competition dynamics within loblolly pine-switchgrass systems (Albaugh and others 2012, Blazier and others 2012). In our study, loblolly pine second year heights within the PS-4 and Pine treatments were over twice that of the PS-0 treatment and significantly greater than the PS-2 treatment at the highly competitive Pontotoc site. Competitive relationships were less drastic at Starr due to lower switchgrass yields. Interestingly, 2012 tree heights in the PS-4 treatment at Starr exceeded those of the Pine treatment, suggesting on some sites a moderate level of herbaceous competition may promote tall, slender crop trees as opposed to the undesirable lateral-branching characteristics often found when trees are open grown.

Our switchgrass biomass yields are in agreement with those of Albaugh and others (2012), who found no significant differences in switchgrass growth parameters when examined in relation to distance from loblolly pine tree rows, early in stand rotation. However, switchgrass biomass yields were significantly reduced beneath loblolly pine canopies by mid-rotation in Louisiana (Blazier and others 2012). As pine trees mature at our sites, we expect to observe significant switchgrass yield reductions in relation to distance from tree rows.

This study demonstrates that interspecific competition is a legitimate concern for those considering intercropping switchgrass between rows of loblolly pine. Early in the stand rotation, switchgrass appears to be the dominant competitive species and exerts considerable competitive pressures on loblolly pine seedlings. However, we expect loblolly pines will develop extensive root systems and dense canopies as stands mature which will increase their competitive importance. Overall, our data suggests the establishment of a competition-free zone around pine seedlings can improve resource availabilities for pines and increase tree growth to levels comparable to a pine-only system.

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LITERATURE CITED

Albaugh, J.M.; Sucre, E.B.; Leggett, Z.H. [and others]. 2012. Evaluation of intercropped switchgrass establishment under a range of experimental site preparation treatments in a forested setting on the lower coastal plain of North Carolina, U.S.A. Biomass and Bioenergy. 46: 673-682.

Baker, J.B.; Langdon O.G. 1990. *Pinus taeda* L. Loblolly pine. In: Burns, R.M; Honkala, B.H., tech. coords. Silvics of North America. Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 497-512. Vol. 1.

Blazier, M.A.; Clason, T.R.; Vance, E.D. [and others]. 2012. Loblolly pine age and density affects switchgrass growth and soil carbon in an agroforestry system. Forest Science. 58(5): 485-496.

Buhler, D.D.; Netzer, D.A.; Riemenschneider, D.E.; Hartzler, R.G. 1998. Weed management in short rotation poplar and herbaceous perennial crops grown for biofuel production. Biomass and Bioenergy. 14(4): 385-394.

Cain 1991. The influence of woody and herbaceous competition on early growth of naturally regenerated loblolly and shortleaf pines. Southern Journal of Applied Forestry. 15: 179-184.

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.

Keshwani, D.R.; Cheng, J.J. 2009. Switchgrass for bioethanol and other value-added applications: a review. Bioresource Technology. 100: 1515-1523.

McLaughlin, S.; Bouton, J.; Bransby, D. [and others]. 1999. Developing switchgrass as a bioenergy crop. In: Janick, J.J., ed. Proceedings of the 4th national new crops symposium. Alexandria, VA: ASHS Press: 282-299.

McLaughlin, S.B.; Walsh, M.E. 1998. Evaluating environmental consequences of producing herbaceous crops for bioenergy. Biomass and Bioenergy. 14(4): 317-324.

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: 169-179.

Morris, L.A.; Morris, S.A.; Garbett, W.S. 1993. Competitive interference between selected herbaceous and woody plants and *Pinus taeda* L. during two growing seasons following planting. Forest Science. 39: 166-187.

Nilsson, U.; Allen, H.L. 2003. Short- and long-term effects of site preparation, fertilization, and vegetation control on growth and stand development of planted loblolly pine. Forest Ecology and Management. 17: 367-377.

NRCS [Natural Resources Conservation Service]. 2013. Soil climate analysis network site 2089. Portland, OR: U.S. Department of Agriculture Natural Resources Conservation Service, National Water and Climate Center. http://www.wcc.nrcs.usda.gov/. [Date accessed: June 20, 2013].

Parrish, D.J.; Fike, J.H. 2005. The biology and agronomy of switchgrass for biofuels. Critical Reviews in Plant Sciences. 24: 423-459.

Sanderson, M.A.; Adler, P.R.; Boateng, A.A. [and others]. 2006. Switchgrass as a biofuels feedstock in the USA. Canadian Journal of Plant Science. 86: 1315-1325.

Scott, D.A.; Tiarks, A. 2008. Dual-cropping loblolly pine for biomass energy and conventional wood products. Southern Journal of Applied Forestry. 32: 33-37.

Shultz, R.P. 1999. Loblolly - the pine for the twenty-first century. New Forest. 17: 71-88.

Zutter, B.; Miller, J.H. 1998. Eleventh-year response of loblolly pine and competing vegetation to woody and herbaceous plant control on a Georgia flatwoods site. Southern Journal of Applied Forestry. 22(2): 88-95.

PREDICTING SMALL-DIAMETER LOBLOLLY PINE ABOVEGROUND BIOMASS IN NATURALLY REGENERATED STANDS

Kristin M. McElligott, Don C. Bragg, and Jamie L. Schuler[1]

Abstract--There is growing interest in managing southern pine forests for both carbon sequestration and bioenergy. For instance, thinning otherwise unmerchantable trees in naturally regenerated pine-dominated forests should generate biomass without conflicting with more traditional forest products. However, we lack the tools to accurately quantify the biomass in these submerchantable size classes. To help remedy this, we destructively sampled 54 small-diameter loblolly pines (*Pinus taeda* L.) from stands on the Crossett Experimental Forest (CEF) in southeastern Arkansas. After harvesting, each tree was divided into stemwood, foliage and branch, and taproot components, then oven-dried and weighed. We then fit an exponential equation based on the National Biomass Estimator (NBE) to predict aboveground oven-dry biomass as a function of diameter at breast height (d.b.h.). The resulting model fit the sample data well (pseudo-R^2 = 0.996). Comparing the CEF local biomass model with the pine submodel of the NBE suggested that the NBE would consistently underestimate small-diameter loblolly pine biomass on the CEF. While this difference appeared small, its cumulative effect could be appreciable. For example, in a young loblolly pine stand averaging 5 cm d.b.h. and 5,000 stems ha^{-1}, the NBE would predict almost 9 percent less biomass than the new CEF biomass model—a difference of nearly 2 metric tons ha^{-1}. Such locally derived equations offer silviculturists opportunities to better assess the potential of naturally regenerated pine stands to produce fiber and thus may be worth the investment of time and resources to develop.

INTRODUCTION

To better understand the potential influence southern pine forests will have on emerging cellulosic biofuel, bioenergy, and carbon markets, it is imperative that we estimate the quantity of biomass stored in these forests as accurately as possible. Naturally regenerated pine-dominated forests present unique opportunities for these potential markets because they are often overstocked when young, and thinning otherwise unmerchantable material could generate some of the biomass needed to meet biofuel or bioenergy targets without conflicting with more traditional forest products (Koch and McKenzie 1976, Westbrook and others 2007). However, in many regions we lack the tools needed to quantify the biomass in these submerchantable size classes and are therefore unable to accurately estimate the fiber production, total biomass, or carbon in forests (Chaturvedi and Raghubanshi 2013, McGarrigle and others 2011).

Stand-level biomass is typically estimated from an allometric equation that predicts oven-dry biomass for individual stems based on diameter at breast height (d.b.h.) and then summed to yield biomass per unit land surface area (Whittaker and Woodwell 1968). Because of the growing need for landowners to quantify tree biomass, managers often apply equations from other regions, stand conditions, and species than those actually found on the site. Research has repeatedly shown that such application is less than ideal (Bragg 2011, Chave and others 2005, Crow and Schlaegel 1988, Melson and others 2011, Parresol 1999, Payadeh 1981, Ruark and others 1987, Zianis and others 2005) because model choice and implementation can dramatically impact estimates, and errors in biomass estimation can accumulate when used incorrectly [for instance, if applied to dissimilar species or extrapolated beyond the original diameter range for which the model was derived (Chave and others 2005, Fonseca and others 2012, Parresol 1999)].

Small-diameter trees constitute a significant proportion of naturally regenerated forests in southeastern Arkansas. While numerous models capable of predicting aboveground live-tree biomass for southern pines have been developed (Jenkins and others 2003, Newbold and others 2001, Van Lear and others 1986), few of these were actually derived using Arkansas forests and fewer still specifically included small stems. Therefore, to more accurately predict biomass for small-diameter loblolly pines, we developed a site- and species-specific biomass equation for the USDA Forest Service's Crossett Experimental Forest (CEF). To evaluate the predictive ability of this locally fit

[1]Program Technician, Arkansas Forest Resources Center, Monticello, AR 71656; Research Forester, USDA Forest Service, Southern Research Station, Monticello, AR 71656; and Assistant Professor, West Virginia University, Division of Forestry and Natural Resources, Morgantown, WV 26506.

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model, we compared our predictions with the commonly used National Biomass Estimator (NBE; Jenkins and others 2003). Additionally, we also contrasted cumulative (per ha) biomass predictions using the local model and the NBE to determine the influence of model choice on stand-level biomass estimates.

MATERIALS AND METHODS

Study Site

This study was conducted on the CEF, which is located 11 km south of the city of Crossett in Ashley County, AR. Established in 1934 by the Forest Service, the CEF covers nearly 680 ha in southeastern Arkansas and is dominated by naturally regenerated forests of loblolly (*Pinus taeda* L.) and shortleaf (*Pinus echinata* Mill.) pine, with a minor hardwood component. The relatively flat, rolling terrain of the CEF varies between 36 and 48 m above sea level, with local differences rarely exceeding 3 m. The soils of the CEF are primarily silt loams with a loblolly pine site index of 25 to 30 m (50 year base age) (Gill and others 1979).

Sample Tree Selection and Measurement

Small-diameter (< 15 cm d.b.h.) loblolly pines were destructively sampled across the naturally regenerated pine-dominated forests of the CEF. Trees of this size were chosen to address logistical issues related to collecting and weighing above- and belowground biomass of large specimens. In addition, the smallest trees from this diameter range (those < 10 cm d.b.h.) are often not sampled when developing biomass equations (Snowdon and others 2000), yet may compose a significant fraction of many forests, including those on the CEF.

Model development entailed destructively sampling these pines. Smaller sample trees were pulled directly from the soil using a small tractor with a hydraulic boom extension lift. Bigger pine trees that could not be lifted from the ground were partially excavated using a backhoe attachment for the tractor, and then pulled. Once out of the ground, pines were separated into aboveground (foliage + branch and stemwood), and belowground (taproot) components. For this study, only the aboveground components were modeled. All components were dried in an air-forced oven at 90^{o} C to a constant weight, and the stem, branch, and foliage components then summed to produce aboveground, oven-dry biomass.

Model Comparison

Aboveground live-tree oven-dry biomass (B_D, in kilograms) values were then fit to the CEF model (based on the NBE) using ordinary least squares regression:

$$B_D = b_0 + e^{b_1 + b_2(ln(d.b.h.))} \qquad (1)$$

where b_i are fitted coefficients. The slightly different NBE also predicts B_D:

$$B_D = e^{-2.5356 + 2.4349(ln(d.b.h.))} \qquad (2)$$

The NBE is a conservative and well-documented national model and is generally considered the standard biomass equation used nationwide by researchers and agencies to estimate tree- and stand-level forest biomass, including the official greenhouse gas inventories for the United States (EPA 2008).

RESULTS

Individual Tree Biomass

We sampled 54 loblolly pines from 0.9- to 15.0-cm d.b.h. (average of 4.6 cm; standard deviation of 3.6 cm). After processing and drying, the measured B_D for these trees ranged from 0.23 to 60.87 kg, averaging 7.19 kg (standard deviation = 12.77 kg).

Local Model Fit

The following CEF biomass model:

$$B_D = 0.174544 + e^{-2.4571 + 2.41911(ln(d.b.h.))} \qquad (3)$$

fit the data well (pseudo-R^2 = 0.996). Not surprisingly, a local equation using a single species did a better job of fitting loblolly pine data from the CEF than the more general NBE (fig. 1). For the size range we considered, the NBE had few prominent departures but consistently underestimated B_D across the sampled d.b.h. range. Because of the scale of figure 1, this propensity was not readily apparent. To better demonstrate the differences between the actual B_D data and both sets of model predictions from the 0- to 15-cm d.b.h. range, we have enlarged that section of sampled data and model predictions for pines up to 5-cm d.b.h. (fig. 2).

The most noticeable difference appeared to be in the smallest of the trees (those < 3 cm d.b.h.), for which the NBE underestimated B_D at a

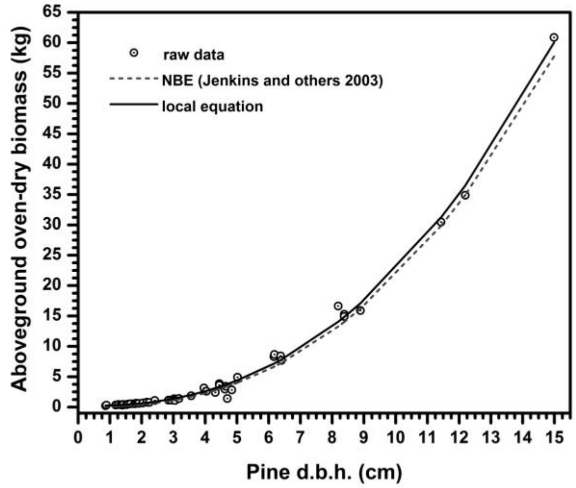

Figure 1--Observed and predicted aboveground live-tree oven-dry biomass as a function of stem diameter for small (up to 15 cm d.b.h.) loblolly pine from the Crossett Experimental Forest.

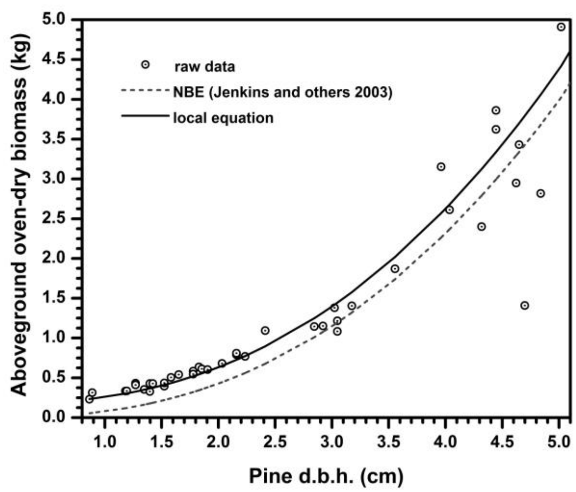

Figure 2--Observed and predicted aboveground live-tree oven-dry biomass as a function of stem diameter for the smallest (0.9- to 5.0-cm d.b.h.) loblolly pines from the Crossett Experimental Forest sampled for this study.

proportionally higher rate with decreasing diameter (table 1). As an example, for a loblolly pine 5 cm in d.b.h., the CEF model forecast an individual stem biomass of 4.4 kg while the NBE predicted 4.0 kg, a relative difference of 9 percent. However, at the smallest size class measured (1.0 cm), the CEF model predicted an individual stem biomass of 0.26 kg and the NBE predicted 0.08 kg, or 70 percent less biomass.

Table 1—Predicted aboveground live oven-dry biomass (B_D) for 1.0-, 2.5-, 5.0-, and 15.0-cm d.b.h. loblolly pines

	----------Modeled B_D----------		
D.b.h.	NBE	Local CEF	Relative difference[a]
--cm--	*----------------kg--------------*		*percent*
1.0	0.08	0.26	70
2.5	0.74	0.96	23
5.0	3.99	4.38	9
10.0	21.56	22.67	5
15.0	57.87	60.15	4

[a]Relative B_D difference = [(NBE B_D – local B_D) / local B_D] × 100.

DISCUSSION

Small-diameter trees can make up a large fraction of biomass in southern pine forests, making the accurate prediction of this component important. A significant initial step in improving the reliability of predictive models is understanding the role of model choice on the quantification of biomass. While this rapidly advancing field has witnessed considerable improvement over the years, many landowners have few other choices than to apply models developed for more general conditions, as is the case with perhaps the most commonly applied design, the NBE. In many instances, the use of generalized forms such as the NBE can be as effective as more site- and species-specific equations applied to individual trees in a stand (Snowdon and others 2000). However, there is also evidence that biomass models developed for other regions or silvicultural origins will yield notably different predictions from those of locally derived equations (Bragg 2011) because of considerable geographic variability in the growth and yield of most tree species (especially loblolly pine) as a function of genetics, site conditions, growth rate, and other factors (Jordan and others 2008, Mitchell and Wheeler 1959, Schultz 1997).

It is therefore quite possible that the use of a general national-scale allometric equation may underestimate the biomass of some species and overestimate the biomass of others, resulting in errors in calculating the biomass of the forest as

Table 2—Per ha differences in stand-level predicted pine-only aboveground biomass (B_D) and potential carbon dioxide equivalent (CO_2e) revenue, assuming the given stocking at the specified average tree d.b.h.

			-----Modeled B_D-----		---Modeled CO_2e---		-------Potential CO_2e revenues------		
D.b.h.	Stocking	Basal area	NBE	Local CEF	NBE	Local CEF	NBE	Local CEF	Difference
--cm--		*--m²--*	*metric tons*				*dollars*		
1.0	5,000	0.4	0.4	1.3	0.7	2.4	9.90	32.52	22.62
2.5	5,000	2.5	3.7	4.8	6.8	8.8	92.16	120.06	27.91
5.0	5,000	9.8	19.9	21.9	36.6	40.2	498.32	547.30	48.98
10.0	3,000	23.6	64.7	68.0	118.7	124.8	1,616.73	1,699.40	82.68
15.0	2,500	44.2	144.7	150.4	265.5	275.9	3,615.94	3,758.41	142.48

[a]$CO_2e = B_D \times 0.5 \times 3.67$
[b]Potential revenues assumes $13.62 per ton of CO_2e, as experienced in February 2013 California Air Resources Board allowance auction (California ARB 2013).

a whole (Zhou and Hemstrom 2009). The underestimates of loblolly pine from the CEF predicted by the NBE almost certainly arose from how the pine submodel of the NBE was derived. The NBE pine equation is not specific to loblolly pine; rather, it was generated from "pseudodata" produced by 43 different equations using 14 different species of *Pinus* ranging from eastern white pine (*Pinus strobus* L., specific gravity = 0.34) to longleaf pine (*Pinus palustris* Mill., specific gravity = 0.54) (Jenkins and others 2003, 2004). Because of this generality, and the fact that southern pine species such as loblolly and shortleaf (specific gravities = 0.47) often have significantly denser wood than most other pines (Miles and Smith 2009), it is not surprising the NBE under-predict loblolly pine sampled on the CEF. These results demonstrate the importance of locally derived models, especially when biomass predictions are extrapolated to stands.

Even though the absolute differences between the CEF local biomass model (equation 3) and the NBE (equation 2) are not dramatic for individual loblolly pines in this sample range (table 1), subtle and consistent biases for individual trees can have a major cumulative impact when scaled up to stand- or regional-scale estimates. For example, in a very young loblolly pine forest averaging 5,000 stems ha^{-1} and 1.0 cm in d.b.h., the CEF biomass model predicted biomass of over three times the amount forecast by the NBE (1.3 versus 0.4 tons ha^{-1}). In a thinned pine stand with 2,500 stems ha^{-1} averaging 15.0 cm in d.b.h., the NBE predicted B_D of 144.7 tons ha^{-1} of biomass, while the CEF model yielded 150.4 tons, a difference of 5.7 tons (table 2).

Such a disparity may not be noticed if the biomass products are weight-scaled at a mill, but those purchased in terms of standing stocks may be affected greatly. Sequestered carbon, for instance, can be traded (sold) in the form of carbon dioxide equivalents (CO_2e) stored in trees based on modeled values. Converting the biomass predictions of both the NBE and local CEF models to CO_2e, and then assuming the February 2013 price ($13.62 per ton of CO_2e) of carbon allowances on the California Air Resources Board market (California ARB 2013) makes this point clearly. For the 15-cm d.b.h. scenario, this difference amounts to $142.48 ha^{-1} difference between the model outcomes (table 2). A consistent disparity simply from using a different model, with no change to tree size or stocking, has considerable economic and silvicultural implications. Whether in terms of more traditional forest products (chips, pulpwood, or even hog fuel) or the new currency of sequestered carbon, failure to capture the actual value of the biomass on a given parcel is detrimental to the landowner and may also misrepresent the carbon stored by a particular treatment, especially when a large number of small-diameter stems are involved.

CONCLUSIONS

Expanding markets for biomass in energy production improves the silvicultural opportunities for balancing sawtimber and woody biomass by providing much-needed small-diameter and low-grade markets (Koning and Skog 1987). Southern forests provide an abundance of underutilized wood, such as logging residues and small-diameter timber. Using this material for bioenergy markets could alleviate the competition with traditional timber commodities while providing multiple forest management options for increased growth, improved forest health, and supplemental income (Munsell and Germain 2007). However, accurate biomass estimations from stem to stand scale and across a range of size classes are needed to better realize these opportunities in emerging woody bioenergy markets and carbon accounting procedures being implemented by various commercial enterprises and regulatory agencies (for example, California ARB 2009). The determination of biomass quantities in the face of carbon-driven forest management may also impact forest policy (Galik and others 2013).

While general models like the NBE make the simulation process easier and yield reasonable biomass estimates for many species with little to no existing biomass information, it can obscure significant differences resulting in potentially major consequences for forest managers and landowners. The availability of locally derived equations, such as our model for small-diameter loblolly pine on the CEF, offers silviculturists a means to better assess the potential of naturally regenerated pine stands to produce fiber, and thus may be worth the investment of time and resources to develop.

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LITERATURE CITED

Bragg, D.C. 2011. Modeling loblolly pine above-ground live biomass in a mature pine-hardwood stand: a cautionary tale. Journal of the Arkansas Academy of Science. 65: 31-38.

California ARB [Air Resources Board]. 2009. California's 1990-2004 greenhouse gas emissions inventory and 1990 emissions level. California Environmental Protection Agency, Air Resources Board technical support document. http://www.arb.ca.gov/cc/inventory/doc/methods_v1/ghg_inventory_technical_support_document.pdf. [Date accessed: May 8, 2013].

California ARB [Air Resources Board]. 2013. Quarterly auction 2. http://www.arb.ca.gov/cc/capandtrade/auction/february_2013/auction2_feb2013_summary_results_report.pdf. [Date accessed: May 7, 2013].

Chaturvedi, R.K.; Raghubanshi, A.S. 2013. Aboveground biomass estimation of small diameter woody species of tropical dry forest. New Forests. 44(4): 509-519.

Chave, J.; Andalo, C.; Brown, S. [and others]. 2005. Tree allometry and improved estimation of carbon stocks and balance in tropical forests. Oecologia. 145: 87-99.

Crow, T.R; Schlaegel, B.E. 1988. A guide to using regression equations for estimating tree biomass. Northern Journal of Applied Forestry. 5: 15-22.

EPA [U.S. Environmental Protection Agency]. 2008. Inventory of U.S. greenhouse gas emissions and sinks: 1990–2006. EPA 430-R-08-005. Washington, DC: U.S. Environmental Protection Agency Office of Atmospheric Programs. [Not paged].

Fonseca, W.; Alice, F.E.; Rey-Benayas, J.M. 2012. Carbon accumulation in aboveground and belowground biomass and soil of different age native forest plantations in the humid tropical lowlands of Costa Rica. New Forests. 43: 197-211

Galik, C.S.; Murray, B.C.; Mercer, D.E. 2013. Where is the carbon? Carbon sequestration potential from private forestland in the southern United States. Journal of Forestry. 111: 17-25.

Gill, H.V.; Avery, D.C.; Larance, F.C.; Fultz, C.L. 1979. Soil survey of Ashley County, Arkansas. Washington, DC: U.S. Department of Agriculture Soil Conservation Service. 92 p. In cooperation with: the U.S. Department of Agriculture Forest Service and the Arkansas Agricultural Experiment Station.

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.

Jenkins, J.C.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A. 2004. Comprehensive database of diameter-based biomass regressions for North American tree species. Gen. Tech. Rep. NE-319. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station. 45 p.

Jordan, L.; Clark, A.; Schimleck, L.R. [and others]. 2008. Regional variation in wood specific gravity of planted loblolly pine in the United States. Canadian Journal of Forest Research. 38: 698- 710.

Koch, P.; McKenzie, D.W. 1976. Machine to harvest slash, brush, and thinnings for fuel and fiber - a concept. Journal of Forestry. 74: 809-812.

Koning, J.W.; Skog, K.E. 1987. Use of wood energy in the United States - an opportunity. Biomass. 12: 27–36.

McGarrigle, E.; Kershaw, J.A.; Lavigne, M.B. [and others]. 2011. Predicting the number of trees in small diameter classes using predictions from a two-parameter Weibull distribution. Forestry. 84: 431-439.

Melson, S.L.; Harmon, M.E.; Fried, J.S.; Domingo, J.B. 2011. Estimates of live-tree carbon stores in the Pacific Northwest are sensitive to model selection. Carbon Balance and Management. 6:2. 16 p.

Miles, P.D.; Smith, W.B. 2009. Specific gravity and other properties of wood and bark for 156 tree species found in North America. Res. Note NRS-38. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 35 p.

Mitchell, H.L.; Wheeler, P.R. 1959. Highlights of Mississippi survey: the search for wood quality. Forest Farmer. 18: 4-6.

Munsell, J.F.; Germain, R.H. 2007. Woody biomass energy: an opportunity for silviculture on nonindustrial private forestlands in New York. Journal of Forestry. 105: 398-402.

Newbold, R.A.; Baldwin, V.C., Jr.; Hill, G. 2001. Weight and volume determination for planted loblolly pine in north Louisiana. Res. Pap. SRS-26. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 19 p.

Parresol, B. 1999. Assessing tree and stand biomass: a review with examples and critical comparisons. Forest Science. 45: 573-593.

Payadeh, B. 1981. Choosing regression models for biomass prediction equations. Forestry Chronicle. 57: 229-232.

Ruark, G.A.; Martin, G.L.; Bockheim, J.G. 1987. Comparison of constant and variable allometric ratios for estimating *Populus tremuloides* biomass. Forest Science. 33: 294-300.

Schultz, R.P. 1997. Loblolly pine: the ecology and culture of loblolly pine (*Pinus taeda* L.). Agric. Handb. 713. Washington, DC: U.S. Department of Agriculture. 493 p.

Snowdon, P.; Eamus, D.; Gibbons, P. [and others]. 2000. Synthesis of allometrics, review of root biomass and design of future woody biomass sampling strategies. National carbon accounting system Tech. Rep. 17. Canberra, Australia: Australian Greenhouse Office. 113 p.

Van Lear, D.H.; Taras, M.A.; Waide, J.B.; Augspurger, M.K. 1986. Comparison of biomass equations for planted vs. natural loblolly pine stands of sawtimber size. Forest Ecology and Management. 14: 205-210.

Westbrook, M.D.; Greene, W.D.; Izlar, R.L. 2007. Utilizing forest biomass by adding a small chipper to a tree-length southern pine harvesting operation. Southern Journal of Applied Forestry. 31: 165-169.

Whittaker, R.H.; Woodwell, G.M. 1968. Dimension and production relations of trees and shrubs in the Brookhaven Forest, New York. Journal of Ecology. 56: 1-25.

Zhou, X.; Hemstrom, M.A. 2009. Estimating aboveground tree biomass on forest land in the Pacific Northwest: a comparison of approaches. Res. Pap. PNW-RP-584. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 18 p.

Zianis, D.; Muukkonen, P.; Mäkipää, R.; Mencuccini, M. 2005. Biomass and stem volume equations for tree species in Europe. Silva Fennica Monographs. 4. 63 p.

ASSESSMENT OF FOREST MANAGEMENT INFLUENCES ON TOTAL LIVE ABOVEGROUND TREE BIOMASS IN WILLIAM B. BANKHEAD NATIONAL FOREST, ALABAMA

Callie Schweitzer, Dawn Lemke, Wubishet Tadesse, and Yong Wang[1]

Abstract-- Forests contain a large amount of carbon (C) stored as tree biomass (above and below ground), detritus, and soil organic material. The aboveground tree biomass is the most rapid change component in this forest C pool. Thus, management of forest resources can influence the net C exchange with the atmosphere by changing the amount of C stored, particularly in landscapes dominated by forests, such as in the southeastern United States. Our work focuses on the influence of prescribed burning and thinning on total live aboveground tree (TLAT) biomass in the William B. Bankhead National Forest, Alabama. We implemented a large-scale study that involved a factorial arrangement of three levels of thinning (heavy thin to 11 m^2 ha^{-1} basal area; light thin to 15 m^2 ha^{-1} basal area; and no thin) and three prescribed fire intervals (no fire, 3-year return, 9-year return). Biomass was assessed among treatments using allometric equations related to tree species and diameter. Pre-treatment stands ranged from 117 to 137 Mg ha^{-1} TLAT biomass. Overall burning showed no significant influence on TLAT biomass. All but one treatment (light thin, no burn) had a higher rate of TLAT biomass gain post-treatment than the control. Control had an average yearly TLAT biomass gain of 4 percent per year, with the thinned treatments having averages ranging from 5 percent to 7 percent per year. Our results provided a first step for reliable and accurate measurement of biomass potential, which is increasingly important, particularly for sustainable forest management, monitoring global climate change, and forest productivity.

INTRODUCTION

Over the last 30 years, carbon dioxide (CO_2) emissions from the use of fossil fuels has grown at an average rate of 1.9 percent per year (Nabuurs and others 2007). Mitigation of atmospheric CO_2 requires an approach that combines CO_2 emission reductions with increased CO_2 storage (Birdsey and others 2006, D'Amato and others 2011, Malmsheimer and others 2008). Forests contain a large amount of carbon (C) stored as tree biomass (above and below ground), detritus, and soil organic material (Fahey and others 2010) and as such have the potential to play a crucial role in the mitigation of atmospheric CO_2 through increased C storage (D'Amato and others 2011, Nabuurs and others 2007). Areas of deforestation, such as tropical rainforests, can be large sources of C (Canadell and Raupach 2008), and areas of growing forest can be large C sinks. It has been estimated that forest ecosystems contain approximately half of the total terrestrial C pool (Dixon and others 1994) and, at a global scale, forests sequester 1.3 to 4.2 $GtCO_2$-equivalents (1.3 to 4.3 billion tonnes) per year (Nabuurs and others 2007). Currently in the United States, forests sequester enough C each year to offset 10 percent of annual emissions from fossil fuels (Birdsey and others 2006).

In the southeastern United States, forests make up over 60 percent of the land area (Wear and Greis 2012). The most rapid component of forest change in this C pool is the aboveground tree biomass (Fahey and others 2010). Thus, management of forest resources can influence net C exchange with the atmosphere by changing the amount of C stored (Canadell and Raupach 2008, Malmsheimer and others 2008). It has been suggested that net C sequestration can theoretically be maximized by maintaining the landscape at a maximal stage of net ecosystem productivity (Fahey and others 2010), and that forest management targets both mitigation (using the forest to sequester C) and adaptation (increasing forest health and resiliency) (Malmsheimer and others 2008). Changing species composition, rotation length, fire, harvest management practices, and other

[1]Research Forester, USDA Forest Service, Southern Research Station, Huntsville, AL 35801; and Research Associate, Associate Professor, and Professor, respectively, Alabama A&M University, Department of Natural Resources and Environmental Science, Normal, AL 35762.

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biotic and abiotic disturbances can have important impacts on biomass, C stocks, and fluxes (Malmsheimer and others 2008).

This study assessed the influence of prescribed burning and thinning practices on biomass change over a 3-year period within the William B. Bankhead National Forest, in the southeastern United States. We used biomass as a surrogate for C storage; biomass estimates can be converted to C estimates using a factor of 50 percent C (Brown and others 1986, Hall and Uhlig 1991, Marland and Schlamadinger 1997).

METHODS

Study Area

The study was implemented in the William B. Bankhead National Forest (BNF; fig. 1) as part of a broader collaborative effort to experimentally test the ecosystem responses of the conversion of predominantly pine stands to upland hardwood forest cover type. The BNF was established by proclamation in 1914 and has a long history of repeated logging and of soil erosion caused by poor farming practices during the Depression era. The 73 000-ha BNF is in the Strongly Dissected Plateau sub-region of the Southern Cumberland Plateau, within the southern Appalachian Highlands (Smalley 1979). Study stands are located on slightly undulating tabletop sites, and stands are non-managed loblolly pine (*Pinus taeda* L.) plantings established 25 to 45 years ago and with substantial hardwood encroachment. Under the current management plan, much of the area is under restoration to a hardwood-dominated system (USDA Forest Service 2003). Base age 50 site indices for loblolly pine, red oaks [northern red oak (*Quercus rubra* L.), black oak (*Q. velutina* Lam.), scarlet oak (*Q. coccinea* Münchh.), and southern red oak (*Q. falcata* Michx.)], and white oaks [white oak (*Q. alba* L.) and chestnut oak (*Q. prinus* L.)] are 23-, 20-, and 20-m, respectively (Smalley 1979).

Within the BNF, a randomized complete block design with a 3 x 3 factorial treatment arrangement and four replications of each treatment was used to assess the impact of burning and thinning. The treatments were three thinning types (heavy thin 11.5 m^2 ha^{-1} residual basal area, light thin 17.2 m^2 ha^{-1} residual basal area, and an unthinned control) with three fire frequencies (burns every 3 years, burns every 9 years and an unburned control) (table 1). Each treatment is replicated four times for a total of 36 stands. Treatments were representative of management practices described in the BNF's Forest Health Restoration Plan for restoring upland oak-dominated forests and woodlands (USDA Forest Service 2003).

Field Methods

We established five, 0.08-ha vegetation measurement plots in each stand. All plot centers were permanently marked with rebar and flagging, and GPS coordinates were recorded. All trees ≥14.2 cm in diameter at breast height (d.b.h.) were permanently marked with aluminium tags, identified to species, and d.b.h. was measured. Stand selection and data collection began in the summer of 2004 and, to date, three vegetation measurements at each treatment stand were taken: pre-treatment, immediately post-treatment, and 3 years post-treatment. Frequently burned stands had received two burns, and infrequently burned stands received one burn; all burns are dormant-season burns, occurring between January and March.

Data Analysis

We used total live aboveground tree biomass (TLAT biomass) as the total aboveground biomass, calculated using allometric equations (Jenkins and others 2004). The TLAT is defined as the aboveground mass of wood and bark in live trees ≥ 14.2 cm d.b.h. from the ground to the tip of the tree, excluding all foliage (leaves, needles, buds, fruit, and limbs < 13 mm in diameter). TLAT biomass is expressed as oven-dry mass, and the unit is kg $tree^{-1}$. Equations of individual tree TLAT biomass have been developed for most tree species or species groups in the United States (see for example Jenkins and others 2004). The TLAT biomass was calculated for each tree using Jenkins and

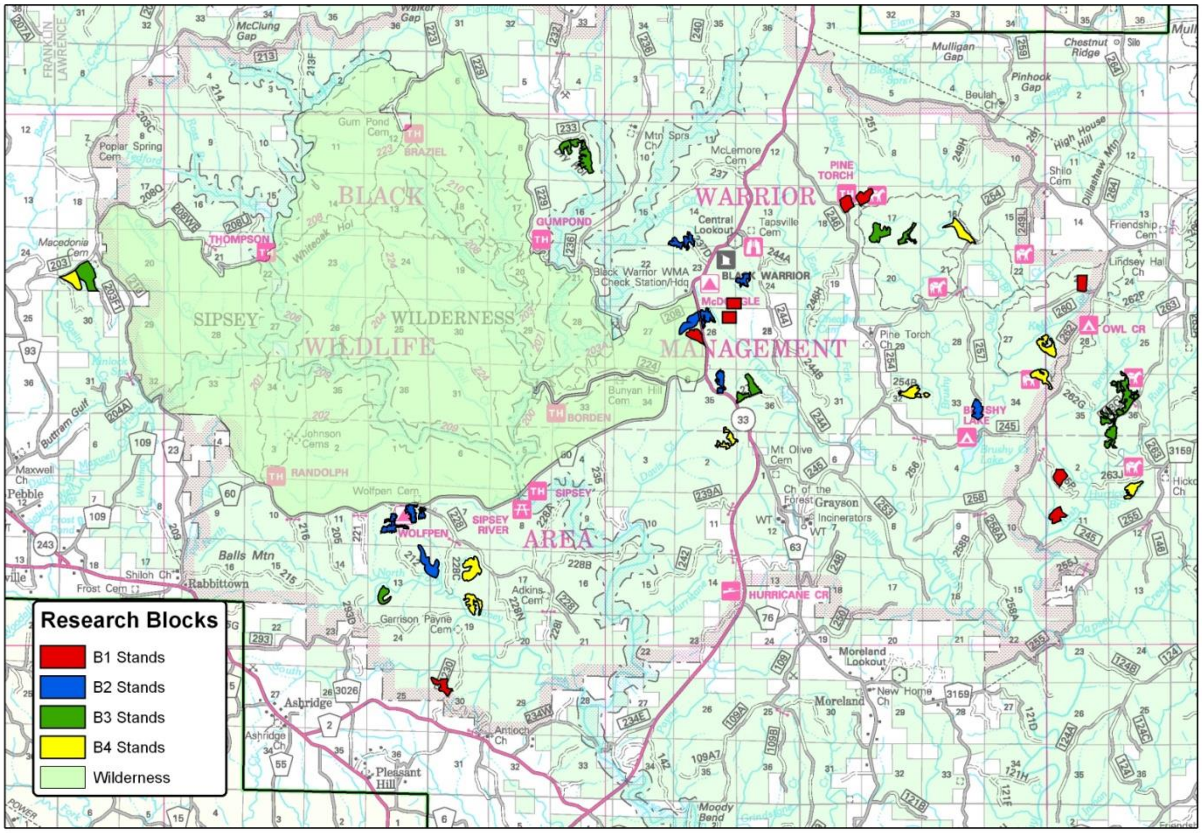

Figure 1--Map of the William B. Bankhead National Forest in north-central Alabama with depiction of study stand locations.

Table 1--Management treatments for silvicultural research on the William B. Bankhead National Forest, Alabama

Treatment number	Treatment
1	Control (no thin, no burn)
2	No thin, infrequent burn (9 years)
3	No thin, frequent burn (3 years)
4	Heavy thin (11.5 m^2/ha residual stand density), no burn
5	Light thin (17.2 m^2/ha residual stand density), no burn
6	Heavy thin, frequent burn
7	Light thin, frequent burn
8	Heavy thin, infrequent burn
9	Light thin, infrequent burn

others (2004) equations with Mg ha^{-1} calculated for each plot and stand for all species combined, all pine, and all hardwoods. We assessed productivity by evaluating the proportion gain in TLAT biomass across treatments.

Analyses of variance was used to assess the nine treatments pre-treatment, treatment against post-treatment, gain between immediate post-treatment and 3 years after post- treatment, and proportional gain between immediate post-treatment and 3 years after post-treatment. Pairwise comparisons using Tukey's contrasts were applied when appropriate. These analyses were performed using data sets containing: all trees, pines only, and hardwoods only. This included assessment of pre-treatment differences and productivity differences post-treatment.

RESULTS AND DISCUSSION

We permanently marked and measured 10,172 trees for this study. Pre-treatment basal area (BA in m^2 ha^{-1}) in the study stands ranged from 27.9 to 31.8 [standard deviation (std) 4.8 to 6.2], and stems ha^{-1} (SPH) ranged from 1,155 to 1,386 (std 227 to 349) (Schweitzer and Wang 2013). We found no differences for BA [$F = 1.25$ (8, 24), $p = 0.88$] and SPH [$F = 0.82$ (8, 24), $p = 0.86$] among the 9 treatments prior to treatment implementation (table 2). Average tree TLAT biomass pre-treatment ranged from 150 to 206 kg with std between 110 and 180. We tallied 23 dominant or co-dominant tree species. Loblolly pine (838 SPH) and Virginia pine (*P. virginiana* Mill.) (240 SPH) were the most prevalent species pre-treatment, and yellow poplar (*Liriodendron tulipifera* L.) (46 SPH) and chestnut oak (37 SPH) were the most common hardwoods. The thinning treatments resulted in three different levels of residual BA and SPH. Unthinned stands had a BA of 30.2 m^2 ha^{-1} and 1,243 SPH; the light thinned had 15.6 m^2 ha^{-1} BA and 490 SPH; and the heavy thinned stands had a BA of 11.5 m^2 ha^{-1} and 372 SPH.

TLAT Biomass by Treatments

There was no difference in TLAT biomass among stands assigned to different treatments [$F = 0.64$ (8, 24), $p = 0.76$] or blocks [$F = 1.19$ (3, 24), $p = 0.34$] before the treatments were applied. As expected, the overall TLAT biomass decreased immediately post-treatment for all the thinned treatments, with an increase in biomass 3 years post-treatment (table 2). Three years after treatment, the total increase in TLAT biomass was different among treatments [$F = 32.43$ (8, 24), $p < 0.01$]. Pairwise comparison showed control and burn only treatments different from all thinning and thinning/burn treatments (table 2).

We found an effect of treatment on productivity [$F = 5.96$ (8, 24), $p < 0.001$]. Control productivity was different from all thinning treatments apart from the light thin with no burn. No thin, infrequent burn was only different from one treatment, the heavy thin, no burn (table 2). The thinning treatment resulted in the higher biomass productivity (table 2), but there was no difference between heavy thin and light thin. Treatments 1 and 2 had the lowest productivity among all treatment types.

Pine TLAT Biomass

The pine TLAT biomass pre-treatment ranged from 98 to 116 Mg ha^{-1}, with between 258 SPH and 933 SPH in each treatment stand. There was no difference in treatments [$F = 0.48$ (8, 24), $p = 0.86$] pre-treatment (table 3). There was an increase in TLAT biomass for the control (treatment 1) and the burn only (treatments 2 and 3) treatments for both post-treatment measurements. As expected, the overall TLAT biomass declined post-treatment for all the thinned treatments, with substantial increase 3

Table 2--Mean of basal area, stem density, and total live aboveground tree biomass by treatment for three measurement cycles, change in biomass and percent productivity. Column values with the same letters are not significantly different at 0.05

	----------Pre-treatment----------			-------Immediate post-treatment-------			-------3-years post-treatment-------			Change in biomass	Percent produc-tivity
Treatment	BA	Density	Biomass	BA	Density	Biomass	BA	Density	Biomass		
	m^2/ ha	stems/ha	Mg/ha	m^2/ha	stems/ha	Mg/ha	m^2/ha	stems/ha	Mg/ha	Mg/ha	%
Control (1)	30.2 (4.8)	1,155 (235)	136 (17.3)	31.6a (4.8)	1,150b (235)	144a (15.6)	33.8a (4.7)	1,172b (227)	157a (17.0)	13.1ab	9.2a
No thin, Infrequent bum (2)	28.2 (6.1)	1,225 (283)	124 (11.3)	29.2a (6.7)	1198b (301)	134a (7.8)	31.4a (7.5)	1,207b (340)	148a (10.9)	16.4ac	12.5ab
No thin, frequent bum (3)	27.9 (6.2)	1,386 (331)	119 (23.7)	29.9a (6.2)	1381a (331)	128a (22.1)	33.5a (5.8)	1,447a (305)	147a (19.3)	18.3a	15.1ac
Heavy thin, no bum (4)	30.2 (5.2)	1,290 (331)	133 (10.2)	11.6cd (2.3)	375c (105)	57b (8.5)	13.6cd (2.5)	392c (96)	69b (10.6)	12.0ab	21.5c
Light thin, no bum (5)	30.5 (5.4)	1,190 (349)	139 (15.4)	15.5bc (4.3)	462c (161)	77b (13.7)	17.3bc (4.5)	471c (157)	88b (14.3)	10.8bc	15.3ac
Heavy thin, frequent bum (6)	29.3 (6.0)	1,211 (266)	128 (8.7)	11.5cd (2.1)	366c (83)	54b (7.0)	13.0d (2.2)	375c (83)	63b (8.0)	9.4b	17.8bc
Light thin, frequent bum (7)	30.3 (5.9)	1,277 (270)	130 (18.4)	14.8bcd (2.7)	471c (78)	68b (8.5)	17.2bcd (3.0)	484c (83)	81b (10.5)	13.7ab	20.5bc
Heavy thin, infrequent bum (8)	30.3 (5.1)	1,307 (227)	131 (20.7)	11.4d (2.2)	375c (65)	54b (11.1)	13.1cd (2.2)	379c (83)	64b (10.9)	10.3bc	20.1bc
Light thin, infrequent bum (9)	31.8 (5.7)	1,329 (248)	139 (20.6)	16.4b (3.8)	536c (65)	78b (16.1)	19.0b (4.2)	558c (74)	92b (19.9)	13.9ab	18.0bc
Treatments (p value)	0.41	0.17	0.76	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Treatments (F value)	1.03	1.47	0.61	85.77	101.09	32.15	89.67	103.43	32.43	6.69	5.96

Table 3--Biomass for all pines and all hardwoods separate, for pre-treatment and 3-years post-treatment

	Average pre-treatment biomass	Average 3 year post-treatment biomass	Treatments, pre-treatment	Block, pre-treatment	Treatment effect on productivity	Average productivity
	-----------------*Mg/ha*--------------------		--------------------------*p value*---------------------------			*%*
Pines	108	79	0.86	<0.01*	<0.01*	14
Hardwoods	23	22	0.96	0.02*	0.18	19

[a]Asterisk designates significant difference among treatments at 0.05

years post-treatment. There was a difference between treatments for productivity after treatment [F = 3.52 (8, 24), $p < 0.01$]. Productivity for the control (treatment 1) was lower than for heavy thin no burn (treatment 4), light thin frequent (treatment 7), and heavy thin infrequent burn (treatments 8), with the thinning having higher productivity.

Hardwood TLAT Biomass

Hardwood TLAT biomass pre-treatment ranged from 17 to 28 Mg ha^{-1}, with between 16 and 250 SPH in each stand. There was no difference in treatments [F = 0.29 (8, 24), p = 0.96] pre-treatment (table 3). There was an increase in TLAT biomass for the control (treatment 1) and the burn only (treatments 2 and 3) treatments for both post-treatment measurements. As expected, the overall TLAT biomass declined post-treatment for all thinned treatments, with substantial increase 3 years post-treatment. However, there was no difference between treatment productivity after treatment [F = 1.6 (8, 24), p = 0.18] (table 3). The percent change in biomass between post-treatment and 3 year post-treatment was higher than that for pines, ranging from 12 to 41 percent, compared with 7 to 19 percent for pines (table 3). There is more variation in productivity across the thinned stands compared to the unthinned stands.

Silviculture Studies for Biomass

This analysis was undertaken to capitalize on a large-scale study of management techniques applied to aid progression of unmanaged mixed pine-hardwood forests towards upland hardwoods. One of the values of establishing long-term, stand-level silviculture studies is the flexibility in using the data for myriad objectives. Thinnings in this study were used to target both retained species (with an emphasis on hardwoods) and removed species (with an emphasis on pines). A consequence of this is an initial removal of biomass (and sequestered C), but with the potential to increase residual tree productivity. Prescribed fire is the other treatment in this study. For TLAT, prescribed fires at low intensities (cool, dormant season burns) had no direct impact on the biomass of the overstory trees. Neither of these practices was implemented to increase biomass or C storage. However, it has been reported that C (stored in aboveground biomass) can be released back to the atmosphere via disturbances such as wildfires or prescribed burns (Birdsey and others 2006, Canadell and Raupach 2008). Chiang and others (2008) also found no effect of dormant season fires on aboveground biomass except for a reduction in oak biomass; they also found an increase in stem mortality. We have not observed any increase in mortality 3 years post-burning in our study.

Post-treatment average tree TLAT biomass all increased. For the control (treatment 1) and burn only treatments (2 and 3), this is through tree growth and ingrowth over the 3 year sampling period. In the unthinned stands, 9 SPH were counted as ingrowth for the hardwoods and 3 SPH as ingrowth for the pines. For the thinned stands, the increase in average tree biomass was more substantial (about 80 kg), suggesting the thinning had some selection towards removing smaller trees. Increased growing space also increased the recruitment of new stems but only for the hardwoods (ingrowth of 2 SPH after 3 years). These young stands contained predominately smaller diameter trees, and thinning targeted those trees in the 15 to 31 cm diameter classes. There were few tallied trees of any species with a d.b.h. > 46 cm for the 24 stands that were thinned. For both the heavy- and light-thinned treatments, pine SPH in the15-cm d.b.h. class was reduced 90 percent. There was a 78 percent reduction in 20-cm pine and a 65 percent reduction in 25-cm pines.

Three seasons post-treatments, both the light- and heavy-thin stands increased productivity at 18.8 percent, compared to a 14.9 percent for the unthinned stands. Horner and others (2010) also found that moderate thinning resulted in the highest C storage rate, and that the lowest C storage was found in untreated stands. The sustainability of this short-term gain will be impacted by the age, diameter, and species distribution of the residual trees which may or may not continue to respond over time (D'Amato and others 2011, Hoover and Stout 2007). As these stands contained 23 tree species with dominant or codominant crown status, it is possible that stand dynamics, including ingrowth and residual growth, will shift with time and other disturbances (the continuation of the prescribed burns, wind events). Maintaining and enhancing diverse systems with various species, sizes, and functional groups are keys to resiliency to future disturbances, including climate change (Malmsheimer and others 2008, Ruddell and others 2007). Although stocking levels have been shown to explain variation in biomass and C stores, stocking of desirable species, or of species that may be more apt to continue to increase in biomass, are unknowns in these mixed pine-hardwood systems.

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LITERATURE CITED

Birdsey, R.; Pregitzer, K.; Lucier, A. 2006. Forest carbon management in the United States: 1600-2100. Journal of Environmental Quality. 35: 1461-1469.

Brown, S.; Lugo, A.E.; Chapman, J. 1986. Biomass of tropical tree plantations and its implications for the global carbon budget. Canadian Journal of Forest Research. 16: 390-394.

Canadell, J.G.; Raupach, M.R. 2008. Managing forests for climate change mitigation. Science. 320: 145-1457.

Chiang, J.M.; McEwan, R.W.; Yaussy, D.A.; Brown, K.J. 2008. The effects of prescribed fire and silvicultural thinning on the aboveground carbon stocks and net primary production of overstory trees in an oak-hickory ecosystem in southern Ohio. Forest Ecology and Management. 255: 1584-1594.

D'Amato, A.W.; Bradford, J.B.; Fraver, S.; Palik, B.J. 2011. Forest management for mitigation and adaption to climate change: insights from long-term silviculture experiments. Forest Ecology and Management. 262: 803-816.

Dixon, R.K; Brown, S.; Houghton, R.A. [and others]. 1994. Carbon pools and flux of global forest ecosystem. Science. 263: 185-190.

Fahey, T.J.; Woodbury, P.B.; Battles, J.J. [and others]. 2010. Forest carbon storage: ecology, management, and policy. Frontiers in Ecology and the Environment. 8(5): 245-252.

Hall, C.A.S.; Uhlig, J. 1991. Refining estimates of carbon released from tropical land-use change. Canadian Journal of Forest Research. 21: 118-131.

Hoover, C.; Stout, S. 2007. The carbon consequences of thinning techniques: stand structure makes a difference. Journal of Forestry. 105: 266-270.

Horner, G.J.; Baker, P.J.; Mac Nally, R. [and others]. 2010. Forest structure, habitat and carbon benefits from thinning floodplain forests: managing early stand density makes a difference. Forest Ecology and Management. 259: 286-293.

Jenkins, J.C.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A. 2004. Comprehensive database of diameter-based biomass regressions for North American tree species. Gen. Tech. Rep. NE-319. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station. 45 p.

Malmsheimer, R.W.; Heffernan, P.; Brink, S. [and others]. 2008. Forest management solutions for mitigating climate change in the United States. Journal of Forestry. 106: 115-171.

Marland, G.; Schlamadinger, B. 1997. Forests for carbon sequestration or fossil fuel substitution? A sensitivity analysis. Biomass and Bioenergy. 13(6): 389-397.

Nabuurs, G.J.; Masera, O.; Andrasko, K. [and others]. 2007. Forestry. In: Metz, B.; Davidson, O.R.; Bosch, P.R., eds. Climate change 2007: mitigation. Contribution of Working Group III to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK: Cambridge University Press: 339-870.

Ruddell, S.; Sampson, R.; Smith, M. [and others]. 2007. The role for sustainably managed forests in climate change mitigation. Journal of Forestry. 105(6): 314-319.

Schweitzer, C.J.; Wang, Y. 2013. Overstory tree status following thinning and burning treatments in mixed pine-hardwood stands on the William B. Bankhead National Forest, Alabama. In: Guldin, J.M., ed. 2010. Proceedings of the 15th biennial southern silvicultural conference. e-Gen. Tech. Rep. SRS-175. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 57-63.

Smalley, G.W. 1979. Classification and evaluation of forest sites on the southern Cumberland Plateau. Gen. Tech. Rep. SO-38. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 58 p.

USDA Forest Service. 2003. Final environmental impact statement, forest health and restoration project, national forests in Alabama, Bankhead National Forest. Management Bull. R8-MB 110B. Atlanta: U.S. Department of Agriculture Forest Service. 353 p.

Wear, D.N.; Greis, J.G. 2012. The southern forest futures project: summary report. Gen. Tech. Rep. SRS-168. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 68 p.

EASTERN COTTONWOOD AND BLACK WILLOW BIOMASS CROP PRODUCTION IN THE LOWER MISSISSIPPI ALLUVIAL VALLEY UNDER FOUR PLANTING DENSITIES

Ray A. Souter, Emile S. Gardiner, Theodor D. Leininger, Dana Mitchell and Robert B. Rummer[1]

"Wood is an obvious alternative energy source": Johnson and others (2007) declare the potential of short-rotation intensively-managed woody crop systems to produce biomass for energy. While obvious as an energy source, costs of production need to be measured to assess the economic viability of selected tree species as woody perennial energy crops. Further, an energy budget must be established to assess the economic viability of particular species grown using particular methods for energy crops. Such a budget compares the costs of planting stock, site preparation, establishment, tending, and harvests to the benefits of bioenergy production (Stanturf and Portwood 1999).

Eastern cottonwood (*Populus deltoides* W. Bartram ex Marsh.) and black willow (*Salix nigra* Marsh.) are native species of the Lower Mississippi Alluvial Valley (LMAV) with potential to provide significant biomass production under short-rotation growth and harvest regimes (Francis and Baker 1981, Kopp and others 2001, Mohn and Randall 1973, Ruark 2006, Volk and others 2006). As these species are perennial, production can be quite varied in the timing of biomass removal ranging from intervals as short as 2 years to harvests delayed for a decade or more.

Land area actually available for production needs to be considered. Soils selected for experimentation should represent a substantial potential resource. The statement contained in the Mississippi Biomass and Bioenergy Overview (Jackson 2007), "...on Conservation Reserve Program (CRP) land alone, … 3.8 million dry tons of willow and hybrid poplar could be produced each year" is indicative of one such resource. Lands not suitable for annual crop production are often selected for entry into CRP rolls. These lands that are considered marginal for agriculture represent a significant base for energy crop production. Producers considering retirement of land from annual cropping may be reluctant to invest resources in intensive cultural treatments, so it is reasonable to examine a system with low inputs of chemicals, including herbicides and fertilizers, as well as reduced irrigation.

Costs and benefits will be dependent on the species selected, the method of production, and the land available. While substantial experience with the native species exist with regard to their potential in energy cropping systems, little quantitative evidence is available on the effect a range of harvest intervals has on the production of these two species deployed in the LMAV on soils considered marginal for agriculture. An experiment (Study Plan USDA FS-SRS-4155-2011 on file with: Southern Research Station, Center for Bottomland Hardwoods Research, 432 Stoneville Road, Stoneville, MS 38776) designed to assess the costs, benefits, and their timing is underway to evaluate two species, black willow and eastern cottonwood, and four planting densities and harvest regime combinations. Four planting densities each with a distinct harvest regime include: (1) plant 4,100 trees per acre (tpa) with harvests in years 2, 4, 6, 8, and 10; (2) plant 2,489 tpa with harvests in years 4, 7, and 10; (3) plant 807 tpa with a thinning in year 3 followed by a complete harvest with re-establishment in year 5; and (4) plant 302 tpa with a complete harvest in year 10. With replication, a total of 24 experimental units (two species x four regimes x three replicates) are required. A total of 24 plots, each 2.5 acres in size with rectangular dimensions of 660 feet

[1]Research Forester, Research Forester, and Supervisory Research Plant Pathologist, respectively, USDA Forest Service, Southern Research Station, Stoneville, MS 38776; and Research Engineer and Project Leader (retired), USDA Forest Service, Southern Research Station, Auburn, AL 36849.

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long and 165 feet wide, are arranged in a randomized complete block design.

Establishment by hand planting approximately 120,000 cuttings of these species was conducted February 27, 2012 on marginal agricultural land located near Hollandale, MS. The land acquired by the U.S. Army Corps of Engineers is composed predominantly of Sharkey clay soil (Morris 1961). Figure 1 is an aerial image delineating the physical layout of the research installation.

Figure 1--Google Earth[TM] imagery dated November 12, 2012 clearly delineates the physical layout of the 24 experimental units located in Washington County, MS on U.S. Army Corps of Engineers land situated near the eastern boundary of Leroy Percy State Park, the wooded area on the left edge of the picture. Each of the 24 units is 2.5 acres in size with rectangular dimensions of 660 feet long and 165 feet wide. The image (© 2013 Google Inc. All rights reserved) is centered approximately on the coordinates 33° 09' 11.81" N latitude 90° 54' 26.09" W longitude.

The full 10 years of measurements including fuel energy inputs, bioenergy outputs, and impacts on soil properties over complete rotation periods will provide the necessary basis to determine the net energy production and soil effects of this cropping system. Here, initial establishment results of this study are presented (table 1). Early experience indicates chemical weed control is probably necessary to produce conditions that allow trees to perform well as they mature. Mechanical weed control did not provide adequate control in a zone near each planting spot to prevent over-topping of cuttings following their initial growth flush in the spring. Though first-year survival was not impacted by this competition, it is obvious that a more aggressive weed-control strategy must be employed for black willow to capture growing space, while the genetically improved stock available from established breeding programs for eastern cottonwood seemed to dominate the weed competition.

Table 1--Initial establishment results of two species, eastern cottonwood and black willow, planted under four spacings. Spacing (between row and within row distances are measured in feet) is inversely related to density.[ab] Averages of the three plots for each species and planting density combination are presented. Each plot was evaluated by counts of cuttings planted in 10 randomly selected row sections; each section was 66 feet in length. Both sprouting and survival success is considered to be uniformly good in all the treatments

Species	Treatment spacing	Sprouting	Survival
		----------*percent*----------	
Cottonwood	3 x 3	98.32	95.85
	7 x 2.5	99.62	99.23
	9 x 6	100.00	100.00
	12 x 12	100.00	100.00
Willow	3 x 3	100.00	92.43
	7 x 2.5	99.63	99.25
	9 x 6	100.00	99.09
	12 x 12	100.00	100.00

[a]3 x 3 = 4,100 trees per acre (tpa); 7 x 2.5 = 2,489 tpa; 9 x 6 = 807 tpa; and 12 x 12 = 302 tpa.

[b]Sprouting success of the cuttings following planting measured May 14, 2012. First-year survival measured October 24, 2012.

LITERATURE CITED

Francis, J.K.; Baker, J.B. 1981. Biomass and nutrient accumulation in a cottonwood plantation - the first four years. Res. Note SO-278. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 4 p.

Jackson, S.W. 2007. Mississippi biomass and bioenergy overview. http://www.25x25.org/storage/25x25/documents/State Page Documents/Mississippi/mississippi_sungrant.pdf. [Date accessed: May 9, 2008].

Johnson, J.M-F.; Coleman, M.D.; Gesch, R. [and others]. 2007. Biomass-bioenergy crops in the United States: a changing paradigm. The Americas Journal of Plant Science and Biotechnology. 1(1): 1-28.

Kopp, R.F.; Abrahamson, L.P.; White, E.H. [and others]. 2001. Willow biomass production during ten successive annual harvests. Biomass and Bioenergy. 20: 1-7.

Mohn, C.A.; Randall, W.K. 1973. Interaction of cottonwood clones with site and planting year. Canadian Journal of Forest Research. 3: 329-332.

Morris, W.M. 1961. Soil survey of Washington County, Mississippi. Series 1958, No. 3. Washington, DC: U.S. Department of Agriculture Soil Conservation Service. 56 p. In cooperation with: Mississippi Agricultural Experiment Station.

Ruark, G. 2006. Woodys have biomass potential. Inside agroforestry. 15(3). Lincoln, NE: U.S. Department of Agriculture National Agroforestry Center: 3, 11.

Stanturf, J.A.; Portwood, C.J. 1999. Economics of afforestation with eastern cottonwood (*Populus deltoides*) on agricultural land in the Lower Mississippi Alluvial Valley. In: Haywood, J.D., ed. Proceedings of the 10th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-30. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 66-72.

Volk, T.A.; Abrahamson, L.P.; Nowak, C.A. [and others]. 2006. The development of short-rotation willow in the northeastern United States for bioenergy and bioproducts, agroforestry and phytoremediation. Biomass and Bioenergy. 30: 715-727.

Biometrics

Moderator:

Eric Taylor
Texas A&M University
Department of Ecosystem Science and Management

TEMPORAL VALIDATION FOR A LANDSAT-BASED VOLUME ESTIMATION MODEL

Renaldo J. Arroyo, Emily B. Schultz, Thomas G. Matney, David L. Evans, and Zhaofei Fan[1]

Abstract--Satellite imagery can potentially reduce the costs and time associated with ground-based forest inventories; however, for satellite imagery to provide reliable forest inventory data, it must produce consistent results from one time period to the next. The objective of this study was to temporally validate a Landsat-based volume estimation model in a four county study area for possible use in large-scale forest inventories. Data were derived from Landsat Thematic Mapper (TM) imagery of Choctaw, Clay, Oktibbeha, and Winston counties in central Mississippi and compared with field inventory data from the Mississippi Institute for Forest Inventory (MIFI) for 1999 and 2006 (Clay County 2007). Comparisons were made between Landsat-based and MIFI inventory-based (90 percent confidence level) cubic foot volume outside bark to a pulpwood top (CFVOBPW) estimates in each of 2 study years. Results showed that, with one exception for the hardwood forest type cover class, estimates from the Landsat-based volume model fell within a reasonable sampling error range (± 20 percent) when calibrated to the MIFI field inventory. Further research will be conducted on: (1) re-parameterization of the model to correct for bias using a small sample of field plots; (2) the model's spatial validity; and (3) its potential for providing acceptable data substitutes where field inventory data are missing, too costly, or unattainable.

INTRODUCTION

Forests cover roughly 19.8 million acres or approximately 65 percent of Mississippi (Glass 2007) and are integral to the economy, producing over $1.1 billion in harvested timber annually (Henderson and others 2008). Mississippi's forests are also the basis for other aesthetic and economic benefits associated with wildlife, recreation, and tourism (Dahal and others 2013). Because of the importance of forest resources to the economy of Mississippi, timely and precise forest inventories are needed to: (1) attract new forest products industries, (2) provide decision support information to existing forest products industries, and (3) monitor forest change to ensure sustainability. In addition to the precision necessary to minimize risk associated with economic investment decisions at any one point in time, forest inventories must also be consistently valid in order to be applicable across multiple time periods.

Traditional field-based forest inventories are costly, untimely, or unattainable due to budget restraints, logistics, topography, or other circumstances. The need for timely, cost effective data has promoted the use of remotely sensed data such as that derived from Landsat Thematic Mapper (TM) imagery because of its ability to quantify and qualify large areas. While field-based forest inventory systems are already in place at the state and county levels in Mississippi, the use of Landsat TM data alone has potential for providing acceptable volume estimates when USDA Forest Service Forest Inventory and Analysis (FIA) and Mississippi Institute of Forest Inventory (MIFI) data are unavailable or unattainable.

The purpose of this research was to temporally validate Landsat-derived cubic foot volume outside bark to a pulpwood top (CFVOBPW) estimates by comparing them to MIFI field-based CFVOBPW estimates at two separate time periods. Model performance of a 1999 MIFI pilot study of four counties (Choctaw, Clay, Oktibbeha, and Winston) in central Mississippi was compared to 2006 MIFI central region and 2007 north region inventory data to establish a standard on which to calibrate temporal validity for the Landsat-based estimations of the same time periods. Temporal validity can be established because both field plot estimates and Landsat-only based estimates are available for these specific years. The hypothesis of this work was that the Landsat-based estimates would fall within ± 20 percent of the MIFI field-based estimates.

[1]Graduate Research Assistant, Professor, Professor, Professor, and Associate Professor, respectively, Mississippi State University, Forest and Wildlife Research Center, Mississippi State, MS 39762.

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METHODS

MIFI Inventory Data

CFVOBPW was selected for comparing MIFI inventory and Landsat-based volume estimates because it was the MIFI inventory target/design variable and exhibited the least amount of variation among volume estimates. MIFI CFVOBPW estimates were obtained for four Mississippi counties (Choctaw, Clay, Oktibbeha, and Winston) inventoried in a 1999 pilot study (Parker and others 2005) and re-inventoried in the 2006 MIFI central region inventory (for Choctaw, Oktibbeha, and Winston counties) (Glass 2007) and the 2007 MIFI north region inventory (for Clay County) (Glass 2008). MIFI divides the state into five inventory regions sampled yearly on a rotating basis. Choctaw, Oktibbeha, and Winston counties were placed in the central inventory region while Clay County was placed in the north region causing it to be inventoried in a different year in the annual rotation.

MIFI CFVOBPW estimates for the four counties and two time periods were obtained from Mississippi Forest Inventory Dynamic Reporter Version 7 software (Matney and Schultz 2011) available through the Mississippi State University Forest and Wildlife Research Center (FWRC) and downloadable from www.timbercruise.com (Download Center-Desktop-Other Solutions-Mississippi Institute of Forest Inventory Dynamic Inventory Reporter). Dynamic Reporter forest type and species group report selections were made using criteria similar to those employed by MIFI in determining GIS forest cover type sampling strata, pine and hardwood. The pine stratum was specified as consisting of all species groups within all Society of American Foresters (SAF) forest types that were solely composed of pine. The hardwood stratum was specified as consisting of all species groups within all SAF forest types that were hardwood only. There were too few MIFI plots (n = 15) containing mixed pine and hardwood SAF forest types for a comparison of GIS and MIFI volume estimates (Wilkinson 2011). A minimum merchantable age of 10 was used for both pine and hardwood, and non-merchantable plots (non-forest, sub-merchantable, and regeneration) were also excluded.

Landsat Thematic Mapper (TM) Data

Techniques used for deriving volume data from Landsat TM imagery were modified from Wilkinson (2011). Both 1999 and 2006 image data sets were 30-m resolution projected in Mississippi Transverse Mercator (MSTM). Age, forest type, and radiance layers were subset to the four county study area and used as dependent variables in hardwood and pine stratum models. Age layers for 1999 and 2006 were constructed from a sequence of 14 times, representing every third year, from 1972 to 2006 using a Microsoft C++® program. Forest and non-forest cover type distinctions were made using approximately 300 clusters in unsupervised classification (Collins and others 2005). Hardwood and pine forest type pixel classifications were made interpretatively using an 80 percent or greater hardwood or pine rule. Anything < 80 percent was separated into the mixed category. The radiance layer was created using a reflectance to radiance model in ERDAS Imagine's® Model Maker function. Pixel values from bands 1 to 5 and 7of the original satellite images (1999 and 2006) were converted to radiance values by Chander and others (2007) procedures.

Volume Estimates

Once Landsat-derived variables were created, hardwood and pine CFVOBPW per acre models (equations 1 and 2) were constructed using techniques derived from Schultz and others (2006). Hardwood- and pine-modified power models were tested multiple times for efficiency, with age being used as a correction factor for the over/under prediction of the model. Both power models were run with SAS® Version 8 non-linear (NLIN) procedure using the Gauss-Newton method with CFVOBPW per acre as the dependent variable. The power models achieved higher index of fit (1 minus the quantity of the error sum of squares divided by the total sum of squares) than other models tested. Final models were incorporated into a Microsoft C++® program in order to calculate both total and average per acre volume for hardwood and pine. Volume data from both MIFI- and Landsat-based inventories were compared to help calibrate the standard from time period to time period. An acceptable sampling error range of ± 20 percent was evaluated for Landsat-based estimates of CFOBPW when compared to the MIFI field inventory standard.

$$\text{Pine } \widehat{\text{CFVOBPW}} = 22822.8 - 23600.6 AGE^{-0.3476} - 5089.6 R1^{-0.3449} R2^{0.2654} R3^{-0.0510} R4^{0.0756} R5^{0.2832} AGE^{0.1747} \qquad (1)$$

$$\text{Hardwood } \widehat{\text{CFVOBPW}} = 316973 - 319999 AGE^{-0.00544} - 13053.8 R2^{1.6721} R3^{-4.1671} R4^{-2.4279} R5^{4.8113} AGE^{1.4272} \quad (2)$$

where *AGE* = age determined from a forest/non-forest change detection temporal sequence beginning in 1972; *R1* = Landsat radiance band 1; *R2* = Landsat radiance band 2; *R3* = Landsat radiance band 3; *R4* = Landsat radiance band 4; *R5* = Landsat radiance band 5; and a rectangle over the variable designates an estimated value.

RESULTS

MIFI 1999 CFVOBPW per acre volume estimates were 2,607 and 2,380 for hardwood and pine cover classes, respectively, while Landsat-derived per acre volume estimates were 3,808 and 1,978, respectively (table 1). The hardwood Landsat estimate exceeded the hypothesized ± 20 percent sampling error while the Landsat pine cover class fell within the ± 20 percent sampling error. MIFI 2006 CFVOBPW per acre volume estimates were 2,525 for hardwood and 2,064 for pine cover classes. Landsat per acre volume estimates were 2,477 and 2,091 for hardwood and pine cover classes, respectively. Both Landsat estimates were within 2 percentage points of the MIFI estimates, well within the hypothesized ± 20 percent sampling error.

Table 1—Cubic foot volume outside bark to a pulpwood top (CFVOBPW) per acre MIFI field inventory (±20 percent acceptable sampling range at the 90 percent confidence interval) and Landsat TM model derived estimations and percentage difference comparisons

Forest cover class/year	MIFI inventory	Landsat derived	Difference
	------CFVOBPW ac^{-1}--------		*--percent--*
Hardwood			
1999	2,607 ± 521	3,808	46.1
2006/2007	2,525 ± 505	2,477	1.9
Pine			
1999	2,380 ± 476	1,978	16.9
2006/2007	2,064 ± 413	2,091	1.3

DISCUSSION AND SUMMARY

With one exception, the 1999 hardwood cover class, power models based on Landsat data and age produced CFVOBPW per acre estimates that fell within our hypothesized sampling error range of ± 20 percent compared to MIFI inventory estimates. Further calibration and model refinement will be possible in 2014 when additional temporal data is available from the first re-measurement of the MIFI central inventory region and from Landsat 8 satellite imagery. Research is also in progress to spatially validate the models with MIFI data from other inventory regions and at different scales (e.g., county, multi-county, state, working circles, polygons). Extensibility of the procedure to other regions may require recalibration of the model through a small sample of ground truth points.

The models developed here are preliminary, and on-going research is being conducted to refine and improve them. As refinements and validations are made, the Landsat-based models have the potential for providing acceptable data substitutes where field inventory data are missing, too costly, or unattainable.

LITERATURE CITED

Chander, G.; Markham, B.L.; Barsi, J.A. 2007. Revised Landsat-5 Thematic Mapper radiometric calibration. IEEE Geoscience and Remote Sensing Letters. 4(3): 490-494.

Collins, C.A.; Wilkinson, D.W.; Evans, D.L. 2005. Multi-temporal analysis of Landsat data to determine forest age classes for the Mississippi statewide forest inventory - preliminary results. Proceedings of the 3rd international workshop on the analysis of multi-temporal remote sensing images: 10-14. [CD-ROM].

Dahal, R.P.; Munn, I.A.; Henderson, J.E. 2013. Forestry in Mississippi: the impact of the industry on the Mississippi economy - an input-output analysis. Res. Bull. FO 438. Mississippi State, MS: Forest and Wildlife Research Center. 22 p.

Glass, P. 2007. State of Mississippi central district forest inventory. Jackson, MS: Mississippi Forestry Commission, Mississippi Institute for Forest Inventory. 24 p.

Glass, P. 2008. State of Mississippi north district forest inventory. Jackson, MS: Mississippi Forestry Commission, Mississippi Institute for Forest Inventory. 24 p.

Henderson, J.E.; Munn, I.A.; Perez-Verdin, G.; Grebner, D.L. 2008. Forestry in Mississippi: the impact of the forest products industry on the post-Katrina Mississippi economy- an input-output analysis. Res. Bull. FO 374. Mississippi State, MS: Forest and Wildlife Research Center. 31 p.

Matney, T.G.; Schultz, E.B. 2011. Mississippi Forest Inventory Dynamic Reporter Version 7. Mississippi State, MS: Forest and Wildlife Research Center. http://www.mifi.ms.gov. [Date accessed: January 7, 2013]

Parker, R.C.; Glass, P.A.; Londo, H.A. [and others]. 2005. Mississippi's forest inventory pilot program: use of computer and spatial technologies in large area inventories. Bull. FO 274. Mississippi State, MS: Forest and Wildlife Research Center. 22 p.

Schultz, E.B.; Matney, T.G.; Evans, D.L.; Fujisaki, I. 2006. A Landsat stand basal area classification suitable for automating stratification of forest into statistically efficient strata. In: Lang, S.; Blaschke, T.; Schopfer, E., eds. Proceedings of the 1st international conference on object-based image analysis. Salzburg, Austria: International Society for Photogrammetry and Remote Sensing (ISPRS) Vol. XXXVI- 4/C42. 6 p.

Wilkinson, D.W. 2011. Landsat-derived stand structure estimation for optimizing stratified forest inventories. Mississippi State, MS: Mississippi State University. 42 p. Ph.D. dissertation.

USING NONLINEAR QUANTILE REGRESSION TO ESTIMATE THE SELF-THINNING BOUNDARY CURVE

Quang V. Cao and Thomas J. Dean[1]

Abstract--The relationship between tree size (quadratic mean diameter) and tree density (number of trees per unit area) has been a topic of research and discussion for many decades. Starting with Reineke in 1933, the maximum size-density relationship, on a log-log scale, has been assumed to be linear. Several techniques, including linear quantile regression, have been employed to obtain parameters of the self-thinning line. Some authors recently considered that restriction on the maximum diameter at lower spatial densities resulted in a curvilinear relationship. In this study, a nonlinear quantile regression based on the 99th quantile was used to characterize this upper boundary. The resulting self-thinning curve fit the curvilinear boundary much better than did the Reineke's self-thinning line.

INTRODUCTION

The reciprocal relationship between tree size and stand density has been a topic of research and discussion since Reineke (1933) expressed the logarithm of maximum quadratic mean diameter at breast height (Q_m) as a linear function of the logarithm of number of trees per unit area (N). He found that a slope of -1.605 adequately described the relationship for 12 out of 14 species examined. Other researchers have since statistically fit slopes to log-transformed values of Q_m and N and found that they varied, ranging in values from -1.707 to -1.505 (Bailey 1972, Drew and Flewelling 1977, Harms 1981, MacKinney and Chaiken 1935, Williams 1996).

While the relationship between tree size and tree density on a log-log scale can be considered linear within some range of stand density, it might actually be curvilinear throughout the entire range of tree density because trees are ultimately limited in size by their weight, thus restricting the maximum diameter associated with lower spatial densities (Westoby 1984). The curvilinear boundary was evident for southern pines (Zeide 1987) and slash pines (Cao and others 2000) in particular.

Quantile regression (Koenker and Bassett 1978) has recently been employed by scientists from various backgrounds to address research problems in medicine (Austin and Schull 2003), economics (Machado and Mata 2005), education and policy (Haile and Nguyen 2008), and natural resource management (Cade and others 2005). In forestry, quantile regression has been applied to compute stand density index (Ducey and Knapp 2010) or evaluate the spread rate of forest diseases (Evans and Finkral 2010).

One advantage of quantile regression over ordinary least squares regression is that the quantile regression estimates are more robust against outliers. It is particularly useful in estimating the quantiles (or percentiles) of the response variable, for example, tree diameter percentiles (Mehtätalo and others 2008) or maximum diameters in self-thinning stands (Zhang and others 2005).

The objective of this study was to apply nonlinear quantile regression in modeling the self-thinning boundary curve.

MATERIALS AND METHODS

Data

Data from 147 permanent plots of direct-seeded slash pine (*Pinus elliottii* Engelm.) stands were used for this study. These stands were established on cutover sites located in Natchitoches and Rapides Parishes (central Louisiana) and in Washington Parish (southeast Louisiana). Baldwin (1985) and Lohrey (1987) described these data in detail. Plot size ranged from 0.040 to 0.048 ha. Stand age ranged from 8 to 28 years, stand density from 445 to 12,108 trees/ha, basal area from 2.6 to 52.6 m^2/ha, and site index (base age 25 years) from 9 to 23 m. Some plots were precommercially thinned at age 3 or 4 years. Each plot was measured from 3 to 6 times, at 3 to 10 years apart. There was a total of 615 measurements encompassing 468 growth periods. The trajectories of stand density and quadratic mean diameter for these measurements are shown in figure 1.

[1]Professors, Louisiana State University Agricultural Center, School of Renewable Natural Resources, Baton Rouge, LA 70803.

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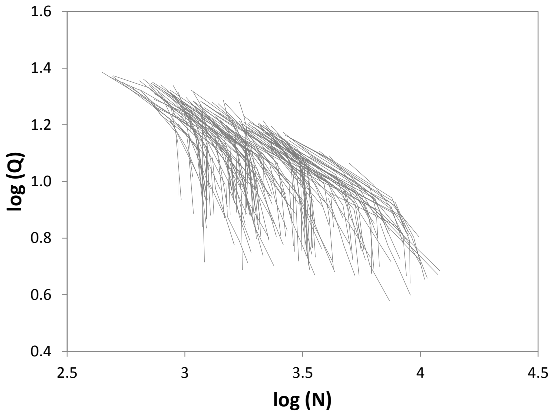

Figure 1--Trajectories of quadratic mean diameter (Q) stand density (N) for measurements of direct-seeded slash pines.

Self-thinning Boundary

The maximum number of trees that can survive on a given land area depends on the quadratic mean diameter and decreases predictably with increasing mean diameter as a stand self-thins. Reineke (1933) described this boundary with the simple linear equation:

$$\log(N) = a + b \log(Q_m) \tag{1}$$

where $b = -1.605$ for many species. This equation can be rewritten as:

$$Q_m = = b_1 N^{-0.623} \tag{2}$$

where $b_1 = 10^a$ and $-0.623 = 1/(-1.605)$.

Cao and others (2000) proposed the following nonlinear relationship for the self-thinning boundary:

$$Q_m = b_1 N^{-0.623}[1 - \exp(b_3 N^{b_4})]. \tag{3}$$

Quantile Regression

Parameters b_1, b_3, and b_4 can be estimated via quantile regression techniques by minimizing:

$$S = \sum_{Q_i \geq \hat{Q}_i} \tau |Q_i - \hat{Q}_i| + \sum_{Q_i < \hat{Q}_i} (1 - \tau)|Q_i - \hat{Q}_i| \tag{4}$$

where Q_i = observed tree diameter at breast height and $\hat{Q}_i$ = predicted τ^{th} quantile of tree diameters. SAS procedure NLP (SAS Institute Inc. 2010) was used for this purpose.

RESULTS AND DISCUSSION

Predicted self-thinning curves for the 90th, 95th, 99th, 100th quantiles from equation (3) (i.e., τ = 0.90, 0.95, 0.99, and 1.00) have different shapes (fig. 2). Compared to the apparent shape of the boundary seen with the data, the 90th and 95th quantiles do not have adequate curvature, and the one based on the 100th quantile is too high for the data.

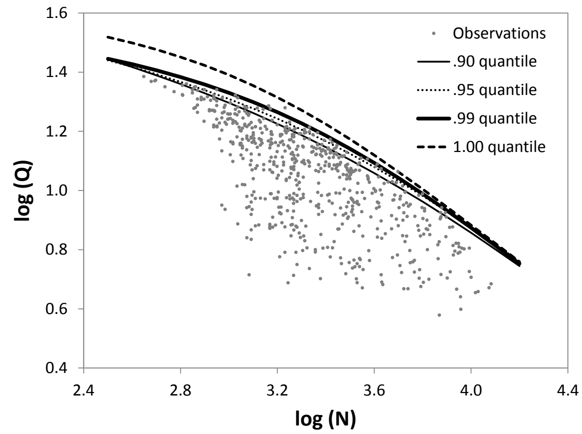

Figure 2--Self-thinning boundary curves predicted from equation (3) by use of quantile regression with four values of τ: 0.90, 0.95, 0.99, and 1.00.

The self-thinning boundary curve based on the 99th quantile appears appropriate to model the relationship between maximum tree diameter and stand density (fig. 3). The final equation is:

$$\hat{Q}_m = 2326\, N^{-0.623}[1 - \exp(-0.013\, N^{0.645})]. \tag{5}$$

For this data set, the self-thinning curve provides a more realistic reciprocal relationship between maximum tree diameter and stand density than does the self-thinning line proposed in 1933 by Reineke (fig. 3). Cao and others (2000) explained that the self-thinning curve is approximately linear for either a narrow range of low stand densities or a wider range of relatively high stand densities. This also explains different slopes of the self-thinning line reported from data of different stand density ranges; the slope varies with the range of *N* included in the data set.

The wide range of stand density from this direct-seeded slash pine data set (from 445 to 12,108 trees/ha) allowed us to construct a meaningful self-thinning curve. While the quantile regression techniques proved to be appropriate for this task, only the 99th quantile appeared to be best for representing the boundary curve.

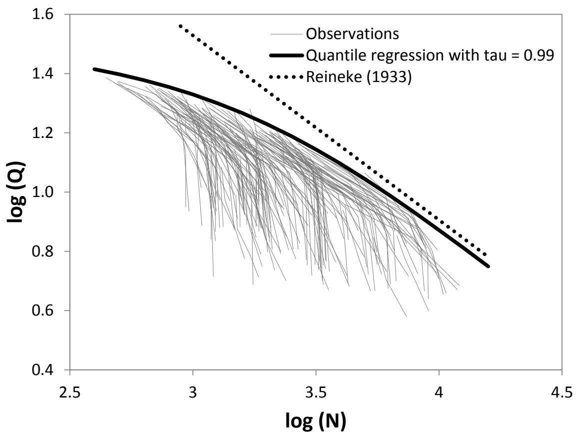

Figure 3--Reineke's (1933) self-thinning line and self-thinning curve from quantile regression with $\tau = 0.99$.

LITERATURE CITED

Austin, P.C.; Schull, M.J. 2003. Quantile regression: a statistical tool for out-of-hospital research. Academic Emergency Medicine. 10: 789-797.

Bailey, R.L. 1972. Development of unthinned stands of *Pinus radiata* in New Zealand. Athens, GA: University of Georgia. 73 p. Ph.D. dissertation [abstract 33/09−B−4061].

Baldwin, V.C., Jr. 1985. Survival curves for unthinned and early-thinned direct-seeded slash pine stands. In: Shoulders, E., ed. Proceedings of the 3rd biennial southern silvicultural research conference. Gen. Tech. Rep. SO-54. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 460-465.

Cade, B.S.; Noon, B.R.; Flather, C.H. 2005. Quantile regression reveals hidden bias and uncertainty in habitat models. Ecology. 86: 786-800.

Cao, Q.V.; Dean, T.J.; Baldwin, V.C., Jr. 2000. Modeling the size-density relationship in direct-seeded slash pine stands. Forest Science. 46: 317-321.

Drew, T.J.; Flewelling, J.W. 1977. Some recent Japanese theories of yield-density relationships and their application to Monterey pine plantations. Forest Science. 23: 517-534.

Ducey, M.J.; Knapp, R.A. 2010. A stand density index for complex mixed species forests in the northeastern United States. Forest Ecology and Management. 260: 1613-1622.

Evans, A.M.; Finkral, A.J. 2010. A new look at spread rates of exotic diseases in North American forests. Forest Science. 56: 453-459.

Haile, G.A.; Nguyen, A.N. 2008. Determinants of academic attainment in the United States: a quantile regression analysis of test scores. Education Economics. 16: 29-57.

Harms, W.R. 1981. A competition function for tree and stand growth models. In: Barnett, J.P., ed. Proceedings of the 1st biennial southern silvicultural research conference. Gen. Tech. Rep. SO−34. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 179-183.

Koenker, R.; Bassett, G., Jr. 1978. Regression quantiles. Econometrica. 46: 33-50.

Lohrey, R.E. 1987. Site index curves for direct-seeded slash pines in Louisiana. Southern Journal of Applied Forestry. 11: 15-17.

Machado, J.A.F.; Mata, J. 2005. Counterfactual decomposition of changes in wage distributions using quantile regression. Journal of Applied Economics. 20: 445-465.

MacKinney, A.L.; Chaiken, L.E. 1935. A method of determining density of loblolly pine stands. Tech. Note 15. Asheville, NC: U.S. Department of Agriculture Forest Service, Appalachian Forest Experiment Station. 3 p.

Mehtätalo, L.; Gregoire, T.G.; Burkhart, H.E. 2008. Comparing strategies for modeling tree diameter percentiles from remeasured plots. Environmetrics. 19: 529-548.

Reineke, L.H. 1933. Perfecting a stand-density index for even-aged forests. Journal of Agricultural Research. 46: 627-638.

SAS Institute Inc. 2010. SAS/OR 9.22 User's guide: mathematical programming. Cary, NC: SAS Institute. 1264 p.

Westoby, M. 1984. The self-thinning rule. In: Macfadyen, A.; Ford, E.D., eds. Advances in ecological research. New York: Academic Press: 167-225.

Williams, R.A. 1996. Stand density index for loblolly pine plantations in north Louisiana. Southern Journal of Applied Forestry. 20: 110-113.

Zeide, B. 1987. Analysis of the 3/2 power law of self-thinning. Forest Science. 33: 517-537.

Zhang, L.; Bi, H.; Gove, J.H.; Heath, L.S. 2005. A comparison of alternative methods for estimating the self-thinning boundary line. Canadian Journal of Forest Research. 35: 1507-1514.

AN EVALUATION OF THE HARDWOOD REGENERATION MODEL (REGEN) 16 YEARS POST-HARVEST OF A REGENERATED STAND IN EAST TENNESSEE

Wayne K. Clatterbuck[1]

Abstract--The REGEN model (developed by USDA Forest Service, Southern Research Station, Bent Creek Experimental Forest) was used prior to harvest to predict species composition of hardwoods at crown closure. This study evaluates whether the predictive ability of the model was effective by using post-harvest information after 16 years. Regeneration data were collected prior to harvest in February through April 1997 at the University of Tennessee Forest Resources Research and Education Center near Oak Ridge, TN. Five site preparation treatments were implemented to favor desired species and to control undesired species in the future stand: preharvest slash only, preharvest slash with herbicide stump treatment, post-harvest slash only, post-harvest slash and herbicide stump treatment, and control (no slashing or herbicide). Each set of five treatments was replicated six times for a total of 30 treatment plots (0.33 acre per treatment plot). Predictions of overstory composition and number from the REGEN model (using the Southern Appalachian variant) by site preparation treatment are compared to the actual stand data after 16 years to determine the accuracy and utility of the model in forecasting species composition at crown closure prior to harvest. Results indicate that the REGEN model performed well in predicting species composition at crown closure. The model overcompensated for presence of yellow-poplar (*Liriodendron tulipifera* L.), enough to provide reasonable estimations of the number of overstory stems after 16 years. Pre- or post-harvest site preparation techniques had little effect on presence of light-seeded species such as yellow-poplar and black cherry (*Prunus serotina* Ehrh.). About 10 percent of total overstory stems after 16 years were oaks (*Quercus* spp.), regardless of site preparation treatment. The same percentage of advanced oak seedlings was represented in the pre-harvest regeneration inventory.

INTRODUCTION

Ensuring adequate regeneration of preferred species in mixed hardwood stands following a harvest is often a concern to forest managers. A myriad of different species with different site requirements and growth habits and varying sources of reproduction (seed, sprouts, advance regeneration) make prediction of regeneration complex and sometimes unreliable. Often competition from undesirable trees is too great for the commercially important species to overcome.

Encouraging growth of preferred, regenerating species often is accomplished by limiting through site preparation the growth or presence of more undesirable competing species. Slashing and/or herbicides are two methods of site preparation. Little information is available (either pre- or post-harvest treatment) to assess the relative effectiveness of these various site preparation alternatives. Loftis (1978, 1985, 2004) evaluated the effectiveness and costs associated with preharvest practices on Appalachian hardwoods. The results suggest that preharvest treatments reduce the number of stems of undesirable species and increase the proportion of desirable species in the stand.

Subsequently, a forest regeneration model, REGEN, was developed for the southern Appalachians by the USDA Forest Service, Southern Research Station, Bent Creek Experimental Forest near Asheville, NC (Boucugnani 2005, Loftis 1989) to predict species composition at crown closure based on preharvest regeneration sources. Forest managers can use the model as a tool to determine if the predicted regeneration results are satisfactory to meet their objectives. Otherwise, if the results are unsatisfactory, practitioners can take management actions that may lead to more desirable results. These actions typically modify the environment to benefit regeneration and development of desired species and discourage undesirable species. A few examples of actions that could be taken before the harvest that would influence regeneration potential would be a midstory removal to develop greater size of advance reproduction or the use of herbicides to influence competing vegetation.

[1]Professor, The University of Tennessee, Department of Forestry, Wildlife and Fisheries, Knoxville, TN 37996-4563.

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REGEN is a probabilistic model based on an expert system using a numerical ranking system of species' competitiveness. The model predicts regeneration outcomes following stand-replacement disturbance events. Regeneration sources (seed, sprouts, or advance reproduction) are specified in a preharvest regeneration inventory by species and size. Probabilities and rankings in the form of knowledge bases (species and propagules) determine the relative likelihood that a species/propagule will be a component of the future overstory at canopy closure. For further information about the REGEN model refer to Boucugnani (2005) or Vickers and others (2011). A web-based version of the model is available at http://www.webregen.org.

The REGEN model is a prospective tool for forest land managers to evaluate regeneration potential. However, the predictive model, based on an expert system, has not been thoroughly tested with actual data because of the 10 to 20 years required for a regenerating stand to reach canopy closure. This study provides an assessment of the REGEN model in one stand in east Tennessee that used several site preparation techniques to influence regeneration.

OBJECTIVES

The purpose of this study was to evaluate the predictive outcomes of the REGEN model from pre-harvest regeneration sources by assessing actual overstory composition at canopy closure or stand exclusion 16 years after a commercial clearcut harvest with various site preparation sources.

METHODS

The research was located on a 17-acre watershed at the University of Tennessee Forest Resources Research and Education Center near Oak Ridge, TN in the Ridge and Valley physiographic province. Elevations in the south-facing drainage range from 970 to 1,100 feet above sea level. Soils consist of clayey, kaolinitic, thermic Typic Hapludults from the Fullerton series (Moneymaker 1981). Site index (base age of 50 years) for upland oaks ranges from 65 to 75 feet (Olson 1959). The harvested sawtimber stand was comprised primarily of oaks (*Quercus* spp.) (69 percent by volume), yellow-poplar (*Liriodendron tulipifera* L.) (14 percent), miscellaneous hardwoods (10 percent), and pines (*Pinus* spp.) (6 percent).

Five treatments were implemented: (1) preharvest slash only; (2) preharvest slash with herbicide stump treatment; (3) post-harvest slash only; (4) post-harvest slash and herbicide stump treatment; and (5) control (no slashing or herbicide).

The five treatments were applied to 120- by 120-foot (0.33 acre) plots. This plot size was large enough to distinguish an individual treatment from adjacent treatment while allowing for replications. Each set of five treatments was replicated six times for a total of 30 treatment plots. All treatment plots were located adjacent to each other.

Within each plot, four 1/100 acre subplots were established for sampling. Each subplot was located at a corner of a 60- by 60-foot square contained within each plot. The first subplot was established by running a line bisecting the northern corner of each plot for 42.5 feet. The remaining subplots were positioned by running 60-foot lines parallel to the boundaries of the plot.

Plots were assigned to different replications by establishing groups of plots that were similar in terms of species composition, density, and location. A computer-generated design for incomplete blocks was used to assign treatments to plots.

The initial inventory was conducted in June 1996 before the harvest. All trees above 1 foot in height were measured in subplots during September 1996. Data were collected in several designated classes: 1-foot height classes to 4-feet tall; above 4-feet tall but < 1.5 inches at diameter breast height (d.b.h., 4.5 feet); and by 1-inch diameter class above 1.5 inches. This methodology follows the input data required for the regeneration prediction model (REGEN). The 4 subplots on each of the replicates with the same treatment (24 total subplots for each treatment) were used to predict percentage species composition at crown closure using the REGEN model.

Preharvest treatments were conducted on the designated plots during October 1996. The number of stems cut per plot was recorded as stems > or < 1.5 inches d.b.h. Non-commercial stems >1.5 inches d.b.h. were cut. Garlon® 3A (triclopyr) in a 50:50 mix with water and red dye was used on all non-commercial stumps, primarily red maple (*Acer rubrum* L.), blackgum (*Nyssa sylvatica* Marsh.), sourwood (*Oxydendrum arboreum* DC), dogwood (*Cornus florida* L.), sweetgum (*Liquidambar styriciflua* L.), elms *(Ulmus* spp.), and beech (*Fagus grandifolia* Ehrh.). Herbicide was applied to the stump directly after cutting.

The timber harvest was conducted from February through April 1997. Post-harvest treatments were conducted in August 1997 in the same manner as the preharvest treatments.

All subplots were measured after the second growing season (1999) and the results on species composition and costs associated with pre- and post-harvesting treatments were reported by Hodges and others (2002). Measurements after 10 growing seasons (2006) were collected during December 2006 and January 2007. Data collected on each subplot were stem counts by species for all stems > 4 feet in height. Results after 10 growing seasons were reported by Clatterbuck and Schubert (2010).

Measurements for this study were collected in December 2012 and January 2013. Stems >1.5 inches in d.b.h. on the subplots were defined as overstory stems and were tallied by species. The study area was in the stem exclusion stage with a closed canopy. Data from the subplots were combined by treatment for the six replicates. Percentage species composition was compared between the predicted species composition from the pre-harvest regeneration at stand closure to the actual data 16 years after harvest.

The purpose of the original study was to evaluate site preparation treatments on species composition after a silvicultural clearcut. The experimental design for that study is not appropriate for testing statistical differences between actual and predicted (REGEN model) species composition since only one stand was sampled, although there were different treatments and the treatments were replicated. Thus, this research should be considered a case study comparing predicted species composition percentages of the REGEN model at the stem exclusion stage of stand development and actual species composition percentage of overstory trees at crown closure measured 16 years after harvest.

RESULTS

The stand prior to harvest was in the stem exclusion stage for many years, resulting in a sparse midstory and understory with few stems or seedlings of desirable species. More than 95 percent of the stems were < 2 feet in height, and most of the species present (red maple, dogwood, sourwood, blackgum, beech) were shade tolerant. Red maple stems composed more than 60 percent of the understory composition. Few yellow-poplar or black cherry, both extremely shade-intolerant species, were present in the advance reproduction. Oak species that were < 2 feet in height averaged about nine stems per 1/100 acre plot.

Stand structure 16 years after harvest was a closed canopy with an average canopy height of 42 feet and mean d.b.h. of 3.9 inches with a wide range of diameter of overstory trees from 1.5 to 8.2 inches.

The predicted species composition at canopy closure of the various site preparation treatments from the REGEN model are shown in table 1. Both yellow-poplar and black cherry have the greater number of stems regardless of treatment. The amount of red maple was reduced in the herbicide treatments. Oak species composed < 5 percent of the stems in any of the treatments.

Table 2 presents actual overstory composition by site preparation treatment 16 years after the harvest at canopy closure (stem exclusion). Generally, more oaks were present in the actual data than in the predicted data, and miscellaneous species were much more robust in the actual data. Red maple was less numerous when herbicide was used in both the predicted and actual datasets. Black cherry had greater proportion of stems in the predicted

Table 1—REGEN model prediction of species composition at canopy closure by site preparation treatment prior to harvest for a hardwood stand at the University of Tennessee Forest Resources Research and Education Center near Oak Ridge, TN. Data for each treatment based on n = 24, 1/100-acre regeneration plot

Site preparation treatment	Yellow-poplar	Black cherry	Red maple	All oaks	Misc. species[a]
	percentage of total stems				
Preharvest slash	38	29	21	2	10
Preharvest slash + herbicide	39	40	12	5	4
Post-harvest slash	51	23	20	2	4
Post-harvest slash + herbicide	49	34	9	1	6
Control - no site preparation	40	30	22	4	4

[a]Miscellaneous species include beech, blackgum, dogwood, elms, hollies (*Ilex* spp.), sassafras [*Sassafras albidum* (Nutt.) Nees], sourwood, sweetgum, pines, sumac (*Rhus* spp.), and white ash (*Fraxinus americana* L.).

model than in the actual data. Yellow-poplar percentages did not range widely between the actual and predicted data, herbicide treatments, or pre- and post-harvest treatments. All species percentages were fairly similar between the actual preharvest and post-harvest treatments.

DISCUSSION

The REGEN model performed well in predicting species composition at crown closure. A few of the discrepancies may be because the knowledge base used was formulated for the southern Appalachian mountains and not the Ridge and Valley province. Black cherry is much more prominent at the higher elevations of the mountains than the Ridge and Valley, where it is much more susceptible to pitch pockets, black knot, rot, and crown breakage from wind and ice damage. The REGEN knowledge base would need to be modified to more correctly classify black cherry in this area.

Species that regenerate readily from seed, particularly yellow-poplar and black cherry, dominate the study area after 16 years (table 2). Pre- and post-harvest slashing and the use of herbicide during site preparation have little impact on the regeneration of these species since they do not regenerate from advance reproduction. The seeds remain viable on the forest floor for several years: yellow-poplar for 4 to 7 years (Clark and Boyce 1964) and black cherry for 3 or more years (Wendel 1977). The viable seeds accumulate in the forest floor after seed dissemination each year forming a seed bank ready to germinate when conditions are favorable, especially after a harvest and site preparation. The numerous, viable seeds in the seed bank often overburden the advance regeneration present on the site prior to harvest shifting the species composition toward these species. The REGEN model estimation of yellow-poplar reflected the actual data. However, the amount of black cherry predicted by the model often was overestimated compared to the actual data.

The site preparation treatments with herbicides, both pre- and post-treatment, reduced the amount of red maple in the model and with the actual data. Red maple was the most common species in the midstory and understory in this closed-canopy stand and was the species that received most of the herbicide applied. However, with the overabundance of yellow-poplar from seed, the growth of red maple into the overstory also was affected by the faster growth of yellow-poplar. Red maple does not appear to be aggressive enough to become a major component of the overstory although more advance reproduction of red maple was present than for any other species.

Oaks were not affected by the site preparation treatments. The same proportion of oaks was present in advance reproduction as the proportion of oaks in the overstory after 16 years. The model estimation for oak species under-represented what actually occurred. Presently about 200 oaks per acre (about 10 percent of total overstory stem) remain at 16 years, and that number will probably diminish with time. Oaks will be a component of the next stand but at a much lower density than in the

Table 2—Actual species composition at canopy closure 16 years after harvest by site preparation treatment for a hardwood stand at the University of Tennessee Forest Resources Research and Education Center near Oak Ridge, TN. Data for each treatment based on n = 24, 1/100-acre regeneration plot

Site preparation treatment	Yellow-poplar	Black cherry	Red maple	All oaks	Misc. species[a]
	percentage of total stems				
Preharvest slash	36	16	21	8	20
Preharvest slash + herbicide	42	23	12	8	15
Post-harvest slash	36	14	16	11	23
Post-harvest slash + herbicide	36	12	13	9	31
Control - no site preparation	50	10	13	7	21

[a]Miscellaneous species include beech, blackgum, dogwood, elms, hollies (*Ilex* spp.), sassafras [*Sassafras albidum* (Nutt.) Nees], sourwood, sweetgum, pines, sumac (*Rhus* spp.), and white ash (*Fraxinus americana* L.).

harvested stand. Most of the oak advance reproduction in this study was < 2-feet tall before harvest. The probability of this diminutive oak advance reproduction becoming an overstory tree is < 2 percent (Loftis 2004).

Most of the miscellaneous species, though a component of the overstory at 16 years, will probably diminish with their slower growth capacity. The exception is the pines, primarily Virginia (*P. virginiana* Mill.) and loblolly (*P. taeda* L.) pines regenerating from wind-blown seed.

These pine stems are few in number but of larger diameters. The REGEN model estimated a lower proportion of miscellaneous species at crown closure than the actual field data after 16 years.

Preharvest site preparation treatments compared to post-harvest treatments had little effect on species composition (tables 1 and 2). Seasonality of when these treatments were conducted was similar, October for the preharvest site preparation and August the following year for the post-harvest. More than three times as many stems were treated in the preharvest plots than recorded in the post-harvest plots (Clatterbuck and Schubert 2010). The harvesting operation resulted in many of the stems in the post-harvest plots being severed before the treatment being applied. Thus, the miscellaneous species category for post-harvest with herbicide was greater because the herbicide was not applied to those stems impacted by the harvest. The preharvest treatments were more costly to conduct than the post-harvest because more stems were treated in the pre-harvest treatments (Hodges and others 2002).

FUTURE CONSIDERATIONS

If the REGEN model is to be used effectively in the Ridge and Valley of east Tennessee, the black cherry rankings and probabilities in the model should be adjusted accordingly to reflect conditions of the study area. The southern Appalachian knowledge base was used in this study and probably reflects conditions in the area where it was developed rather than the Ridge and Valley. The proportion of black cherry in the overstory at canopy closure was often overestimated by the model compared to the actual data. Familiarity with the model and the knowledge base is necessary to make species modifications in the model. Practitioners should realize the silvics of a particular species differs gradually from locality to locality and region to region. The silvics of black cherry in western Pennsylvania is different from the southern Appalachians of North Carolina and different from the Piedmont. The knowledge base of the model should reflect those local conditions where the model is applied.

Hardwood regeneration models are complex and difficult to develop because of the variation in species, sites, reproductive sources, regeneration methods, competitive interactions among and between species, and temporal and spatial scales. Even with this inherent variation, Vickers and others (2011) suggested that model development and information or knowledge contained within the model are responsible for most of the divergence involved between predicted and actual values. The discrepancies encountered in this study, especially those with estimating the proportion of black cherry in the overstory at crown closure, reflect inconsistencies in the knowledge base for black cherry in the Ridge and Valley area. Otherwise, the REGEN model provided reasonable predictions of species composition at canopy closure when compared to actual data 16 years after harvest.

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LITERATURE CITED

Boucugnani, D.A. 2005. REGEN AGENT: a modular and generalized forest regeneration agent. Athens, GA: University of Georgia. 57 p. M.S. thesis.

Clark, F.B.; Boyce, S.G. 1964. Yellow-poplar seed remains viable in the forest litter. Journal of Forestry. 62: 564-567.

Clatterbuck, W.K.; Schubert, M.R. 2010. Effects of pre- and post-harvest site preparation treatments on natural regeneration success in a mixed hardwood stand after 10 years. 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: 451-454.

Hodges, D.G.; Evans, R.L.; Clatterbuck, W.K. 2002. Comparing alternative slashing techniques on a mixed hardwood forest: 2-year results. 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: 412-414.

Loftis, D.L. 1978. Preharvest herbicide control of undesirable vegetation in southern Appalachian hardwoods. Southern Journal of Applied Forestry. 2: 51-54.

Loftis, D.L. 1985. Preharvest herbicide treatment improves regeneration of southern Appalachian hardwoods. Southern Journal of Applied Forestry. 9: 177-180

Loftis, D.L. 1989. Species composition of regeneration after clearcutting. In: Miller, J.H., comp. Proceedings of the 5th biennial southern silvicultural research conference. Gen. Tech. Rep. SO-74. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 253-257.

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.

Moneymaker, R.H. 1981. Soil survey of Anderson County, Tennessee. Washington, DC: U.S. Department of Agriculture Soil Conservation Service. 165 p.

Olson, D.J., Jr. 1959. Site index curves for upland oak in the southeast. Res. Note SE-126. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 2 p.

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: 790-822.

Wendel, G.W. 1977. Longevity of black cherry, wild grape, and sassafras seed in the forest floor. Res. Pap. NE-375. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 6 p.

MULTI-SCALE MODELING OF RELATIONSHIPS BETWEEN FOREST HEALTH AND CLIMATIC FACTORS

Michael K. Crosby, Zhaofei Fan, Xingang Fan, Martin A. Spetich, and Theodor D. Leininger[1]

Abstract--Forest health and mortality trends are impacted by changes in climate. These trends can vary by species, plot location, forest type, and/or ecoregion. To assess the variation among these groups, Forest Inventory and Analysis (FIA) data were obtained for 10 states in the southeastern United States and combined with downscaled climate data from the Weather Research and Forecasting (WRF) model. A variable was created for analysis at the intersection of ecoregions, climate divisions, and forest type. Spatial autoregressive (SAR) modeling was employed to determine if mortality patterns over two inventory cycles were clustered and differed with climate variables. Models were developed showing the relationship between mortality and a series of climate indicators. This information could prove useful to forest managers if projected climate changes are verified.

INTRODUCTION

A variety of factors contribute to forest health and mortality. These factors can be biotic (e.g., insects, diseases, etc.), abiotic (e.g., drought, temperature, etc.), or a combination of the two (drought stress leading to insect infestation) (Crosby and others 2012, Fan and others 2012). However, the impacts of these factors may not be continuous across the landscape. Patterns of change in forest health and mortality can vary by species groups or forest type. Further, there may be a time lag effect between the onset of disturbance factors (e.g., drought) and the impacts on forested areas (Fan and others 2012). Analysis of relationships between climatic factors and forest health indicators will allow for model development to account for spatial clusters of mortality. In an effort to develop predictive models our objectives were to: (1) assess mortality (using percentages of dead basal area) trends for the southeastern United States; and (2) determine whether there is a relationship between mortality and climate variables during forest inventory cycles.

METHODS

Data for this study were obtained from a variety of sources. Forest Inventory and Analysis (FIA) data were obtained from the USDA Forest Service (http://apps.fs.fed.us/fiadb-downloads/datamart.html) for 10 southeastern states (Alabama, Arkansas, Florida, Georgia, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, and Texas). Drought data, i.e. Palmer's Drought Severity Index (PDSI), were obtained from the National Climate Data Center (http://www7.ncdc.noaa.gov/CDO/CDODivisionalSelect.jsp). Average annual temperature and average annual temperature range were derived from downscaled Weather Research and Forecasting data. To effectively analyze the relationship between mortality and climatic factors, polygons were created by intersecting Bailey's ecoregions, a forest type map, and climate divisions in the southeastern United States (fig. 1).

Variables were extracted based upon the polygons in figure 1 (percent dead basal area, growing season PDSI, average annual temperature, and annual temperature range) for each of two FIA inventory cycles (cycle 1 from 2000-2004 and cycle 2 from 2005-2009). Variables for both inventory periods were divided into periods of drought/non-drought (based on PDSI values) prior to and during the inventory period (table 1). Spatial autoregressive (SAR) modeling was then utilized to determine the relationship between percent dead basal area and the climate variables. The SAR model is defined as:

$$Y = X\beta + \varepsilon \qquad (1)$$

[1]Assistant Professor, Shorter University, Rome, GA 30165; Associate Professor, Mississippi State University, Forest and Wildlife Research Center, Mississippi State, MS 39762; Assistant Professor, Western Kentucky University, Department of Geography and Geology, Bowling Green, KY 42101-1066; Research Forest Ecologist, USDA Forest Service, Southern Research Station, Hot Springs, AR 71902; and Supervisory Research Plant Pathologist, USDA Forest Service, Southern Research Station, Stoneville, MS 38776-0227.

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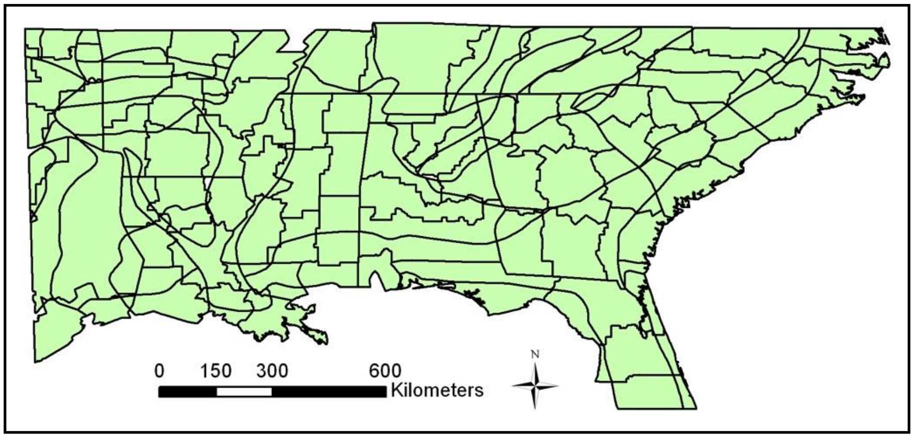

Figure 1--Study area depicting polygons created as a result of over-laying climate divisions, ecoregions, and forest type maps.

$$\varepsilon = \rho W \varepsilon + \upsilon \quad (2)$$

Where: Y = percentage of dead basal area; X = PDSI, average annual temperature, and annual temperature range; β_i = regression coefficients to be estimated; v = independent error vector (assumed normally distributed); ρ = SAR error coefficient; and W = spatial weight matrix. Akaike's Information Criteria (AIC) was used to select the "best" model and Nagelkerke's pseudo R^2 is used to assess goodness of fit for each SAR model.

Table 1--Variable definitions for the inventory cycles used in analysis where D represents periods of drought and ND represents periods of non-drought

Cycle 1 (2000-2004)	Cycle 2 (2005-2009)
D1=1999	D1=2006-2008
D2=2000-2002	D2=1999-2002
ND1=1994-1998	ND1=2005
ND2=2003-2004	ND2=2003-2004
	ND3=1994-1998
	ND4=2009

RESULTS AND DISCUSSION

Cycle 1 (2000-2004)

The percentages of dead basal area for the created polygons show values that are high across vast portions of the southeastern United States [fig. 2(a)]. The relationship between mortality and climate variables are clustered in portions of eastern Texas and western Louisiana and across portions of central South Carolina [fig. 2(b)]. Results from the SAR model show that the most significant variables for selecting the best model for cycle 1 are diameter at breast height (d.b.h.), PDSI in 1999, and PDSI from 2000-2002 (table 2). This model indicates that drought periods prior to and during the inventory cycle and d.b.h. are related to mortality. The D1 drought period (table 1) only considers 1 year of drought conditions prior to measurement, which could lead to the confounding result of higher PDSI values (i.e., non-drought conditions) being related to more mortality. The D2 period (table 1) indicates a negative relationship, where a lower PDSI (i.e., more severe drought) would be related to higher mortality as has been found by previous studies (Fan and others 2012).

Cycle 2 (2005-2009)

The percent dead basal area for inventory cycle 2 shows an altogether different pattern from that in cycle 1. The greatest values found in cycle 2 occurred across eastern Texas, central Arkansas, Tennessee, northern Georgia, and western portions of North and South Carolina [fig. 3(a)]. The Moran's I result [fig. 3(b)] indicates clusters in western Louisiana, eastern Arkansas/western Texas, and across portions of Florida, North Carolina, and South Carolina. Accounting for these clusters, the SAR model for cycle 2 had more significant variables than cycle 1 (table 3). D.b.h., height, PDSI, average annual temperature, and average temperature range all proved significant. While the model selected had

Table 2--Variables and coefficients from the best SAR model for inventory cycle 1[a]

	Estimate	Std. Error	z value	Pr(>lzl)
(Intercept)	0.045778	0.024856	1.842	0.065511
DIA	0.015508	0.004459	3.478	0.000505
PDSI(D1)	0.022962	0.009414	2.439	0.014722
PDSI(D2)	-0.032851	0.010241	-3.208	0.001337

[a]AIC: -228.47; R^2 = 0.511.

Table 3--Variables and coefficients from the best SAR model for inventory cycle 2[a]

	Estimate	Std. error	z value	Pr(>|z|)
(Intercept)	-0.352817	0.087946	-4.0117	0.000060
DIA	0.030346	0.002287	13.267	< 2.2e-16
HT	0.000945	0.000446	2.1185	0.034129
PDSI(D1)	0.010183	0.002528	4.0277	0.000056
PDSI(D2)	-0.006817	0.002919	-2.3352	0.019531
PDSI(ND3)	0.006016	0.003453	1.7426	0.081411
Avg temp(ND2)	0.010114	0.002677	3.7788	0.000158
Temp range (D1)	0.034717	0.007482	4.6401	0.000003
Temp range (D2)	-0.039571	0.006700	-5.9058	0.000000
Temp range (ND4)	0.010769	0.004680	2.3012	0.021380

[a]AIC: -513.55; R^2= 0.824

the best fit, the individual variables proved somewhat confounding. For example, greater PDSI values indicate non-drought conditions but the relationship would indicate that this condition led to increased mortality. Further analysis and model refinement will be necessary to critically analyze such relationships. Of note, however, are the PDSI and annual temperature range relationships with mortality in the drought period prior to inventory measurement. These relationships indicate that a larger temperature range (increased high temperatures or decreased low temperatures) and more severe drought conditions (lower PDSI values) are related to increased mortality. These conditions indicate that extreme temperatures (higher temperature ranges) and drought could act to stress trees, which may not succumb for a period of several years, indicating a lag effect previously discussed (Fan and others 2012), although further analysis is required.

It is difficult to compare cycle 1 and cycle 2 aside from the general trend that greater mortality levels were detected in northern portions of the region in cycle 2. In-growth was not accounted for in the analysis which could, at least partially, explain the lower percent dead basal area results for cycle 2. Drought was indicated to have played a role in greater mortality in both inventory cycles, and the relationship between mortality and the selected climate variables seem to be clustered in certain areas. These areas could be selected for further study to provide insight into local stand/site factors that could be contributing to these relationships although more intense data collection would be required. These preliminary results could indicate areas that should be monitored as drought conditions become apparent. Future research will seek to refine the analysis presented here to present more detailed information (e.g., models for species groups). Also, this study only utilized data from trees that were found to have died from competition; such conditions as drought and high temperature ranges could act to stress trees and lead to the infestation by insects and increase fuel loads for forest fires.

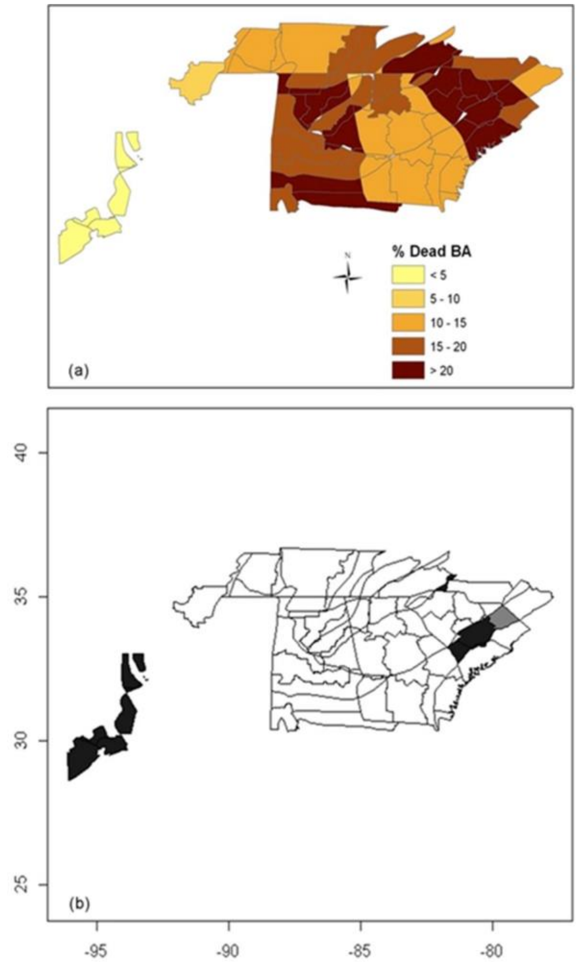

Figure 2--Pattern of percent dead basal area (a) and Moran's I results showing spatial clusters of mortality (b) for inventory cycle 1.

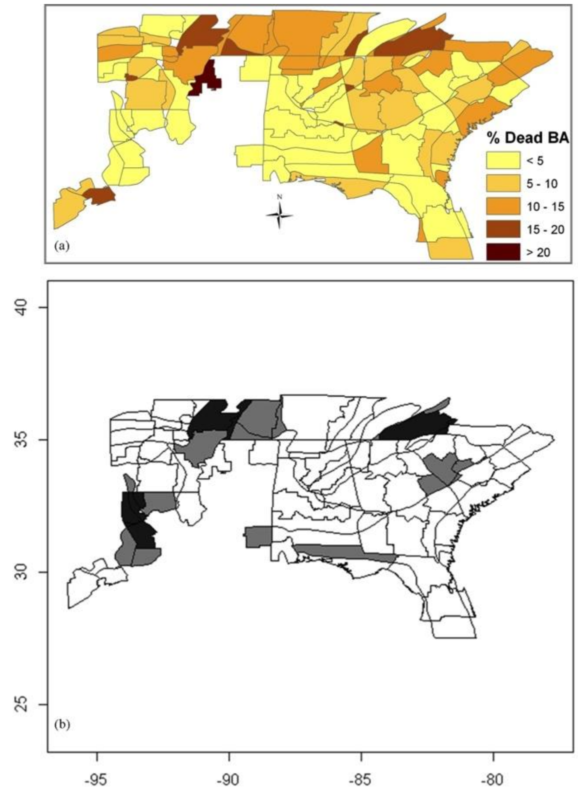

Figure 3--Pattern of percent dead basal area (a) and Moran's I results showing spatial clusters of mortality (b) for inventory cycle 2.

CONCLUSIONS

While the two inventory cycles had different values for mortality (percent dead basal area), there were significant clusters of mortality. Both cycles exhibited significant clusters across portions of Texas, Louisiana, and South Carolina. The spatial autoregressive model proved useful for accounting for spatial relationships among variables (clusters) in model development. The models developed show a relationship between mortality and biometric (e.g., d.b.h.) and climatic factors (drought and temperature range). While this is not a novel discovery, it is interesting that the polygons analyzed were similarly clustered for both inventory cycles. This suggests that these variables are interacting in a similar manner though a more thorough analysis and treatment of the variables is warranted. This study provides a good basis for the further examination of spatial clusters which could yield more robust models for the prediction of the relationship between mortality and climate across the region.

ACKNOWLEDGMENTS

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LITERATURE CITED

Crosby, M.K.; Fan, Z.; Fan, X. [and others]. 2012. Assessing forest mortality patterns using climate and FIA data at multiple scales. In: Morin, R.S.; Liknes, G.C., comps. Moving from status to trends: Forest Inventory and Analysis (FIA) symposium. [CD-ROM]. Gen. Tech. Rep. NRS-P-105. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 478 p.

Fan, Z.; Fan, X.; Crosby, M.K. [and others]. 2012. Spatio-temporal trends of oak decline and mortality under periodic regional drought in the Ozark Highlands of Arkansas and Missouri. Forests. 3(3): 614-631.

CULTURAL INTENSITY AND PLANTING DENSITY EFFECTS ON INDIVIDUAL TREE STEM GROWTH, STAND AND CROWN ATTRIBUTES, AND STAND DYNAMICS IN THINNED LOBLOLLY PINE PLANTATIONS DURING THE AGE 12- TO AGE 15-YEAR PERIOD IN THE UPPER COASTAL PLAIN AND PIEDMONT OF THE SOUTHEASTERN UNITED STATES

Evan Johnson, Michael Kane, Dehai Zhao, and Robert Teskey[1]

Three existing loblolly pine (*Pinus taeda* L.) installations in the Plantation Management Research Cooperative's Upper Coastal Plain/Piedmont Culture Density Study were used to examine the effects of two cultural intensities, four initial planting densities, and their interactions on stem growth at the individual tree level from age 12 to 15 years and at the stand level in thinned plantations during the age 13 to 15 year period. These plots were also studied to determine how treatments affected crown characteristics such as leaf area index (LAI), specific leaf area (SLA), intercepted photosynthetically active radiation (IPAR), and foliar nitrogen concentration and content. The two cultural intensities include maximum intensity, which consisted of sustained competing vegetation control and frequent fertilization and operational intensity, which consisted of less frequent competing vegetation control and fertilization.

The four planting densities were 740; 1,480; 2,220; and 2,960 trees ha^{-1}. Each installation of three planting densities (1,480; 2,220; and 2,960 trees ha^{-1}) was thinned during the age 12 dormant season. These plots were thinned to the same trees ha^{-1} on the 740 trees ha^{-1} density sites with corresponding cultural intensities.

There were no significant effects of culture or culture x density treatments on any individual tree-level or stand-level attribute (table 1) measured during the 14th or 15th year. Density had a significant effect on many post-thin attributes. Crown attributes were mainly stable during the time period analyzed. The general stand-level density trends were that trees planted at 740 trees ha^{-1} were largest, followed by 1,480 trees ha^{-1}; then 2,220 trees ha^{-1}; and finally 2,960 trees ha^{-1}. Stand level-stem attributes in thinned plots continued to increase and approach attribute measures on the non-thinned low-density counterpart (fig. 1). No significant effects were detected on growth efficiency measures, but growth efficiencies were greater under operational culture when soil nutrition was sufficient. Individual tree-level diameter at breast height growth post-thinning was greater in higher densities but declined in the highest density, due to the initial size of individual trees relative to others in plots of the same density and also due to ongoing crown development in the highest density at age 15. These results indicate that when site resource availability is adequate or abundant, density has greater effects on stem, crown, and growth efficiency attributes than culture on these installations during the time period analyzed.

[1]Graduate Research Assistant, Professor, Research Scientist, and Distinguished Research Professor, respectively, University of Georgia, Warnell School of Forestry and Natural Resources, Athens, GA 30602.

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Table 1—P-values for the effects of culture, density, and their interactions on mean stem, crown, and efficiency attributes on three thinned loblolly pine installations during the 14th and 15th growing seasons and at ages 14 and 15

	--14th growing season and age 14--			--15th growing season and age 15--		
	Culture	Density	Interaction	Culture	Density	Interaction
Stem attributes						
D.b.h.	0.1466	0.0001	0.3808	0.1240	<0.0001	0.5263
Total stem height	0.3886	0.3695	0.5261	0.3741	0.3154	0.2639
Total stem volume ha^{-1}	0.2000	0.0001	0.5621	0.1824	<0.0001	0.7811
BA ha^{-1}	0.1435	0.0001	0.5258	0.1273	<0.0001	0.6227
Current density	0.3315	0.3317	0.6842	0.3266	0.2196	0.7295
Gross CAI	0.1929	0.0055	0.5615	0.1917	0.5225	0.1643
New CAI	0.1816	0.9149	0.8455	0.3239	0.4117	0.5504
Crown attributes						
Live crown length	0.8100	0.0003	0.9993	0.2294	0.0541	0.3806
Crown ratio	0.2170	0.0002	0.8255	0.9644	0.1399	0.1998
SLA	0.2788	0.4355	0.3681	---	---	---
Foliar N concentration	0.2959	0.0001	0.1404	---	---	---
Foliar biomass	0.2312	0.0001	0.7527	0.3364	0.0026	0.6889
LAI	0.1816	0.0001	0.6526	0.2419	0.0038	0.6551
Foliar N content	0.0700	0.0001	0.5277	0.2614	0.0001	0.6453
IPAR	---	---	---	0.1317	0.0008	0.1252
Efficiency attributes						
GE_{FOLBIO}	0.4585	0.4919	0.2958	0.6926	0.2018	0.5126
GE_{LAI}	0.3754	0.6887	0.5890	0.6076	0.2152	0.6315
NUE	0.1531	0.3110	0.3400	0.5707	0.0730	0.6334
IPAR efficiency	0.2194	0.0001	0.0904	0.2719	0.3958	0.1749

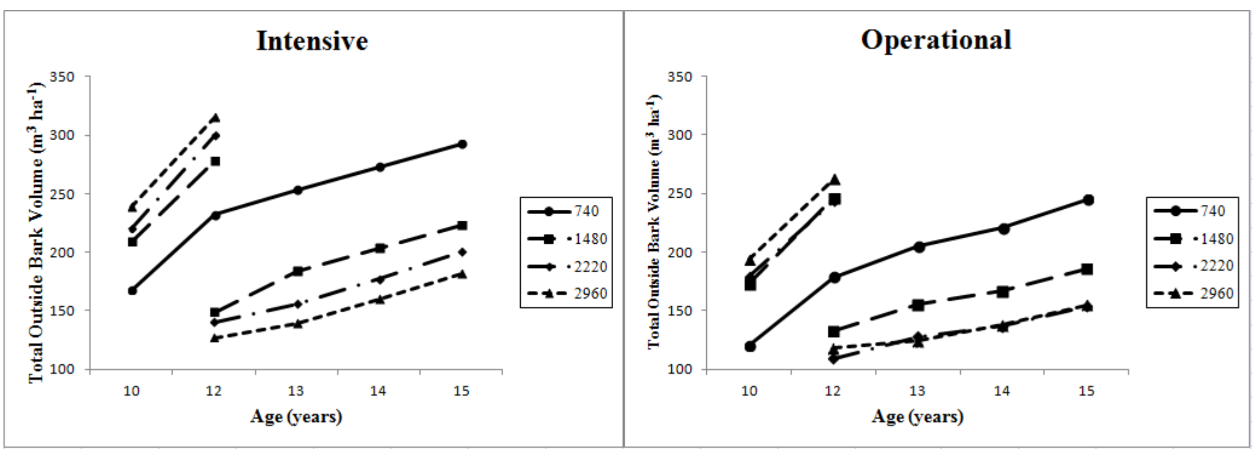

Figure 1--Total volume in thinned installations from age 10 to age 15.

CROWN SIZE RELATIONSHIPS FOR BLACK WILLOW IN THE LOWER MISSISSIPPI ALLUVIAL VALLEY

Jamie L. Schuler, Bradley Woods, Joshua Adams, and Ray Souter[1]

Abstract--Growing space requirements derived from maximum and minimum crown sizes have been identified for many southern hardwood species. These requirements help managers assess stocking levels, schedule intermediate treatments, and even assist in determining planting densities. Throughout the Mississippi Alluvial Valley, black willow (*Salix nigra* Marsh.) stands are common along major rivers as single-species stands. Management of these stands is limited in part by the lack of information regarding crown size-stem density relationships. We developed equations to predict maximum crown sizes from open-grown black willow trees and minimum crown sizes from natural, even-aged black willow stands. This information is being utilized to construct stocking guides for thinning treatments and provide preliminary planting density guidelines for plantations based on a target diameter size. Our initial results show that a polynomial tree area equation predicted the maximum density data well; however, the data associated with the development of the maximum crown area equation (leading to the average minimum density line) were much more variable.

INTRODUCTION

Black willow (*Salix nigra* Marsh.) is a common species occupying alluvial sites throughout major river bottoms in the Lower Mississippi Alluvial Valley. Flooding along these river systems lays down new sediments, which are often colonized by black willow. As a result, new stands typically occur as "bands" ranging from tens to hundreds of feet wide. Under these conditions, black willow stands are often single-species, but depending on the amount of soil deposition, they may co-occur with eastern cottonwood (*Populus deltoides* Bartram ex. Marsh.) and/or sandbar willow (*Salix interior* Rowlee). Following initial colonization, stand development proceeds rapidly. Newly established stands may have 10,000 or more stems per acre. Stem density declines rapidly, as willow is very shade intolerant. On the best sites, black willow may grow 5 feet per year in height and 1 inch in diameter at breast height (d.b.h.) (Lamb 1915), although these growth rates decrease as stands age.

The scant evidence available suggests black willow responds positively to thinning (Johnson and McKnight 1969). However, information related to stocking is absent in the literature. No guidelines are available for predicting relative density or stocking percent in order to objectively manage black willow stands.

The goal of this study was to quantify the maximum and minimum areas that black willow occupies for a given d.b.h. (i.e. tree areas). This information can then be used to develop density management diagrams to predict upper and lower bounds for maximum and minimum stocking levels.

METHODS

Tree area ratios were used to predict growing space requirements for stands growing at their maximum density for a given d.b.h. (Chisman and Schumacher 1940). Plots were established in stands where black willow was the dominant species, canopy gaps were absent, and mortality present to indicate the area contained its maximum density. Data were collected from 23 stands located near the confluence of the Arkansas, White, and Mississippi rivers. The number (1 to 7) and size (0.01 to 0.2 acre) of measurement plots per stand varied depending on stand conditions and tree size, with small plots used in stands with small average diameters.

At each plot, d.b.h. and species for all trees > 1 inch d.b.h. were recorded. To facilitate stand development studies and additional research objectives, the location of all plot centers was recorded using GPS, and each tree was permanently tagged with a unique number for future monitoring. A tree area (TA) equation, originally developed by Chisman and Schumacher (1940) to estimate the area occupied by trees growing at their maximum density, was fit to the data in the following form:

$$TA = b_0N + b_1\Sigma D + b_2\Sigma D^2 \quad (1)$$

[1]Assistant Professor, West Virginia University, Division of Forestry and Natural Resources, Morgantown, WV 26506; Graduate Research Assistant and Assistant Professor, respectively, University of Arkansas-Monticello, School of Forest Resources, Monticello, AR 71656; and Research Forester, USDA Forest Service, Southern Research Station, Stoneville, MS 38776.

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where b_0, b_1, and b_2 are regression coefficients, N is trees per unit area, and D is tree d.b.h.

To determine the average minimum density (minimum number of trees for a given diameter that can fully occupy a unit area), we measured willows that were open-grown without competition. Crown width (CW) was estimated as the average of the north-south and east-west crown extents and recorded along with d.b.h. Crown width was then predicted as a linear function of d.b.h.:

$$CW = b_0 + b_1 D \quad (2)$$

where CW is in feet and all other parameters are as previously defined.

Maximum crown area or TA_{max} was estimated using the following equation outlined by Krajicekand others (1961):

$$TA_{max} = \pi (CW/2)^2 \div 43.56 \quad (3)$$

where TA_{max} is in milacres.

Once expanded the equation takes on the form:

$$TA_{max} = b_0 + b_1 D + b_2 D^2 \quad (4)$$

RESULTS AND DISCUSSION

A tree area ratio equation was used to predict area required for a tree of a given size in fully stocked stands (TA_{min}). The equation for stands with quadratic mean diameter (QMD) $\geq$ 4 inches was highly significant ($P < 0.0001$):

$$TA_{min} = -1.66 + 0.52375D + 0.00005480788D^2 \quad (5)$$

where TA_{min} is tree area in milacres and D is d.b.h. in inches.

The curve for black willow was more linear than curves for several other bottomland hardwood species (fig. 1). Black willow was most closely aligned with equations for cottonwood, silver maple (*Acer saccharinum* L.), and sycamore (*Platanus occidentalis* L.) (Larsen and others 2010). The application of the bottomland hardwood equations developed by Goelz (1995) would severely over-predict tree areas and substantially under-predict stocking for black willow (fig. 1).

Crown width of open-grown trees was a linear function of d.b.h. Data points were dispersed around the regression line that predicted tree area using d.b.h. ($r^2 = 0.68$, fig. 2). The fit for our equation was relatively poor compared to many published crown-width equations, where d.b.h. often accounts for more than 90 percent of the variation (e.g., Johnson and others 2009, Krajicek and others 1961). However, black willow branches are brittle and susceptible to breakage during storm events (e.g., Seischab and others 1993), which may partially account for the poor fit.

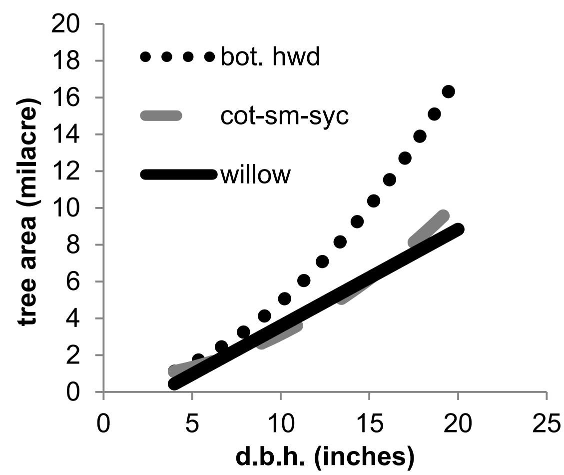

Figure 1--Minimum tree area curve for black willow compared to published equations for bottomland hardwoods (bot. hwd) (Goelz 1995) and cottonwood-silver maple-sycamore (cw-sm-syc) (Larsen and others 2010).

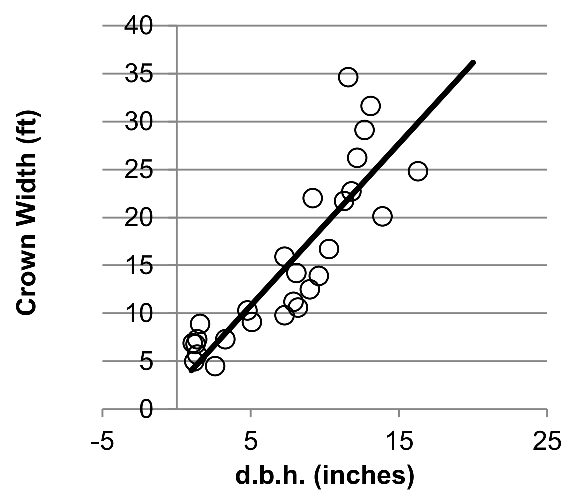

Figure 2--Predicted crown width for open grown black willow stems.

The TA_{max} was calculated as:

$$TA_{max} = 0.1002 + 0.1436D + 0.05143D^2 \quad (6)$$

Maximum tree areas were about 70 to 170 percent > the minimum tree areas for 6- to 20-inch d.b.h. trees, respectively (fig. 3). For black willow stands, this difference amounts to 355 and 673 tree per acre for 6-inch d.b.h. trees, 220 and 395 trees per acre for 8-inch d.b.h. trees, 150 and 279 trees per acre for 10-inch d.b.h. trees, 108 and 216 trees per acre for 12-inch d.b.h. trees, 82 and 176 trees per acre for 14-inch d.b.h. trees, and 64 and 148 trees per acre for 16-inch d.b.h. trees.

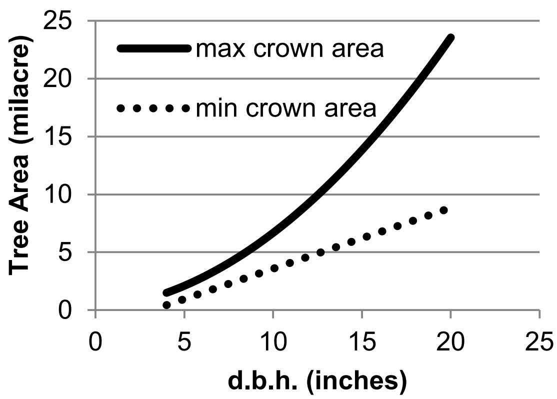

Figure 3--A comparison of minimum and maximum tree curves for black willow.

These data can be used by managers to assess bounds for managing stem density for thinning activities. Similarly, the maximum tree areas can be used to estimate planting density. For example, if a particular management objective was to grow 8-inch d.b.h. black willow trees, the maximum planting density assuming no thinning would be 395 trees per acre, or a plant spacing of 10- by 11-feet.

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LITERATURE CITED

Chisman, H.H.; Schumacher, F.X. 1940. On the tree-area ratio and certain of its applications. Journal of Forestry. 38: 311-317.

Goelz, J.C.G. 1995. A stocking guide for southern bottomland hardwoods. Southern Journal of Applied Forestry. 19: 103-104.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2009. The ecology and silviculture of oaks. 2[d] ed. Wallingford, UK: CABI Publishing, CAB International. 580 p.

Johnson, R.L.; McKnight, J.S. 1969. Benefits from thinning black willow. Res. Note SO-89. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 6 p.

Krajicek, J.E.; Brinkman, K.A.; Gingrich, S.F. 1961. Crown competition: a measure of density. Forest Science. 7: 35-42.

Lamb, G.N. 1915. Willows: their growth, use, and importance. Bull. No. 316. Washington, DC: U.S. Department of Agriculture. 52 p.

Larsen, D.R.; Dey, D.C.; Faust, T. 2010. A stocking diagram for midwestern eastern cottownood-silver maple-American sycamore bottomland forests. Northern Journal of Applied Forestry. 27(4): 132-139

Seischab, F.K.; Bernard, J.M.; Eberle, M.D. 1993. Glaze storm damage to western New York forest communities. Bulletin of the Torrey Botanical Club. 120(1): 64-72.

Growth and Development

Moderator:

Dean Coble
Stephen F. Austin State University
Arthur Temple College of Forestry and Agriculture

LOBLOLLY PINE FOLIAR PATTERNS AND GROWTH DYNAMICS AT AGE 12 IN RESPONSE TO PLANTING DENSITY AND CULTURAL INTENSITY

Madison Katherine Akers, Michael Kane, Dehai Zhao, Richard F. Daniels, and Robert O. Teskey[1]

Examining the role of foliage in stand development across a range of stand structures provides a more detailed understanding of the processes driving productivity and allows further development of process-based models for prediction. Productivity changes observed at the stand scale will be the integration of changes at the individual tree scale, but few studies have analyzed crown attributes at the individual tree level. Studies analyzing loblolly pine (*Pinus taeda* L.) stand response to common silvicultural practices such as fertilization, control of competing vegetation, and density management are numerous. However, the physiological mechanisms that drive this response are not thoroughly understood (Jokela and others. 2004, King and others 2008, Tyree and others 2009, Will and others 2005).

Four Plantation Management Research Cooperative (PMRC) study installations were utilized to analyze the effects of planting density and cultural intensity on individual tree stem and crown attributes in non-thinned loblolly pine plantations in the Upper Coastal Plain and Piedmont of Georgia and Alabama. Treatments included six planting densities (740; 1,480; 2,220; 2,960; 3,700; and 4,440 trees ha^{-1}), in split-plot design with two cultural treatments (maximum and operational) that included different levels of fertilization and competition control (table 1). Treatment effects on stem and crown attributes were analyzed at age 12 using destructive sampling techniques.

Trees planted at lower densities were able to maintain larger crowns (increased live crown length, live crown width, leaf area, crown density). Less intra-specific competition in the lower planting density stands allowed for more light to reach the lower branches. Because leaf area is representative of photosynthetic surface area, it is assumed that individual trees planted at the lower densities were intercepting more light, allowing for increases in individual stem growth at the lower planting densities. Carlson and others (2009) and MacFarlane and others (2002) found similar patterns for loblolly pine planted at different densities. Specific leaf area (SLA), however, was generally lower in the lower planting density stands. SLA is typically greater under more light-limited conditions, resulting in more photosynthetic surface area per unit of needle biomass (longer, thinner needles), which may help mitigate the effects of increased shading present in densely stocked stands (Samuelson and others 2008, 2010; Will and others 2001).

Although the maximum cultural regime provided more frequent fertilization and competition control relative to the operational cultural regime, the operational treatment still provided considerable inputs (e.g. chemical competition control at planting and three fertilizer treatments). Although average nitrogen (N) concentration was lower for the trees grown under operational culture, it was still above the critical level of 1.10 percent for loblolly pine (Allen 1987), suggesting that loblolly pine nutrition was not markedly deficient for either treatment at age 12. Increases in foliar N concentration do not lead to a consistent observable increase in photosynthetic capacity for loblolly pine (Munger and others 2003). Additional N acquired by the foliage, however, may serve as a source for subsequent foliage development, which may consequentially drive additional stem growth (Borders and others 2004, Munger and others 2003, Tyree and others 2009, Will and others 2002). Although foliar N concentration significantly differed

[1]Research Professional, Professor, Assistant Research Scientist, and Professors, respectively, University of Georgia, Warnell School of Forestry and Natural Resources, Athens, GA 30602.

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Table 1--Description of operational and maximum cultural treatments on the PMRC culture x planting density study

Site preparation treatment	Growing season	---------------Operational--------------- Chemical and mechanical	------------------Maximum------------------ Chemical and mechanical
Fertilization	at planting	560 kg ha^{-1} 10-10-10	560 kg ha^{-1} 10-10-10
	2nd		673 kg ha^{-1} 10-10-10 + 131 kg ha^{-1} NH_4NO_3 + micronutrients
	4th		131 kg ha^{-1} NH_4NO_3
	6th		336 kg ha^{-1} NH_4NO_3
	8th	224 kg ha^{-1} N + 28 kg ha^{-1} P	224 kg ha^{-1} N + 28 kg ha^{-1} P
	10th		224 kg ha^{-1} N + 28 kg ha^{-1} P
	12th	224 kg ha^{-1} N + 28 kg ha^{-1} P	224 kg ha^{-1} N + 28 kg ha^{-1} P
Competition control (chemical)	1st	280 g ha^{-1} sulfometuron-methyl banded application + glyphosate and tryclopyr direct spraying	280 g ha^{-1} sulfometuron-methyl broadcast application + glyphosate and tryclopyr direct spraying
	2nd		841 g ha^{-1} imazapyr broadcast application
	3rd through 12th		Glyphosate and tryclopyr repeated direct spraying

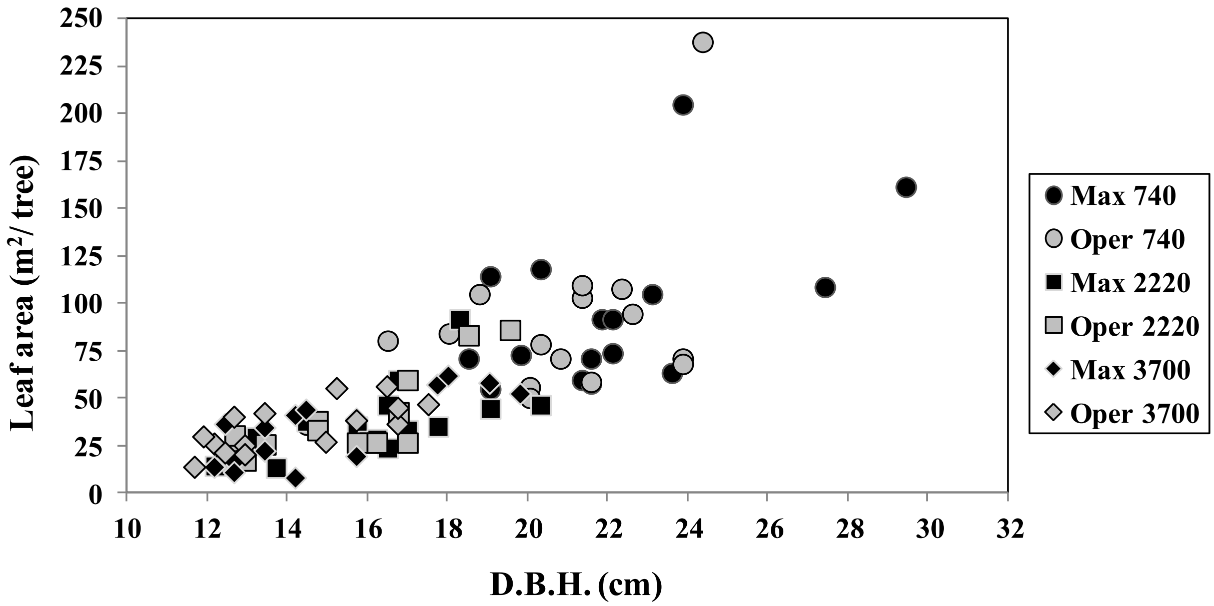

Figure 1--Relationship between individual tree d.b.h. and leaf area for destructively sampled trees on four PMRC loblolly pine installations at age 12. A sub-set of the planting density and culture treatment combinations is displayed for simplicity. The circle, square, and diamond symbols represent the 740; 2,220; and 3,700 trees ha^{-1} planting densities, respectively. The black symbols represent the maximum cultural intensity and the gray symbols represent the operational cultural intensity.

among planting densities, there was no consistent trend. Foliar N content per tree decreased with increasing planting density, primarily because N content is strongly related to foliar biomass.

The results suggest that trees of a given diameter at breast height (d.b.h.) had similar crown characteristics regardless of the silvicultural treatments they received (fig. 1). This does not suggest that culture or planting density do not affect crown characteristics, but rather that from a modeling point of view, the difference in crown characteristics (e.g. leaf area) can be adequately accounted for by variation in d.b.h., without explicitly considering further culture or planting density. It should be noted that this interpretation is limited to the age, genetics, locations, and treatments used in this study. Albaugh and others (2006) found similar results for loblolly pine grown on a nutrient poor, well-drained sandy soil in the Sandhills of North Carolina. Although fertilization had a significant effect on average d.b.h., stem height, and foliar mass, the fertilization effect was not significant in a model predicting individual tree foliar biomass where tree size (stem volume) was an independent variable (Albaugh and others 2006).

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.

Allen, H.L. 1987. Forest fertilizers. Journal of Forestry. 85(2): 37-46.

Borders, B.E.; Will, R.E.; Markewitz, D. [and others]. 2004. Effect of complete competition control and annual fertilization on stem growth and canopy relations for a chronosequence of loblolly pine plantations in the lower coastal plain of Georgia. Forest Ecology and Management. 192(1): 21-37.

Carlson, C.A.; Fox, T.R.; Creighton, J. [and others]. 2009. Nine-year growth responses to planting density manipulation and repeated early fertilization in a loblolly pine stand in the Virginia Piedmont. Southern Journal of Applied Forestry. 33: 109-114.

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(1): 117-130.

King, N.T.; Seiler, J.R.; Fox, T.R.; Johnsen, K.H. 2008. Post-fertilization physiology and growth performance of loblolly pine clones. Tree Physiology. 28(5): 703-711.

MacFarlane, D.W.; Green, E.J.; Brunner, A.; Burkhart, H.E. 2002. Predicting survival and growth rates for individual loblolly pine trees from light capture estimates. Canadian Journal of Forest Research. 32(11): 1970.

Munger, G.T.; Will, R.E.; Borders, B.E. 2003. Effects of competition control and annual nitrogen fertilization on gas exchange of different-aged *Pinus taeda*. Canadian Journal of Forest Research. 33(6): 1076.

Samuelson, L.J.; Butnor, J.; Maier, C. [and others]. 2008. Growth and physiology of loblolly pine in response to long-term resource management: defining growth potential in the southern United States. Canadian Journal of Forest Research. 38(4): 721-732.

Samuelson, L.J.; Eberhardt, T.L.; Butnor, J.R. [and others]. 2010. Maximum growth potential in loblolly pine: results from a 47-year-old spacing study in Hawaii. Canadian Journal of Forest Research. 40(10): 1914-1929.

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(9): 1117-1131.

Will, R.E.; Barron, G.A.; Colter Burkes, E. [and others]. 2001. Relationship between intercepted radiation, net photosynthesis, respiration, and rate of stem volume growth of *Pinus taeda* and *Pinus elliottii* stands of different densities. Forest Ecology and Management. 154(1-2): 155-163.

Will, R.E.; Munger, G.T.; Zhang, Y.; Borders, B.E. 2002. Effects of annual fertilization and complete competition control on current annual increment, foliar development, and growth efficiency of different aged *Pinus taeda* stands. Canadian Journal of Forest Research. 32(10): 1728-1740.

Will, R.E.; Narahari, N.V.; Shiver, B.D.; Teskey, R.O. 2005. Effects of planting density on canopy dynamics and stem growth for intensively managed loblolly pine stands. Forest Ecology and Management. 205(1-3): 29-41.

STRUCTURE AND REGROWTH OF LONGLEAF PINE FORESTS FOLLOWING UNEVEN-AGED SILVICULTURE AND HURRICANE DISTURBANCE AT THE ESCAMBIA EXPERIMENTAL FOREST

Kimberly Bohn, Christel Chancy, and Dale Brockway[1]

In recent decades, considerable attention has been placed on restoring and managing longleaf pine (*Pinus palustris* Mill.) ecosystems across the southeastern United States. Although, historically, these forests have been successfully regenerated following even-aged shelterwood reproduction methods, uneven-aged silviculture has received increasing attention because it is thought to meet a diversity objectives including timber production, biodiversity enhancement, and habitat conservation and is also thought to emulate some of the natural disturbance regimes that have historically sustained these ecosystems. A long-term project, entitled the "Comparative Analysis of Reproduction Techniques (CART) for Sustainable Management of Longleaf Pine Ecosystems" at the Escambia Experimental Forest in Brewton, AL, was designed as an operational-scale effort to compare a variety of uneven-aged silvicultural methods (single-tree and group-tree selection cutting to 11.5 m^2/ha with small and large gap openings respectively) and even-aged methods (shelterwood cutting to 6 m^2/ha). However in September 2004, 2 months after harvesting was complete, the study sites were directly impacted by Hurricane Ivan resulting in significant damage to the recently cut shelterwood treatments on six plots. Forest damage was less severe, though noticeable, on uncut control plots and plots treated with selection system methods. While the original intent of the study design was to compare regeneration and growth following specific silvicultural treatments to specific residual stocking and spacing, the addition of hurricane disturbance provides an opportunity to evaluate the interactive effects of natural disturbance and the forest reproduction methods and to determine how best to adapt management in "the hurricane zone". The specific objectives of this research were to compare regeneration, density, and basal area and to quantify the spatial structure of residual trees 6 years following harvesting and hurricane impact.

Data were collected in 2003 prior to treatment, in 2005 after silvicultural treatments and hurricane impact, and again in 2010. In 2010, we revisited the remaining intact 9-ha plots comprised of three uncut control plots, three single-tree selection treatment plots and three group selection treatment plots. Within each 9-ha treatment plot, regeneration densities were measured on five randomly established 20- by 50-m subplots. Regeneration was classified as grass stage (< 0.3-m tall), bolt stage [0.3- to 1.8-m tall and < 2 cm in diameter at breast height (d.b.h.)], or sapling (at least 2 cm and < 10 cm d.b.h.). Diameters of all trees at least 2 cm and above were also recorded. Regeneration densities were analyzed by treatment (single-tree, group selection, control) and time (2003, 2005, and 2010). Total basal area of residual overstory trees ≥ 10 cm d.b.h. was analyzed by treatment in 2005 and 2010. Additionally, in each of the nine treatment areas, two 50- by 100-m plots were established in which all trees bolt-sized and larger were stem-mapped. Spatial pattern on each of the stem maps was evaluated using the Ripley's K point pattern process, which evaluated whether a pattern is clumped (also termed aggregated), random, or uniform (also termed dispersed). We used SPPA2.0 software to analyze distances ranging from 2 to 24 m, by 2-m intervals.

By 2010, significant regeneration had developed across the forest due to one or more relatively heavy seed-crop years between 2005 and 2010.

[1]Associate Professor and Research Assistant, respectively, University of Florida, School of Forest Resources and Conservation, Milton, FL 32584; and Research Ecologist, USDA Forest Service, Southern Research Station, Auburn, AL 36849.

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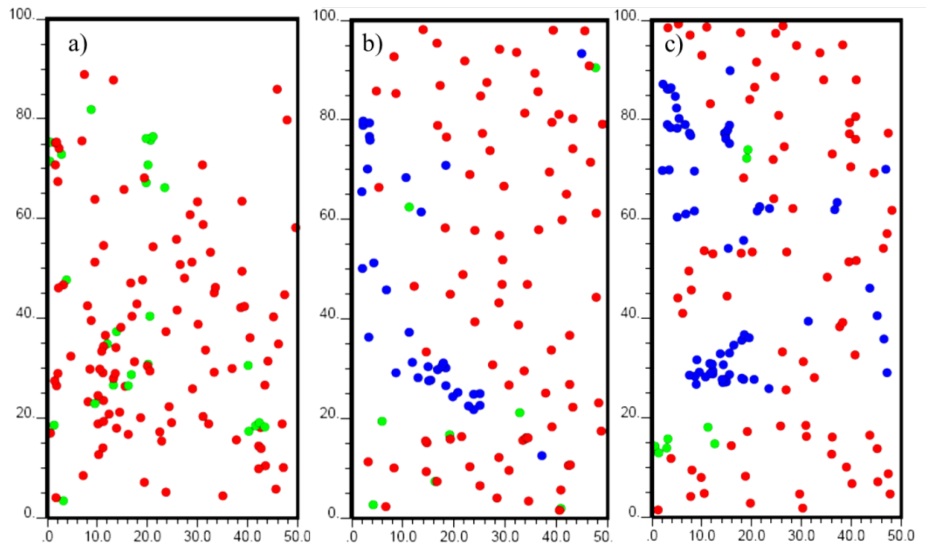

Figure 1--Example stem maps illustrating the spatial distribution of longleaf pine trees in plots treated with: (a) single-tree selection, (b) group selection, and (c) an uncut control and following Hurricane Ivan. Differently colored dots represent residual overstory trees > 10 cm (red), sapling regeneration (green) and bolt-sized regeneration (blue) 6 years following silvicultural treatments and hurricane damage.

The group selection plots contained significantly more grass-stage regeneration in 2010 (17,870 seedlings/ha) followed by single-tree selection (11,190 seedlings/ha) and the uncut control plots (8,070 seedlings/ha). Conversely, by 2010 there were significantly fewer bolt-stage regeneration in the group selection plots (53 stems/ha) than in the single-tree (605 stems/ha) or control plots (278 stem/ha). The number of longleaf saplings was not significant by treatment and ranged from 64 to 103 stems/ha. By 2010, the mean total basal area of residual overstory trees in the control plots was 15.6 m^2/ha, 11.0 m^2/ha in group selection, and 8.8 m^2/ha and in single-tree selection. Between 2005 and 2010, total basal area increased by about 1 m^2/ha in the control and group selection plots and by 0.6 m^2/ha in the single-tree plots, though relative growth rates after compensating for differences in post-hurricane basal areas were not significantly different.

Point pattern analyses using the Ripley's K function also indicate variability in the spatial distribution of residual overstory trees within and between treatments. Point pattern analysis in four of the six plots in single-tree selection treatment areas indicated clumped patterns at all or most spatial scales (fig. 1a), and two plots exhibited random patterns. Clumped patterns at small spatial scales indicates that there were several groups of closely spaced trees, while clumping at large spatial scales is typically indicative of large gap openings with no trees in between the smaller tree groups. Upon visual examination of the stem maps, it is evident that the plots exhibiting the highest degree of clumping also had very large gap openings, which likely was a result of hurricane damage. Conversely, two of the group selection plots exhibited significantly uniform patterns at smaller spatial scales of about 2 to 8 m (fig. 1b), though five of the six plots did show clumped spatial patterns at some larger scale. Overstory trees in most of the control plots exhibited random patterns at all spatial scales (fig. 1c).

Though more regeneration had reached bolt sizes or larger in single-tree selection plots than group selection, basal area growth of larger residual trees appear to be recovering at the same rate across plots. Further, the range of gaps sizes and spatial distribution of trees in either the single-tree or group selection plots did not vary much following the hurricane impacts. Although application of uneven-aged selection system methods requires special attention to both residual basal area and size of regeneration openings, in hurricane-prone areas attention to gap size may be may be less consequential than attention to residual basal area.

THICKNESS AND ROUGHNESS MEASUREMENTS FOR AIR-DRIED LONGLEAF PINE BARK

Thomas L. Eberhardt[1]

Abstract--Bark thicknesses for longleaf pine (*Pinus palustris* Mill.) were investigated using disks collected from trees harvested on a 70-year-old plantation. Maximum inner bark thickness was relatively constant along the tree bole whereas maximum outer bark thickness showed a definite decrease from the base of the tree to the top. The minimum whole bark thickness followed the same trend as the inner bark thickness while maximum whole bark thickness followed the same trend as the outer bark thickness. Greater bark thicknesses were observed along the northern face of the tree bole. Minimum/maximum whole bark thickness ratios, used as a measure of bark roughness, were fairly constant from the base of the tree up to a relative height near 60 percent; increasing values further up the bole reflected decreasing bark roughness. Comparisons of the data from the northern and southern faces suggested asymmetries in bark roughness around the circumference. Altogether, results demonstrate intriguing aspects of longleaf pine bark variability along the tree bole and in the four cardinal directions.

INTRODUCTION

The barrier properties imparted by tree bark to seal in moisture and protect against external damaging agents (e.g., fire, insects) are primarily served by the essentially dead outer bark (rhytidome). Particularly for the pines, the inner bark (phloem) contributes to the chemical and physical barriers formed via exudation of protective resins. Together, the outer bark and inner bark provide a complex and multifunctional system essential for secondary growth. Adding to the complexity of the outer bark is the presence of fissures that form as the girth of the tree bole expands. Since a fissure presents a zone of thinner outer bark, it would appear that minimum whole bark thickness may be more relevant than maximum whole bark thickness when speculating on the functionality of bark as a barrier. Although these fissures provide concealment for arthropods (Hanula and Franzreb 1998, Hooper 1996), the extent that they are a conduit to insect attack has yet to be clearly established. To date, whole bark thicknesses for longleaf pine (*Pinus palustris* Mill.) have been suggested to impact insect populations (Hanula and others 2000) and resistance to mortality from fire (Hare 1965, Martin 1963, Wang and Wangen 2011).

Measurements of bark thickness on the standing tree are also of interest from the perspective of utilization. Bark yields for the southern pines are roughly 10 percent along the bole (Cole and others 1966) and up to 60 percent for small branches (Phillips and others 1976). Given ongoing longleaf pine restoration efforts, a renaissance in longleaf pine harvesting (Landers and others 1995) will afford bark residues for utilization. Taking into consideration the functionality of bark on the living tree and its potential to be an actively-managed forest biomass resource, longleaf pine inner bark and outer bark maximum thicknesses were reported using fresh tree disks collected at five different sampling heights (Eberhardt 2013); in the present study, measurements were taken from these disks after drying under ambient conditions. In addition, minimum and maximum whole bark thicknesses in the air-dried state were measured on the complete set of tree disks taken every 0.61 m along each tree bole. These data are discussed in the context of the functionality of longleaf pine bark on the living tree.

MATERIALS AND METHODS

Fifteen 70-year-old longleaf pine trees were harvested in the summer (July 6 through July 25) from the J.K. Johnson Tract of the Palustris Experimental Forest located in the Kisatchie National Forest, LA. The felled trees were measured and then sectioned to afford 2-inch-thick disks every 0.61 m from the stump cut (0.15 m above ground level). Disks taken at approximately 0.15, 5, 10, 15, and 20 m, that had been measured for inner and outer bark thicknesses in the fresh state (Eberhardt 2013), were measured again after drying for 2 months under ambient conditions. One measurement for each thickness was taken with a digital caliper from each of the four quadrants designated on each disk: southwest, southeast, northwest, and northeast. Remaining disks (i.e., those not taken at the five above-specified

[1]Research Scientist, USDA Forest Service, Southern Research Station, Pineville, LA 71360.

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heights) were allowed to dry before taking any bark thickness measurements. Maximum and minimum whole bark thickness measurements were taken from all air-dry disks. Microsoft Excel 2010 was used to conduct two-sample paired *t*-tests to test for differences between measurements from neighboring quadrants (e.g., northeast and northwest) and averages of values for each semicircle representing opposing cardinal directions (i.e., north vs. south, east vs. west). Since none of the comparisons between the eastern and western faces were significant, probabilities are only shown for comparisons of the northern and southern faces. It is acknowledged that handling of the tree disks may have unavoidably resulted in some losses of very loose outer bark layers.

RESULTS AND DISCUSSION

Trees used in this study had a wide range of growth rates with values for diameter at breast height ranging from 14.7 cm to 45.5 cm (Eberhardt 2013). Total heights ranged from 17.6 m for the most suppressed tree to approximately 29 m for dominant trees. Using the values for total tree height and the height at which each disk was taken, values for relative height were calculated. Maximum inner and outer bark thicknesses were previously reported for the fresh tree disks (Eberhardt 2013); figure 1 shows the corresponding plot using the data from the disks after air-drying. Since the trends are unchanged, data plots for the air-dried disks could be extrapolated to estimate values for fresh disks after accounting for bark shrinkage; air-dried disks are easier to process in that they are not constrained by timely measurements as are fresh disks. The caveat would be accounting for any variability in shrinkage around the circumference of the tree. Calculating values for shrinkage using thickness values for the fresh and air-dried disks suggested greater shrinkage for the northern face compared to the southern face (Eberhardt 2013). After drying, the differences in the inner and outer bark thicknesses between the northern and southern faces were more subtle (table 1) when compared by two-sample paired *t*-tests (inner bark, $P = 0.0139$, outer bark $P = 0.0184$); prior to drying, the differences were highly significant (inner bark, $P = 0.0009$, outer bark $P = 0.0002$). Thus, while variability in moisture content has a significant impact on the observed variability in bark thicknesses around the circumference of a tree, other factors (e.g., environment, microclimate) may promote greater bark thicknesses along the northern face. The insulation capacity of bark has been shown to be a function of its thickness and moisture content (Bauer and others 2010). Accordingly, the direction of a fire could be a minor factor for tree survival in instances where there are definite differences in bark moisture content and thickness around the circumference of the tree that would afford different levels of insulation.

Similar to the inner and outer bark thicknesses, the minimum and maximum whole bark thicknesses gave the best fit with logarithmic models when plotted against relative height (fig. 2); however, a better fit was obtained with the minimum bark thickness ($R^2 = 0.7491$, fig. 2) compared to the maximum inner bark thickness ($R^2 = 0.3806$, fig. 1). The plot for the minimum and maximum whole bark thicknesses relative to inside bark radius (fig. 3) was also similar to the previously presented plot for inner and outer bark thicknesses (Eberhardt 2013). Observations showed the minimum whole bark thickness and the maximum inner bark thickness were both relatively constant along the bole of the tree; as for the maximum whole bark thickness and the maximum outer bark thickness, both decreased logarithmically from the base of the tree to the top.

Comparison of the maximum and minimum whole bark thicknesses for neighboring quadrants gave higher values for the northern quadrants relative to the corresponding southern quadrants; comparing the northern face to the southern face showed the maximum whole bark thickness to be greater ($P = 0.0049$) on the northern face (table 2). There was no difference ($P = 0.8850$) between the northern and southern faces for the minimum whole bark thickness. At this juncture, it should be noted that the mean maximum whole bark thickness (6.81 mm) is essentially the same as the corresponding maximum outer bark thickness (6.73 mm). This discrepancy likely resulted from the use of data from five sampling heights in

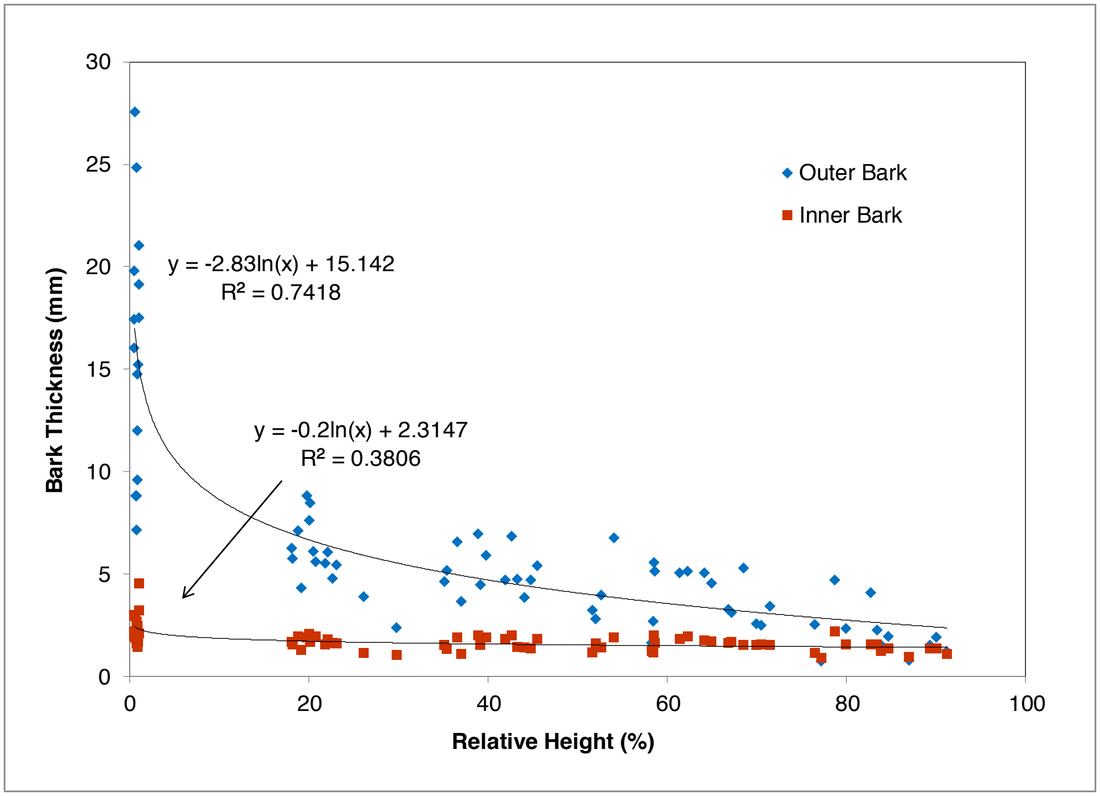

Figure 1--Maximum inner and outer bark thicknesses at relative heights up tree bole.

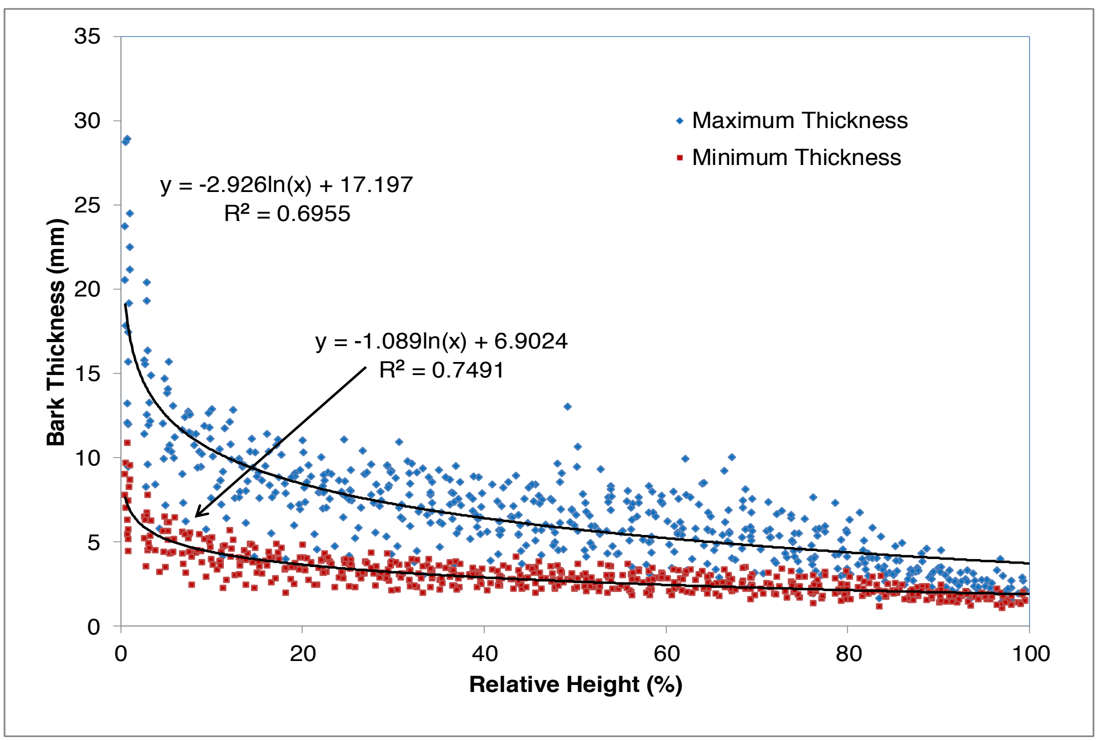

Figure 2--Maximum and minimum bark thicknesses at relative heights up tree bole.

Table 1--Two-sample paired mean analyses of maximum inner and outer bark thicknesses for northern and southern sampling orientations: comparisons for sample measurements taken before and after drying

Analysis	Means compared	Thickness measurement	Timing	Mean	*P* value
				mm	
1	North South	Inner bark	Before drying	2.183 2.013	0.0009
2	North South	Outer bark	Before drying	7.760 6.954	0.0002
3	North South	Inner bark	After drying	1.760 1.683	0.0139
4	North South	Outer bark	After drying	6.966 6.499	0.0184

Table 2--Two-sample paired mean analyses of whole bark thicknesses for northern and southern sampling orientations: comparisons for maximum and minimum bark thickness values

Analysis	Means compared	Thickness measurement	Mean	*P* value
			mm	
1	North South	Maximum	6.888 6.726	0.0049
2	North South	Minimum	3.037 3.035	0.8850

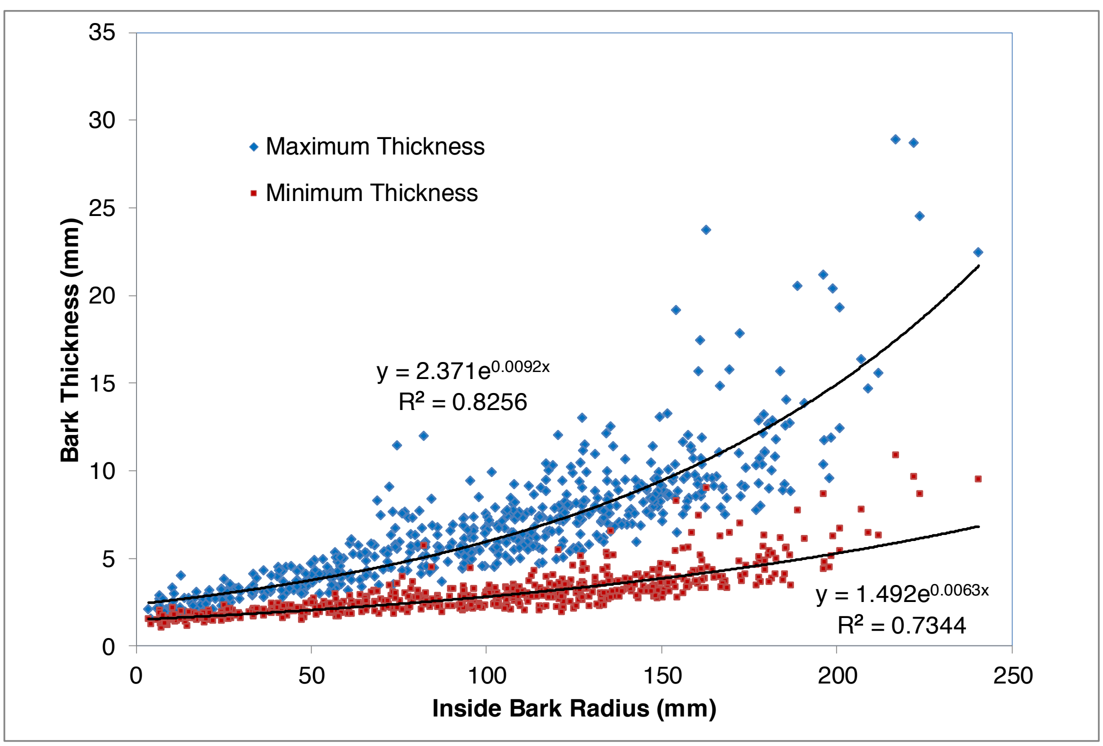

Figure 3--Maximum and minimum bark thicknesses relative to inside bark radius.

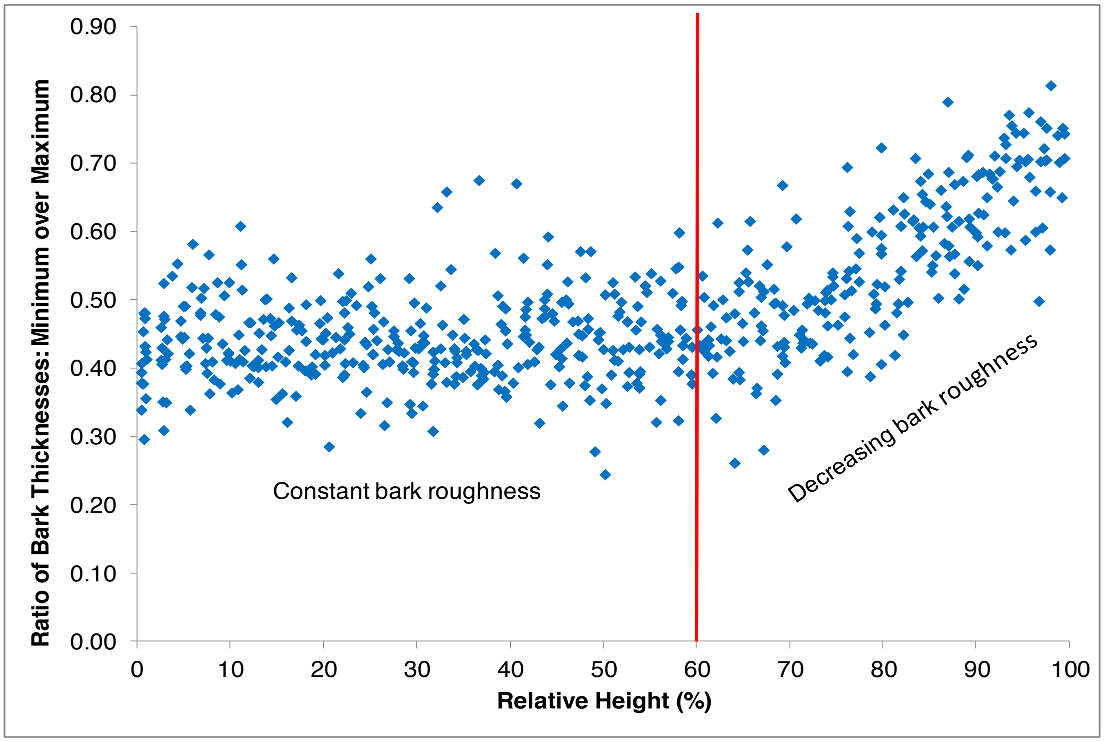

Figure 4--Ratio of bark thicknesses (minimum over maximum) at relative heights up tree bole.

Table 3--Two-sample paired mean analyses of whole bark minimum/maximum thickness ratios for northern and southern sampling orientations: comparisons at different relative sampling height ranges

Analysis	Means compared	Relative sampling height	Mean	*P* value
		%	*mm*	
1	North	0-60	0.4342	0.0077
	South		0.4443	
2	North	60-100	0.5547	0.5943
	South		0.5580	
3	North	0-100	0.4801	0.0261
	South		0.4875	

one case (outer bark thickness) and all sampling heights in the other (maximum whole bark thickness).

Measurement of the minimum and maximum whole bark thicknesses, and calculating a ratio of the two values, provided a crude assessment of the roughness of the bark; thus, dividing the minimum whole bark thickness by the maximum whole bark thickness gives the proportion of the whole bark that provides a continuous barrier around the circumference of the tree. Focusing on the microhabitat provided by bark fissures, more elaborate measures (e.g., bark-fissure index, fissure depth class) describe the fissure depth (MacFarlane and Luo 2009, Michel and others 2011). In the present study, plotting these minimum/maximum whole bark thickness ratios against the corresponding relative heights showed a fairly constant value (approximately 0.45) from the base of the tree up to a relative height near 60 percent (fig. 4). Above that relative height, the difference between the two thicknesses declined as it approached a ratio value of 0.80 at the top; an increasing ratio corresponds with decreasing bark roughness (i.e., decreasing fissure depth relative to the whole bark thickness). While differences in insect populations up the tree bole have been observed (Gargiullo and Berisford 1981, Hanula and Franzreb 1998), any relationships to differences in bark roughness have received limited attention. As with the other measurements of bark thickness, there is a statistically significant difference (table 3) between the northern and southern faces with a lower minimum/maximum bark thickness ratio (i.e., greater roughness) on the northern face along the main bole (relative height = 0 to 60 percent). It would be particularly intriguing if differences around the circumference of a tree could impact bark suitability for insect inhabitation.

CONCLUSIONS

While variability in moisture content impacts the observed variability in bark thicknesses around the circumference of a tree, other factors may promote greater bark thicknesses along the northern face. In longleaf pine, minimum whole bark thickness parallels the maximum inner bark thickness while maximum whole bark thickness parallels the maximum outer bark thickness. Measurement of the minimum/maximum whole bark thickness ratio appears to provide a simple measure of longleaf pine bark roughness asymmetries.

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LITERATURE CITED

Bauer, G.; Speck, T.; Blömer, J. [and others]. 2010. Insulation capability of the bark of trees with different fire adaptation. Journal of Material Science. 45: 5950-5959.

Cole, D.E.; Zobel, B.J.; Roberds, J.H. 1966. Slash, loblolly, and longleaf pine in a mixed natural stand; a comparison of their wood properties, pulp yields, and paper properties. Tappi Journal. 49(4): 161-166.

Eberhardt, T.L. 2013. Longleaf pine inner and outer bark thicknesses: measurement and relevance. Southern Journal of Applied Forestry. 37(3): 177-180.

Gargiullo, P.M.; Berisford, C.W. 1981. Effects of host density and bark thickness on the densities of parasites of the southern pine beetle. Environmental Entomology. 10(3): 392-399.

Hanula, J.L.; Franzreb, K.E. 1998. Source, distribution and abundance of macroarthropods on the bark of longleaf pine: potential prey of the red-cockaded woodpecker. Forest Ecology and Management. 102: 89-102.

Hanula, J.L.; Franzreb, K.E.; Pepper, W.D. 2000. Longleaf pine characteristics associated with arthropods available for red-cockaded woodpeckers. Journal of Wildlife Management. 64(1): 60-70.

Hare, R.C. 1965. Contribution of bark to fire resistance in southern trees. Journal of Forestry. 63: 248-252.

Hooper, R.G. 1996. Arthropod biomass in winter and the age of longleaf pine. Forest Ecology and Management. 82: 115-131.

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(11): 39-44.

MacFarlane, D.W.; Luo, A. 2009. Quantifying tree and forest bark structure with a bark-fissure index. Canadian Journal of Forest Research. 39: 1859-1870.

Martin, R.E. 1963. Thermal properties of bark. Forest Products Journal. 13: 419-426.

Michel, A.K.; Winter, S.; Linde, A. 2011. The effect of tree dimension on the diversity of bark microhabitat structures and bark use in Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*). Canadian Journal of Forest Research. 41: 300-308.

Phillips, D.R.; Clark, A.; Taras, M.A. 1976. Wood and bark properties of southern pine branches. Wood Science. 8(3): 164-169.

Wang, G.G.; Wangen, S.R. 2011. Does frequent burning affect longleaf pine (*Pinus paulstris*) bark thickness? Canadian Journal of Forest Research. 41: 1562-1565.

DETERMINATION OF LOBLOLLY PINE RESPONSE TO CULTURAL TREATMENTS BASED ON SOIL CLASS, BASE PRODUCTIVITY, AND COMPETITION LEVEL

David Garrett, Michael Kane, Daniel Markewitz, and Dehai Zhao[1]

The objective of this research is to better understand what factors drive loblolly pine (*Pinus taeda* L.) growth response to intensive culture in the University of Georgia Plantation Management Research Cooperative's Culture x Density study in the Piedmont and Upper Coastal Plain. Twenty study sites were established ranging from southern Alabama to South Carolina in 1998 or 1999. Treatments included six planting densities [300; 600; 900; 1,200; 1,500; and 1,800 trees per acre (TPA)], in a factorial combination with two cultural treatments (intensive and operational). The intensive culture contained complete competing vegetation control, fertilization at time of planting and additional fertilization before the 3rd, 4th, 6th, 8th, 10th, and 12th growing seasons. The operational treatment included first-year banded weed control, fertilization at planting, and fertilization before the 8th and 12th growing seasons. Age 12 growth response was calculated as the difference between tree and stand values for intensive versus operational culture on the plots planted at 600 TPA.

Sites and soils at each installation were classified into four groups: (1) Piedmont with mixed clay subsoil and > 3 inches topsoil (two installations); (2) Piedmont with kaolinitic subsoil and > 3 inches topsoil (six installations); (3) Upper Coastal Plain with < 20 inches to the argillic horizon (nine installations); and (4) Upper Coastal Plain with > 40 inches to argillic horizon (three installations). Base site productivity at each installation was defined as the expressed site index of the operational culture plots planted at 600 TPA.

Competing vegetation measurements were taken on the operational plots only since the intensive culture plots had complete weed control. The competition measurements, taken at ages 2, 4, 6, 8, and 10, include percent cover and height of the following: andropogon grasses, other grasses, and broadleaf plants, as well as an herbaceous measurement which was the sum of all three. Measurements were also taken on small and large woody stems in the form of height, sum height per acre, area per acre, volume per acre, and number of stems per acre. Small woody material is defined as material < 1.6 inches diameter at breast height (d.b.h.) or < 4.5-feet tall, and large woody is defined as > 1.6 inches d.b.h. or taller than 4.5 feet.

Age 12 loblolly pine mean d.b.h., height, basal area per acre, and volume per acre for intensive culture, operational culture and the response to intensive culture are presented in table 1 along with standard deviations and ranges. The operational culture plots averaged 6.9 inches in d.b.h., 46 feet in height, 146 square feet per acre in basal area, and 3,256 cubic feet per acre in volume. The growth response averaged 0.6 inch in d.b.h., 3 feet in height, 24 square feet per acre in basal area, and 762 cubic feet per acre in volume. Growth response was highly variable ranging from -644 to 1,665 cubic feet per acre. Two study sites exhibited no growth response to the intensive treatment; the cause of this is unknown but possibly due to microsite conditions and only having single replicates.

We did not observe a strong relationship between age 12 volume per acre response and soil class, base site productivity, or competing vegetation level on operational plots Although significant differences were not detected among the site-soil classes (fig. 1), the Upper Coastal Plain sites with > 40 inches to the argillic have the largest observed pine growth response. Age 12 response was poorly related to base site

[1]Masters Student, Professor, Professor, and Research Professional, respectively, University of Georgia, Warnell School of Forestry and Natural Resources, Athens, GA 30605.

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Table 1--Summary statistics of age 12 tree and stand attributes for intensive culture, operational culture, and volume per acre responses for 20 study locations in the Piedmont and Upper Coastal Plain

Culture	Attributes	Mean	Std. dev.	Range
Intensive	Volume (*feet*^{3}*/acre*)	4,018	715	2,286 - 5,169
	BA (*feet*^{2}*/acre*)	170	23	111 - 215
	Height (*feet*)	50	4	41 - 57
	D.b.h (*inches*)	7.5	0.56	5.9 - 8.3
Operational	Volume (*feet*^{3}*/acre*)	3,256	688	1,878 - 4,482
	BA (*feet*^{2}*/acre*)	146	24	98 -187
	Height (*feet*)	46	4	39 - 53
	D.b.h. (*inches*)	6.9	0.74	5.6 - 8.9
Response	Volume (*feet*^{3}*/acre*)	762	608	-644 -1,665
	BA (*feet*^{2}*/acre*)	24	17	-15 - 54
	Height (*feet*)	3	3	-5 -10
	D.b.h. (*inches*)	0.6	0.69	-1.4 - 1.5

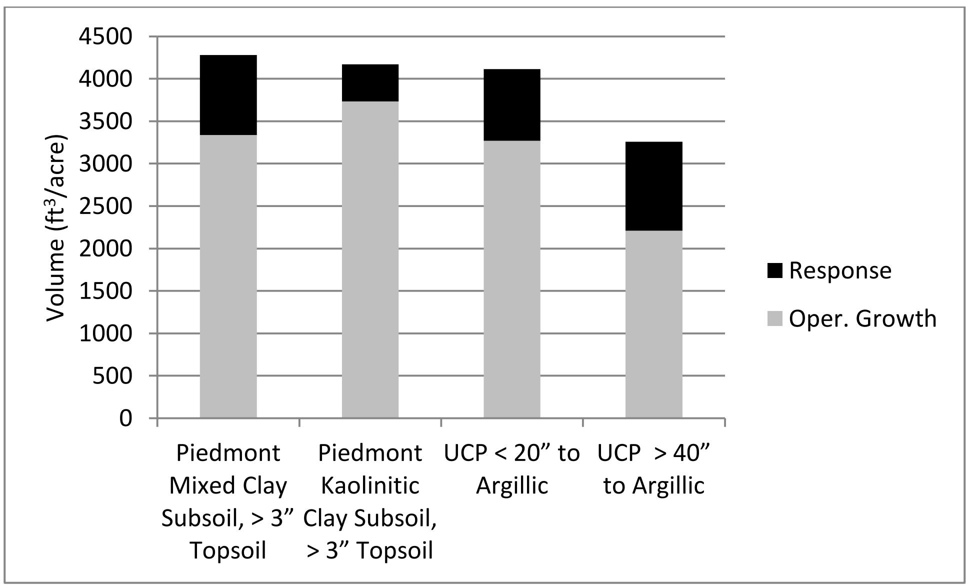

Figure 1--Age 12 mean loblolly pine standing volume per acre for operational culture and response to intensive culture by soil groups.

productivity on the operational plot (R^2 = 0.03, p = 0.40) although there did appear to be a trend of smaller responses as the site index increased. The best relationships between response and competition values were found using age 4 competition data. Age 12 volume response was significantly correlated with age 4 percent grass cover (R^2 = 0.21, p = 0 .031) and mean grass height (R^2 = 0.30, p = 0.011). The growth response tends to increase as competition levels increase on the operational plots.

Controlling competing vegetation has been proven to produce significant growth gains in a loblolly pine plantation system (Miller and others 2003). In addition, the application of fertilizer to loblolly pine has also increased the growth rates of southern plantations (Fox and others 2007). The combination of competition control and fertilization of loblolly pine produces the best results to increase pine growth response (Borders and Bailey 2001).While there was a significant range of age 12 response to intensive culture, the factors examined (soil class, base site productivity, and competing vegetation) were not strong determinants of pine plantation response magnitude on these well drained Piedmont and Upper Coastal Plain sites. Using a different soil classification system could prove to be a better predictor for pine growth response. Also, incorporating a measure of soil moisture and rainfall records could provide additional insight to pine growth response. It should be noted that the response measured in this study is to a combination of fertilization and competition control, and the effects of only fertilization or only competition control cannot be calculated. Finally, even the operational treatment used in this study could be considered intensive compared to typical field operations, and it is possible that the growth responses exhibited in this experiment could have been achieved with fewer cultural inputs.

LITERATURE CITED

Borders, B.E.; Bailey, R.L. 2001. Loblolly pine: pushing the limits of growth. Southern Journal of Applied Forestry. 25(2): 69-74.

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.

Miller, J.H.; Zutter, B.R.; Zedaker, S.M. [and others]. 2003. 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.

COMPARISON OF PLANTED LOBLOLLY, LONGLEAF, AND SLASH PINE DEVELOPMENT THROUGH 10 GROWING SEASONS IN CENTRAL LOUISIANA—AN ARGUMENT FOR LONGLEAF PINE

James D. Haywood, Mary Anne S. Sayer, and Shi-Jean Susana Sung[1]

Abstract—Two studies were established in central Louisiana to compare development of planted loblolly (*Pinus taeda* L.), longleaf (*P. palustris* Mill.), and slash (*P. elliottii* Engelm.) pine. Study 1 was on a Beauregard silt loam, and Study 2 was on Ruston and McKamie fine sandy loams. After 10 growing seasons, stocking ranged from 1,165 longleaf to 1,606 loblolly pines per ha in Study 1. Slash (9.8 m) and loblolly (8.9 m) pine trees had similar average total heights, and both were taller than longleaf pine (5.3 m). Volume production was comparable between slash (134 m^3/ha) and loblolly (111 m^3/ha) pine, and longleaf pine (24 m^3/ha) had the least volume per ha. In Study 2, stocking ranged from 1,907 longleaf to 2,356 slash pines per ha. Slash (11.2 m) and loblolly (10.8 m) pine trees had similar average total heights, and both were taller than longleaf pine (9.2 m). Volume production was similar between slash (181 m^3/ha) and loblolly (162 m^3/ha) pine, and both produced more volume per ha than longleaf pine (96 m^3/ha). Although outcomes in growth and yield among species were similar in both studies, the magnitude of differences between longleaf versus loblolly and slash pine was greater in Study 1 than Study 2 for several reasons. While longleaf pine had the poorest growth and yield, its early development normally lags behind that of other southern pines, and longleaf pine grew sufficiently well to warrant consideration if other values are taken into account, which are herein discussed.

INTRODUCTION

Through much of the 20th century, land managers had serious problems regenerating longleaf pine (*Pinus palustris* Mill.); thus, many managers favored loblolly (*P. taeda* L.) and slash (*P. elliottii* Engelm.) pine over longleaf (Croker 1987). Despite past favoritism, longleaf pine might be as productive as loblolly or slash pine by age 20 to 25 years on some sites provided there is good survival and an absence of brown-spot needle blight (caused by *Mycosphaerella dearnessii* Barr.) that can arrest seedling growth (Derr 1957, Kais and others 1986, Schmidtling 1987, Shoulders 1985).

More specifically, longleaf pine has a grass-stage of seedling development, during which there is little above-ground stem growth, whereas loblolly and slash pine do not. The grass stage can continue for the first 3 to 6 years after planting (or longer under adverse conditions) as the root system develops (Harlow and Harrar 1969). On silt loam soils in central Louisiana, planted longleaf pine seedlings typically remain in the grass stage (seedlings being no more than 12 cm tall) for 1 to 6 years unless other factors continue to stunt growth (Haywood 2005, 2007; Haywood and others 2012).

Controlling competition by herbicidal and mechanical means for site preparation and after planting loblolly and slash pine are widely accepted practices in the southern United States (Moorhead and others 2012). Similarly, to establish longleaf pine seedlings, some type of vegetation management program is usually necessary because brush outgrows young longleaf pine seedlings and saplings (Haywood 2009a, Haywood and Grelen 2000, Haywood and others 2001).

Prescribed fire is commonly used in longleaf pine plantations to control vegetative competition even when the pines are seedling-sized—a practice that would be avoided in managing other southern pine species. However, fire is not essential for growing longleaf pine seedlings provided other vegetation management practices are used to control competition (Haywood 2005, 2007). In fact, intensive vegetation management employing herbicides was the best treatment for increasing height growth of planted longleaf pine in central Louisiana rather than prescribed fire (Haywood 2011), and intensive vegetation management has benefited both slash and loblolly pine development as well (Haywood and Tiarks 1990; Haywood and others 1994, 2003; Tiarks and Haywood 1981).

Presently, there is an interest in restoring longleaf pine across its native range in the southeastern United States that is partly focused on increasing its acreage from 1.4 million to 3.2 million ha (3.4 to 8 million acres) by 2024 (America's Longleaf 2009). The Longleaf Partnership Council estimated that in 2012 there were 1.7 million ha (4.2 million acres) of forest

[1]Supervisory Research Forester and Research Plant Physiologists, respectively, USDA Forest Service, Southern Research Station, Pineville, LA 71360.

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dominated by longleaf pine, and it has been proposed by state organizations that there will be 2.4 million ha (6 million acres) of longleaf pine by 2027 (Gaines 2012). To achieve either of these outcomes will require forests, pastures, and croplands to be reforested or converted to longleaf pine. The principal method will be to plant longleaf pine seedlings.

Given that one can successfully establish longleaf pine plantations, should one choose longleaf pine rather than loblolly or slash pine? To help answer this question, two studies in central Louisiana were established to compare the survival, growth, and yield of these three southern pines through 10 growing seasons. Results are directly applicable to management of planted pines in the West Gulf Coastal Plain and southeastern United States in general.

METHODS

Study Site Descriptions

Study 1 is located on the Kisatchie National Forest (KNF) in central Louisiana at 53 m above sea level on a gently sloping (1 to 3 percent) Beauregard silt loam soil (fine-silty, siliceous, superactive, thermic Plinthaquic Paleudult) (Kerr and others 1980). The natural pine and mixed hardwood forest cover was clearcut harvested in the mid-1980s, and the site was sheared and windrowed in 1991. Prescribed fire was applied in March 1993 and 1996 to the low cover of herbaceous and scattered woody vegetation that developed after windrowing. Vegetation, which was dominated by bluestem grasses (*Andropogon* spp. and *Schizachyrium* spp.), was rotary mowed in fall 1996 before plot establishment.

Study 2 is on two soil complexes on the KNF. The first one at 55 m above sea level is comprised of Ruston fine sandy loam (fine-loamy, siliceous, semiactive, thermic Typic Paleudult) with a slope of 1 to 10 percent (Kerr and others 1980). The other complex at 66 m above sea level is comprised of McKamie fine sandy loam (fine, mixed, superactive, thermic Vertic Hapludalfs) with a slope of 1 to 10 percent. Before harvesting, Study 2 was a closed canopy, mature, loblolly pine-hardwood forest. The understory vegetation was mostly hardwood trees, shrubs, vines, and scattered shade tolerant herbaceous plants.

The two study sites are within the humid, temperate, coastal plain and flatwoods province of the West Gulf Region of the southeastern United States (McNab and Avers 1994). The climate is subtropical. From 1997 through 2007, December had the lowest average mean temperature of 10.3 °C, and August had the highest average mean temperature of 28.3 °C (National Climatic Data Center 2012). Annual precipitation averaged 1456 mm with 1045 mm during the growing season, which included the months of March through November. Both studies are on uplands suitable for restoring longleaf pine forests (Turner and others 1999).

Treatment Establishment

In both studies, research plots were established in a randomized complete block design of four blocks with three treatments (three pine species) each (Steel and Torrie 1980). Study 1 was installed in fall 1996. Each of the 12 research plots measured 39 by 39 m and contained 16 rows of 16 seedlings arranged in 2.4- by 2.4-m spacing. The center 100 seedlings (10 rows of 10 seedlings each) formed the measurement plot of each research plot. The three southern pines studied, loblolly, longleaf, and slash pine, were randomly assigned to different plots within each block. Blocking was based on slope.

Study 2 was installed in fall 1997 as at Study 1 with the following exceptions: each of the 12 research plots measured 29 by 29 m and contained 16 rows of 16 seedlings arranged in 1.8- by 1.8-m spacing. Blocking was by soil complex (two blocks on each complex) and topographic location within each complex.

For both studies, seeds for all three species were supplied by the Stuart Seed Orchard, KNF, LA, and were open-pollinated native Louisiana parent trees. Seedlings were grown in containers by Forest Service personnel in Pineville, LA. Study 1 was planted in March 1997, and Study 2 was planted in March 1998. Both studies utilized dibbles with tips of the correct size and shape for the 3.8-cm wide and 14-cm deep root plugs.

In April 1997, sethoxydim (2-[1-(ethoxyimino) butyl]-5-[2-(ethylthio)propyl]-3-hydroxy-2-cyclohexen-1-one] was banded over the rows of planted pine seedlings to control bluestem grasses on all 12 plots at Study 1. In April 1998, sethoxydim was again applied, and hexazinone [3-cyclohexyl-6-(dimethylamino)-1-methyl-1,3,5-triazine-2,4(1H,3H)-dione] was banded over the rows of planted seedlings for general herbaceous plant control. Within the 0.9-m bands, the rate of sethoxydim was 0.37 kg active ingredient (ai)/ha, and for hexazinone, the rate was 1.12 kg ai/ha. In addition, triclopyr ([(3,5,6-trichloro-2-pyridinyl)oxy]acetic acid) at 0.0048 kg acid equivalent/liter was tank mixed with surfactant and water and applied as a directed foliar spray to the scattered hardwood trees and shrubs in April 1998 and 1999.

At Study 2, hexazinone was banded over the rows of planted pine seedlings in April 1998 on all 12 plots; sethoxydim was not needed for bluestem grass control. The triclopyr tank-mix was applied as a directed foliar spray to hardwood trees and shrubs in April 1998 and June 1999.

Brown-spot needle blight on the longleaf pine seedlings became a concern at Study 1. To control the disease, prescribed fire was applied to only the longleaf pine plots in May of the third growing season. Byram's fire intensity averaged 278 kJ/s/m of fire front, which was a low fire intensity for grass-dominated fuels in central Louisiana (Haywood 2009a, 2011). Prescribed fire was not needed for brown-spot needle blight control at Study 2.

Measurements and Data Analysis

Total tree heights were measured with a calibrated pole through eight growing seasons at Study 1 and seven growing seasons at Study 2. Thereafter, a laser instrument (Criterion 400 Survey Laser, Laser Technology, Inc., Centennial, CO) was used. The change in measurement equipment is evident in the shape of the total tree height curves for loblolly and slash pine in figures 1 and 2. Tree d.b.h. was measured with a diameter tape after 8 through 10 growing seasons at Study 1 and 7 through 10 growing seasons at Study 2. Total height and d.b.h. were used to calculate outside-bark stem volume of loblolly pine with Baldwin and Feduccia's (1987) formula, longleaf pine with Baldwin and Saucier's (1983) formulas, and slash pine with Lohrey's (1985) formula.

For each study, number of living pines per ha after 10 growing seasons; average total height, basal area, and volume per tree; and pine basal area and volume per ha were compared among loblolly, longleaf, and slash pine with a randomized complete block design model at $\alpha = 0.05$ using SAS Statistical Software (SAS Institute Inc. 1985). If there were significant species differences, mean comparisons were made with Duncan's Multiple Range Tests at $\alpha = 0.05$ (Steel and Torrie 1980).

RESULTS

At Study 1, longleaf pine seedlings began emerging from the grass stage in the third growing season (25 percent emergence), and 74 percent were in height growth by the end of the fourth growing season. This trend is shown in figure 1 by the relative flatness of the height curve for longleaf pine at ages 1 to 3 years compared to the other two pine species. Ninety-nine percent of the surviving longleaf pines were in height growth by age 8 years. The remaining 1 percent of the longleaf pines were perhaps planted too deep or were overtopped by competing vegetation.

Sixty-four percent of the longleaf pine seedlings at Study 1 were infected with brown-spot needle blight at age 2 years, and to control the disease, a prescribed fire was applied in May of the third growing season. At the end of the third year, only 4 percent of the longleaf pines were evidently infected with brown-spot. The percentage of brown-spot infection remained below 5 percent through seven growing seasons; disease incidence was not recorded thereafter.

Partly because of a high percentage of brown-spot needle blight and slow rate of height growth initiation at Study 1, average height, basal area, and volume per tree were significantly lower for longleaf pine than for loblolly and slash pine after 10 growing seasons (fig. 1). Volume per slash and loblolly pine averaged 85 and 69 dm^3/tree, respectively, and was 20 dm^3/tree for longleaf pine (probability [P] > F-value [F] < 0.0001). There were no statistically significant differences in average height, basal area, and volume per tree between loblolly and slash pine based on Duncan Multiple Range Test comparisons.

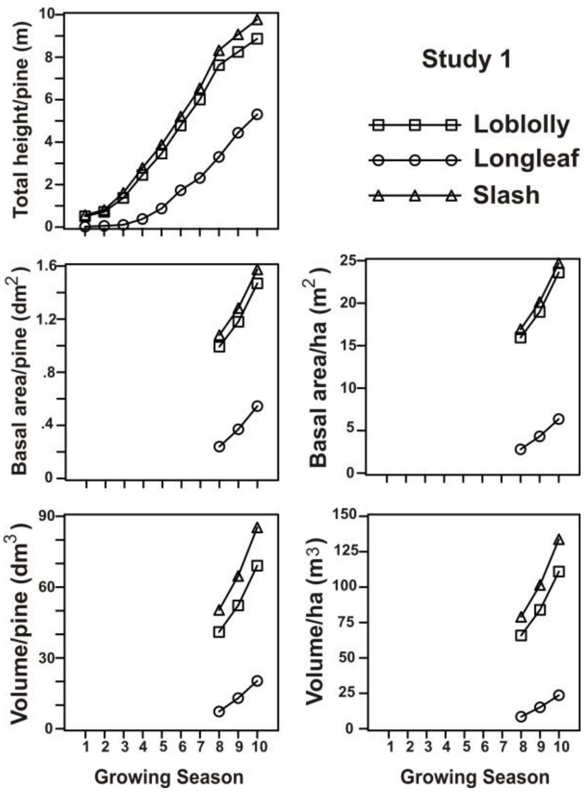

Figure 1—Comparing total height, basal area, and volume per tree and basal area and volume per ha among loblolly, longleaf, and slash pine planted on a Beauregard silt loam soil through 10 growing seasons.

Pine stocking was also significantly greater for loblolly (1,606 trees/ha) and slash (1,569 trees/ha) pine than for longleaf pine (1,165 trees/ha) ($P > F < 0.0001$) with no significant difference between loblolly and slash pine in Study 1. As a result of the differences in stocking and average tree size, basal area and volume per ha were also significantly greater for loblolly and slash pine than for longleaf pine (fig. 1). Volume per ha was 134 and 111 m^3/ha for slash and loblolly pine, respectively, and was 24 m^3/ha for longleaf pine ($P > F < 0.0001$). There were no significant differences in basal area and volume per ha between loblolly and slash pine.

A different longleaf pine growth pattern occurred in Study 2 (fig. 2). On these sandy loam soils, 36 percent of the longleaf pine seedlings emerged from the grass stage in the first growing season. Ninety-seven percent were in height growth after 2 years, and all surviving longleaf pine seedlings were in height growth after 3 years. The quick height initiation is evident in figure 2. In addition, brown-spot needle blight infected only 1 percent of the longleaf pine seedlings through age 4 years.

Despite the rapid height initiation and low disease incidence among longleaf pine in Study 2, average height, basal area, and volume per tree were still significantly greater for loblolly and slash pine than for longleaf pine after 10 growing seasons (fig. 2). Volume per slash and loblolly pine averaged 78 and 73 dm^3/tree, respectively, and was 51 dm^3/tree for longleaf pine ($P > F = 0.0074$). There were no significant differences in average height, basal area, or volume per tree between loblolly and slash pine.

Pine stocking was not significantly different among the three pine species after 10 growing seasons ($P > F = 0.0564$) in Study 2. Stocking was 2,356; 2,259; and 1,907 trees/ha for slash, loblolly, and longleaf pine, respectively. However, as a result of the differences in average tree size and the small differences in stocking, basal area and volume per ha were also significantly greater for loblolly and slash pine than for longleaf pine (fig. 2). Volume per ha of slash and loblolly pine were 181 and 162 m^3/ha, respectively, and was 96 m^3/ha for longleaf pine ($P > F = 0.0019$). There were no significant differences in basal area and volume per ha between loblolly and slash pine.

DISCUSSION

Longleaf pine growth and production were less than for loblolly and slash pine on both study sites through 10 growing seasons. This is not surprising as the development of young longleaf pines normally lags behind that of other southern pines. However, this does not mean that landowners should avoid longleaf pine and choose loblolly or slash pine if the alternative values that longleaf pine produce are desired. These alternatives, which do not necessarily involve management of federally listed threatened or endangered species, would include: (1) hurricane tolerance; (2) growth on droughty, low-nutrition sites; (3) contributes to habitat diversification for wildlife and game animals; (4) being the pine species of choice in arson-prone areas; (5) conveys sustainable forestry certification; (6) increases product quality and diversity; and (7) need not require special management.

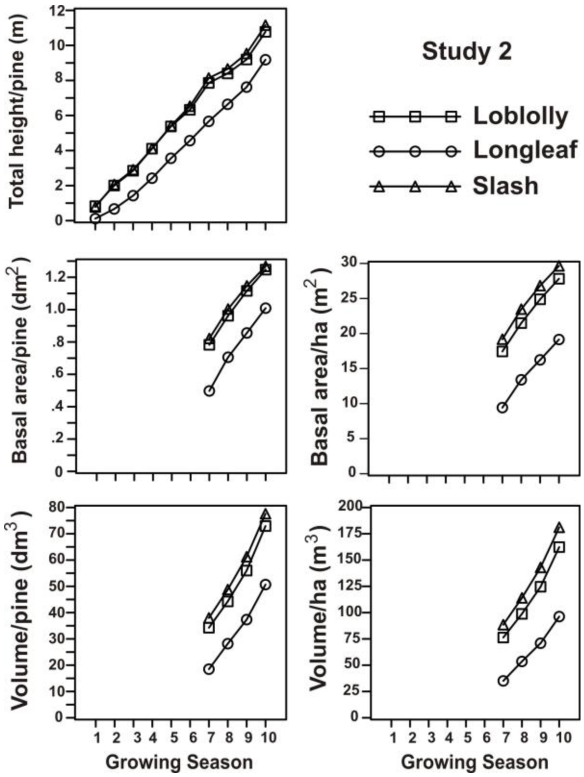

Figure 2—Comparing total height, basal area, and volume per tree and basal area and volume per ha among loblolly, longleaf, and slash pine planted on Ruston and McKamie sandy loam soils through 10 growing seasons.

The tolerance of longleaf pine to hurricane-force winds was reported by Johnsen and others (2009) wherein longleaf pine stands damaged by Hurricane Katrina suffered less mortality than loblolly pine stands and less loss in overstory basal area than loblolly or slash pine stands. The poor growth of intensively managed, short-rotation loblolly and slash pine on sites similar to Study 1 is believed to result from phosphorus deficiencies that greatly reduce stand growth and yield in subsequent rotations (Haywood and Tiarks 2002). Longleaf pine has lower phosphorus and nitrogen requirements than loblolly and slash pine and lower calcium and magnesium requirements than loblolly pine (Dickens and Moorhead 2009). Thus, sites such as Study 1 might be good candidates for longleaf pine reforestation if 25- to 30-year multiple rotations are planned, and management options do not include nutrient amendment. In addition, longleaf pine is better adapted to soil water deficit compared to the other southern pines (Barrett 1995), and once longleaf pine seedlings are established, they tolerate severe to extreme drought conditions (Haywood 2005, 2007).

Longleaf pine is also highly resistant to pine beetles and fusiform rust (*Cronartium quercuum* f. sp. *fusiforme*) (North Carolina Forest Service 2012, The Longleaf Alliance 2012).

Kerr and others (1980) reported that the McKamie soil series in Study 2 has a site index at base age 50 years of 25.9 m (85 feet) for both loblolly and longleaf pine. Therefore, Study 2 is a site where longleaf pine might be recommended for reforestation based on its predicted growth rather than being tolerant of soil resource limitations. In addition, total height growth patterns through age 10 years for loblolly and longleaf pine were similar to those reported by Schmidtling (1987) on an unfertilized, well-drained, upland fine sandy loam in southern Mississippi. In his study, longleaf pine was taller than loblolly pine by age 25 years. This suggests that longleaf pine may be overlooked by forest managers as a reforestation species of choice for sites where it could grow better than expected (Shoulders 1985). For example, longleaf pine was reported to have a site index of 21.3 m (70 feet) at base age 50 years on Smithdale sandy loams (fine-loamy, siliceous, subactive, thermic Typic Hapludults) in central Louisiana (Kerr and others 1980). However, in recent work, the site index of longleaf pine on a Smithdale soil was measured to be 26.2 m (86 feet) at age 50 years (Haywood 2009b), which is similar to the expected site index for loblolly and slash pine on this soil series (Kerr and others 1980).

Because longleaf pine stands can be prescribed burned even as seedlings (Haywood 2005, 2007), the maintenance of an open understory of herbaceous plants and low brush with fire can provide a forest habitat for wildlife different from nearby, unburned stands. This diversity in forest cover should increase game management options and improve the value of property to be leased for hunting. Open forest structure can also be aesthetically pleasing and the rich understory cover typical of longleaf pine forests provides biological diversity that is sought by some landowners. Furthermore, longleaf pine is a forest type that has historic and cultural value for many (North Carolina Forest Service 2012, The Longleaf Alliance 2012, Way 2012). As long as the longleaf pine overstory is not allowed to become too dense, these desirable attributes can be maintained with fire (Haywood 2012, Wolters 1981). In addition, the tolerance of longleaf pine to fire is why it is the pine species of choice for planting in arson-prone areas.

Because a longleaf pine forest can be biologically diverse, its restoration on a portion of a landowner's property may help them obtain sustainable forestry certification. In today's markets, products derived from longleaf pine can be more valuable than products from other southern pines (The Longleaf Alliance 2012). For example, although the market for pulpwood, lumber, and other solid wood products has declined in recent years, the market for utility poles has not fluctuated significantly (The Longleaf Alliance 2012). Longleaf pine stands produce a high percentage of poles, and since poles are a more valuable product than pulpwood and sawtimber, longleaf pine may afford a stronger economic position than loblolly or slash pine. True, the pole market is small relative to the sawtimber market, but the volume of longleaf pine being brought to market is also small. Thus, longleaf pine provides investment security and reduces risk to landowners because the volatility of the longleaf pine market is low.

Besides wood products, longleaf pine stands produce other preferred market goods such as pine straw for landscaping and weaving of high quality baskets, native herbs, high-end furnishings and furniture, and forage for livestock (Haywood 2012, North Carolina Forest Service 2012, The Longleaf Alliance 2012). Some landowners are adverse to the use of prescribed fire, but prescribed burning is not necessary in longleaf pine stands provided the eventual loss of the herbaceous plant community is not a concern (Haywood 2009a, 2011). Thus, no special management practices are needed to grow longleaf pine. This could be convenient for landowners who only wish to reforest with longleaf pine due to poor soil quality, arson problems, to reduce market risk, or for alternative products because longleaf pine can be established and managed similarly to a landowner's loblolly or slash pine.

In summary, "beauty is in the eye of the beholder". Longleaf pine is not a likely choice for landowners only interested in maximizing wood fiber production. However, landowners who desire values other than wood fiber production may want to add longleaf pine to their suite of crop trees given that it grew well on both study sites, although not as well as loblolly or slash pine.

LITERATURE CITED

America's Longleaf. 2009. Range-wide conservation plan for longleaf pine. http://www.americaslongleaf.org/resources/conservation-plan. [Date accessed: October 30, 2012].

Baldwin, V.C., Jr.; Feduccia, D.P. 1987. Loblolly pine growth and yield prediction for managed West Gulf plantations. Res. Pap. SO-236. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 27 p.

Baldwin, V.C., Jr.; Saucier, J.R 1983. Aboveground weight and volume of unthinned, planted longleaf pine on West Gulf forest sites. Res. Pap. SO-191. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 25 p.

Barrett, J.W. 1995. Regional silviculture of the United States. 3rd ed. New York: John Wiley & Sons, Inc. 643 p.

Croker, T.C., Jr. 1987. Longleaf pine. A history of man and a forest. Forestry Rep. R8-FR-7. Atlanta, GA: U.S. Department of Agriculture Forest Service, Region 8. 37 p.

Derr, H.J. 1957. Effects of site treatment, fertilization, and brownspot control on planted longleaf pine. Journal of Forestry. 55: 364-367.

Dickens, E.D.; Moorhead, D.J. 2009. Sample loblolly, longleaf, and slash pine foliage for nutrient analysis. Bethesda, MD: Society of American Foresters. The Forestry Source. 14(9): 14.

Gaines, G. 2012. Report of the Longleaf Partnership Council state assessment and projections for longleaf pine August 2012. Atlanta, GA: America's Longleaf, Longleaf Partnership Council, Assessment and Reporting Team. 4 p.

Harlow, W.M.; Harrar, E.S. 1969. Textbook of dendrology. New York: McGraw-Hill Book Co: 81-84.

Haywood, J.D. 2005. Effects of herbaceous and woody plant control on *Pinus palustris* growth and foliar nutrients through six growing seasons. Forest Ecology and Management. 214: 384-397.

Haywood, J.D. 2007. Influence of herbicides and felling, fertilization, and prescribed fire on longleaf pine establishment and growth through six growing seasons. New Forests. 33: 257-279.

Haywood, J.D. 2009a. Eight years of seasonal burning and herbicidal brush control influence sapling longleaf pine growth, understory vegetation, and the outcome of an ensuing wildfire. Forest Ecology and Management. 258: 295-305.

Haywood, J.D. 2009b. Influence of pine straw harvesting, prescribed fire, and fertilization on a Louisiana longleaf pine site. Southern Journal of Applied Forestry. 33(3): 115-120.

Haywood, J.D. 2011. Influence of herbicides and felling, fertilization, and prescribed fire on longleaf pine growth and understory vegetation through ten growing seasons and the outcome of an ensuing wildfire. New Forests. 41: 55–73.

Haywood, J.D. 2012. Frequency and season of prescribed fire affect understory plant communities in longleaf pine stands. 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: 137-143.

Haywood, J.D.; Goelz, J.C.; Sword Sayer, M.A.; Tiarks, A.E. 2003. Influence of fertilization, weed control, and pine litter on loblolly pine growth and productivity and understory plant development through 12 growing seasons. Canadian Journal of Forest Research. 33: 1974-1982.

Haywood, J.D.; Grelen, H.E. 2000. Twenty years of prescribed burning influence the development of direct-seeded longleaf pine on a wet pine site in Louisiana. Southern Journal of Applied Forestry. 24(2): 86-92.

Haywood, J.D.; Harris, F.L.; Grelen, H.E.; Pearson, H.A. 2001. Vegetative response to 37 years of seasonal burning on a Louisiana longleaf pine site. Southern Journal of Applied Forestry. 25(3): 122-130.

Haywood, J.D.; Sung, S-J. S.; Sword Sayer, M.A. 2012. Copper root pruning and container cavity size influence longleaf pine growth through five growing seasons. Southern Journal of Applied Forestry. 36(3): 146-151.

Haywood, J.D.; 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.

Haywood, J.D.; Tiarks, A.E. 2002. Response of second-rotation southern pines to fertilizer and planting on old beds—fifteenth-year results. 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: 497-502.

Haywood, J.D.; Tiarks, A.E.; Snow, G.A. 1994. Combinations of fungicide and cultural practices influence the incidence and impact of fusiform rust in slash pine plantations. Southern Journal of Applied Forestry. 18(2): 53-59.

Johnsen, K.H.; Butnor, J.R.; Kush, J.S. [and others]. 2009. Hurricane Katrina winds damaged longleaf pine less than loblolly pine. Southern Journal of Applied Forestry. 33(4): 178-181.

Kais, A.G.; Cordell, C.E.; Affeltranger. C.E. 1986. Benomyl root treatment controls brown-spot disease on longleaf pine in the southern United States. Forest Science. 32(2): 506-511.

Kerr, A., Jr.; Griffis, B.J.; Powell, J.W. [and others]. 1980. Soil survey of Rapides Parish, Louisiana. Baton Rouge, LA: U.S. Department of Agriculture Soil Conservation Service and Forest Service. 87 p. and 114 maps. In cooperation with: the Louisiana Agriculture Experiment Station

Lohrey, R.E. 1985. Stem volume, volume ratio, and taper equations for slash pine in the West Gulf region. In: Shoulders, E., ed. Proceedings of the 3rd biennial southern silvicultural research conference. Gen. Tech. Rep. SO-54. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 451-459.

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 Forest Service. 267 p.

Moorhead, D.J.; Dangerfield, C.W.; Beckwith, J.R. 2012. Opportunities for intensive pine plantation management. http://www.bugwood.org/intensive/98-002.html. [Date accessed: October 30, 2012].

National Climatic Data Center. 2012. U.S. Department of Commerce, NOAA satellite and information service. http:// http://www1.ncdc.noaa.gov/pub/data/cirs/. [Date accessed: October 30, 2012].

North Carolina Forest Service. 2012. Longleaf pine forest – superior economic, environmental, and historical values. Longleaf Leaflet No. 2. Raleigh, NC: North Carolina Forest Service. http://ncforestservice.gov/publications/LongleafLeaflets/LL02.pdf. [Date accessed: October 30, 2012].

SAS Institute, Inc. 1985. SAS user's guide: statistic. 5th ed. Cary, NC: SAS Institute Inc. 956 p.

Schmidtling, R.C. 1987. Relative performance of longleaf compared to loblolly and slash pines under different levels of intensive culture. In: Philips, D.R., comp. Proceedings of the 4th biennial southern silvicultural research conference. Gen. Tech. Rep. SE-42. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 395-400.

Shoulders, E. 1985. The case for planting longleaf pine. In: Shoulders, E., ed. Proceedings of the 3rd biennial southern silvicultural research conference. Gen. Tech. Rep. SO-54. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 255-260.

Steel, R.G.D.; Torrie, J.H. 1980. Principles and procedures of statistics a biometrical approach. 2d ed. New York: McGraw-Hill Book Company. 633 p.

The Longleaf Alliance. 2012. Restoring and managing longleaf pine. http://www.longleafalliance.org/. [Date accessed: October 30, 2012].

Tiarks, A.E.; Haywood, J.D. 1981. Response of newly established slash pine to cultivation and fertilization. Res. Note SO-272. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 4 p.

Turner, R.L.; Van Kley, J.E.; Smith, L.S.; Evans, R.E. 1999. Ecological classification system for the national forests and adjacent areas of the West Gulf Coastal Plain. Nacogdoches, TX: The Nature Conservancy and Stephen F. Austin State University. 305 p.

Way, A. 2012. Longleaf pine ecosystem. The new Georgia encyclopedia, land and resources/environment/conservation and management/environmental history section. http://www.georgiaencyclopedia.org/nge/Home.jsp. [Date accessed: October 30, 2012].

Wolters, G.L. 1981. Timber thinning and prescribed burning as methods to increase herbage on grazed and protected longleaf pine ranges. Journal of Range Management. 34(6): 494-497.

CROWN EXPANSION FOLLOWING THINNING IN NATURALLY REGENERATED AND PLANTED LONGLEAF PINE

Steven B. Jack, Noah A. Jansen, and Robert J. Mitchell[1]

The recent focus on restoration of longleaf pine (*Pinus palustris* Mill.) forests has frequently led to planting longleaf pine on old-field and cutover sites. While many perceptions regarding response of longleaf pine to management are based upon measurements in naturally regenerated stands, it is generally observed that crown development in planted longleaf stands is dissimilar to that observed in natural stands; that is, planted longleaf pine trees tend to have more branches and wider, more "full" crowns at young ages in comparison to naturally regenerated trees. Many planted longleaf stands are reaching the size and age for thinning and with these observed differences in crown characteristics, it is important to explore further whether the response of tree crowns in plantations differs from that of naturally regenerated trees. Few (if any) published studies other than Minor (1951), however, have examined crown dimensions of individual longleaf pine trees as influenced by stand characteristics.

We examined post-thinning crown expansion of planted and natural longleaf pine by comparing trees in thinned and unthinned stands for each establishment type. This study was located at Ichauway, an 11 750-ha preserve located near Newton in southwest Georgia. Planted longleaf were measured in a 25-year-old plantation that was thinned at 17 years (with a portion not thinned). Trees from natural stands were selected from intermediate and small codominant crown classes in plots thinned using individual tree selection and unthinned control plots that are part of a long-term research project. These natural stands are multi-aged, but average age of the canopy dominants is 80 to 90 years. In the plantation, six fixed-area plots were randomly located in both thinned and unthinned areas. In the natural stands, previously mapped trees were randomly selected for sampling, with trees selected from a similar diameter range as those in planted stands. Two crown diameters were measured at right angles for all target trees using a densitometer to identify crown "edges." Diameter at breast height (d. b.h.) was then recorded, and local stocking and competition were characterized using overstory abundance index (OAI, Battaglia and others 2002) in a 7-m radius around each tree. Mean values for stand characteristics were compared for all treatment and stand combinations using ANOVA (PROC MIXED; SAS Institute 2002-2010). Linear regression was used to examine for differences in crown response for different tree sizes and stand conditions by comparing regression parameters between treatment/stand-type combinations. Statistical comparisons of the regression parameters were carried out in SAS using PROC GENMOD (SAS Institute 2002-2010).

Results for average stand characteristics are shown in table 1. Thinned stands of either type had larger average size (d.b.h or crown diameter) and lower OAI than the unthinned stands. Generally, there were no significant differences by stand type, and the interaction between stand type and treatment was only significant for d.b.h. due to the large diameter differences between treatments in the natural stands. These results are not surprising given the well-understood responses of tree size to thinning, and lower OAI is to be expected following removal of trees in the thinning operation. There was, however, a wider range of values around the mean characteristics in the unthinned stands in comparison to the thinned stands (data not shown).

Regression analyses showed mixed results for crown diameter as a function of d.b.h. and OAI (fig. 1), with no significant differences in slope and only occasional differences in intercept parameters between treatment and stand type

[1]Conservation Ecologist, Lead Research Technician, and Scientist (deceased), respectively, Joseph W. Jones Ecological Research Center, Newton, GA 39870-8522.

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Table 1--Mean values, and standard errors (SE), for overstory abundance index (OAI), diameter at breast height (d.b.h.), and crown diameter (CD) by stand type and treatment. Statistical significance was tested at the 0.05 level

	Mean values (SE)				Pr > F		
	Natural thinned	Natural unthinned	Plantation thinned	Plantation unthinned	Treatment	Stand type	Trtmt * Std type
OAI	506 (51.0)	865 (67.6)	749 (43.6)	1,248 (48.6)	<0.0001	<0.0001	0.21
D.b.h. (*cm*)	29.2 (0.92)	20.7 (1.18)	26.3 (0.90)	23.7 (0.54)	<0.0001	0.95	0.0009
CD (*m*)	5.03 (0.19)	3.37 (0.24)	4.55 (0.25)	3.19 (0.13)	<0.0001	0.092	0.46

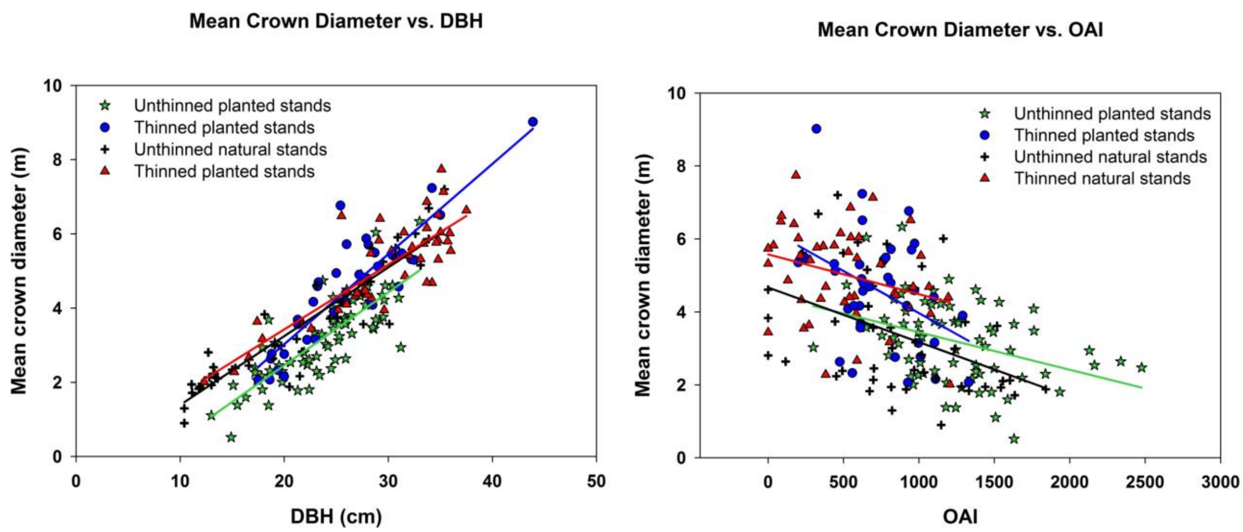

Figure 1--Average crown diameter as a function of diameter at breast height (d.b.h., left) and overstory abundance index (OAI, right). Points represent measured data by treatment and stand type, and lines represent linear regression results by treatment and stand type. In general, slopes were not significantly different across stand type and treatment for either relationship, but some intercept parameters were significantly different between treatment/stand combinations.

combinations. Given the observed differences in crown morphology between planted and natural stands, our hypothesis was that slopes for the different treatment/stand-type combinations would be statistically significant, indicating a varying response to thinning by stand type.

The lack of statistically significant differences in slope was somewhat unexpected given the plotted regression lines shown in figure 1 but is likely due to the variability in the data and, to a lesser degree, the limited range of tree sizes sampled (especially in the plantation stands). The significant differences in intercept terms could indicate a difference in crown width for a given tree size or level of competition, but statistical significance was indicated for only a few comparisons (data not shown) and did not show any definite trends with treatment or stand type. One interesting factor to think about is the influence of height on individual crown characteristics as shown by Murphy and Shelton (1995) for uneven-aged loblolly pine. We did not examine any height influence, but it is perhaps an important factor to consider given the large differences in age and height for the plantation and natural stands used in this study.

Although our working hypothesis was that individual tree crowns in the plantation and natural stands would respond differently to canopy manipulation through thinning (as indicated by differences in regression parameters), the lack of significant results is actually useful, especially in terms of modeling responses for the two stand types: i.e., separate models are likely not needed for plantation and natural stands. The results of this study are preliminary, however, with several confounding factors such as age differences, time since thinning, type of thinning operation and variable spacing. Additional study of these relationships will be required during the development of predictive models.

LITERATURE CITED

Battaglia, M.A.; Mou, P.; Palik, B.J.; Mitchell, R.J. 2002. The effect of spatially variable overstory on the understory light environment of an open-canopied longleaf pine forest. Canadian Journal of Forest Research. 32: 1984-1991.

Minor, C.O. 1951. Stem-crown diameter relations in southern pine. Journal of Forestry. 49: 490-493.

Murphy, P.A.; Shelton, M.G. 1995. Factors affecting crown dimensions in uneven-aged loblolly pine stands. In: Edwards, 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: 176-181.

SAS Institute Inc. 2002-2010. SAS software for Windows. Release 9.3. Cary, NC: SAS Institute, Inc. [Not paged].

ASSESSING CROWN DYNAMICS AND INTER-TREE COMPETITION IN SOUTHERN PINES

Timothy A. Martin, Angelica Garcia, Tania Quesada, Eric J. Jokela, and Salvador Gezan[1]

Genetic improvement of southern pines has been underway for 50 years and during this time, deployment of germplasm has generally evolved from more genetically diverse to less genetically diverse. Information is needed on how deployment of individual genotypes in pure blocks will affect traits such as within-stand variation in individual tree traits, as well as tree-level competitive interactions. Most genetic information for tree breeding programs is derived from single-tree or row-plot progeny tests. In contrast, modern forestry deployment strategies compel us to understand how elite genotypes will grow in single-family or even clonal blocks, where competition is either between closely related individuals or identical genotypes, respectively. This research focuses on understanding how crown characteristics of clones influence inter-tree competition.

The Forest Biology Research Cooperative at the University of Florida has established a clonal block plot trial series titled: Varietal ARchitecture Investigations Examining Tree Interactions on Experimental Sites (VARIETIES). VARIETIES examines tree- and stand-level dynamics of select loblolly pine clones growing in pure- and mixed-genotype plots. Age 2- and 3-year data demonstrate similar growth performance among elite clones in the trials but variation among clones in crown size characteristics such as crown width, crown volume, and crown width/crown height ratio.

As a result, some clones show differences in indices of growth efficiency, such as stem volume/crown volume (table 1). We quantified biomass distribution in three contrasting clones in VARIETIES (ARB-1, ARB-2, and ARB-4) and found patterns suggesting some of the mechanisms underlying efficiency differences. For example, narrow-crowned clone ARB-1 allocated less biomass to branches than the other two clones but relatively more biomass to stem. In contrast, clone ARB-2 allocated relatively less biomass to foliage than the other two clones, suggesting differences in photosynthetic efficiency.

Table 1-- Age 3 year tree-level characteristics: stem volume (SV), crown volume (CV), crown width (CW), relative crown width (CW/H) and stem volume growth efficiency (SV/CV) for four clones in the VARIETIES I experiment near Starke, FL[a]

Clone	SV	CV	CW	CW/H	SV/CV
	dm^3	*m^3*	*m*		*dm^3/ m^3*
ARB-4	14.81b	5.53b	1.91a	0.559c	2.41c
ARB-3	12.85b	4.66b	1.79b	0.517b	2.63ac
ARB-1	14.18b	4.75b	1.77b	0.496a	2.79a
ARB-2	22.87a	6.91a	2.05a	0.524b	3.21b

[a]Within a column and a biomass component, parameter estimates followed by the same lower case letter were not significantly different at α = 0.05 level.

[1]Professor, Graduate Student, Post-doctoral Fellow, Professor, and Assistant Professor, respectively, University of Florida, School of Forest Resources and Conservation, Gainesville, FL 32611.

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EARLY RELEASE IMPROVES LONG-TERM GROWTH AND DEVELOPMENT OF DIRECT-SEEDED NUTTALL OAK SAPLINGS

James S. Meadows, Robert L. Johnson, and Roger M. Krinard[1]

Abstract-- Early growth of bottomland oaks is typically slow, and many oaks eventually become overtopped by trees of other species. Removal of these larger competitors in a young stand might improve growth of the oaks and lead to more free-to-grow oaks as the stand matures. Release treatments were applied in 1980 to an 11-year-old, direct-seeded Nuttall oak (*Quercus texana* Buckl.) plantation on a Sharkey clay soil in west-central Mississippi. Treatments consisted of either cutting or deadening all non-oaks more than 15-feet tall, more than 20-feet tall, and more than 25-feet tall, plus an unreleased control. The average height of free-to-grow Nuttall oaks at the time of treatment was 15 feet. Response data were collected annually for the first 9 years after release, at 11 years after release, and at 30 years after release. Release treatments that removed competitors more than 15-feet tall and those that removed competitors more than 20-feet tall significantly increased stand basal area and quadratic mean diameter of the plantation but did not significantly increase either pulpwood or sawtimber volume per acre, even though differences among treatment means appeared to be large. Release treatments that removed competitors more than 20-feet tall were as effective but less costly than those treatments that removed competitors more than 15-feet tall.

INTRODUCTION

Early growth of bottomland oaks is typically slow. Consequently, many oaks eventually become overtopped by faster-growing trees of other species. Precommercial removal of these larger competitors in a young stand might improve growth of the oaks and encourage the development of more free-to-grow (trees with full sunlight directly overhead) oaks as the stand matures. However, there is a lack of knowledge about how young oaks grow and develop in competition with trees of other species in even-aged stands. If the oaks are unable to overcome early suppression by faster-growing trees of other species, some form of precommercial release cutting will be necessary to assure satisfactory oak development into the overstory of the stand.

Crop-tree release, in which pre-selected crop trees of desirable species are released fully from overhead competition and partially from side competition, has been used successfully to improve growth and development of overtopped saplings of several hardwood species. For example, Downs (1942) reported that crop-tree release stimulated diameter growth and reduced crown class regression in sugar maple (*Acer saccharum* Marsh.), white oak (*Quercus alba* L.), and yellow-poplar (*Liriodendron tulipifera* L.) saplings in the southern Appalachian Mountains. Conover and Ralston (1959) found that a sequence of two crop-tree release cuttings, spaced 8 years apart, increased long-term diameter growth of American elm (*Ulmus americana* L.), white ash (*Fraxinus americana* L.), and American basswood (*Tilia americana* L.) saplings in an 11-year-old northern hardwood stand in Wisconsin.

Della-Bianca (1975) also described increased diameter growth of crop trees after an intensive crop-tree release cutting, in which all woody stems other than crop trees were cut in an 11-year-old mixed-hardwood stand in the southern Appalachian Mountains. Crop trees included various species of red oaks and white oaks, black locust (*Robinia pseudoacacia* L.), and red maple (*Acer rubrum* L.). However, effects of the release cutting on diameter growth of crop trees dissipated by the end of the sixth year after release. Della-Bianca (1975) observed that high-quality stands developed in both the released plots and unreleased control plots and concluded that release cuttings were not necessary to promote adequate oak development on medium-quality sites in the southern Appalachians.

However, efforts to release oak saplings from overtopping competition have not always been successful. For example, Lamson and Smith (1978) found that crop-tree release did not improve diameter growth nor prevent crown class regression in 9-year-old sugar maple, northern red oak (*Q. rubra* L.), black cherry

[1]Principal Silviculturist, Project Leader (retired), and Mensurationist (deceased), USDA Forest Service, Southern Research Station, Stoneville, MS 38776.

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(*Prunus serotina* Ehrh.), and yellow-poplar saplings in West Virginia. They concluded that crop-tree release is inadvisable on good sites in the southern Appalachians.

So, can direct-seeded Nuttall oak (*Q. texana* Buckl.) saplings overcome early suppression by faster-growing trees of other species and develop into the overstory of the stand without the assistance of silvicultural treatment? If not, will some form of precommercial release cutting be effective in promoting satisfactory oak development into the overstory? Insight into the first question may be provided by Oliver (1978), who described a pattern of natural stand development in undisturbed, even-aged, mixed-hardwood stands in New England. Initially, northern red oaks were overtopped by faster-growing red maple and black birch (*Betula lenta* L.). But the northern red oaks gradually overcame this early suppression and were able to out compete the other species as the stand developed. By age 60, northern red oaks dominated the stand and formed a continuous canopy above the other species. Clatterbuck and Hodges (1988) described a similar pattern of stand development for cherrybark oak (*Q. pagoda* Raf.) grown in mixture with sweetgum (*Liquidambar styraciflua* L.) on bottomland sites in Mississippi.

To address these questions, this study was initiated in 1980 to evaluate the effects of seven release treatments designed to favor the development of 11-year-old, direct-seeded Nuttall oak saplings in competition with naturally regenerated non-oaks of the same age. The goal of the release treatments was to increase the number of free-to-grow oaks per acre. Johnson and Krinard (1988) reported 6-year results of the study.

METHODS

Study Site

The study was established on a bottomland site within the Delta Experimental Forest in west-central Mississippi. Soil within the study site is Sharkey clay (very-fine, smectitic, thermic Chromic Epiaquerts). It is poorly drained, slowly permeable, high in montmorillonitic clay, has a high shrink-swell capacity, and is dry in the summer and wet in the winter (Pettry and Switzer 1996). The site is flat and typically experiences annual backwater flooding for several weeks during the winter and spring. Floodwaters occasionally persist well into the summer. Broadfoot (1976) reported an average site index of 91 feet at 50 years for Nuttall oak.

Plantation Establishment

Before establishment of the Nuttall oak plantation, the study site supported a mature, mixed-species, bottomland hardwood stand dominated by American elm, green ash (*F. pennsylvanica* Marsh.), and sugarberry (*Celtis laevigata* Willd.), a typical species association on Sharkey clay flats. All merchantable timber was harvested, and the study site was cleared of all remaining vegetation and logging debris in the summer and fall of 1968. No other site preparation treatments were applied, but the study site was generally free of vegetation before plantation establishment.

The study site was sown with Nuttall oak acorns in the spring of 1969. Rows within the plantation are 10 feet apart; seed spots are spaced at 5-foot intervals within each row. Four Nuttall oak acorns were sown at each seed spot to ensure adequate oak stocking after anticipated high rates of acorn predation and expected low rates of seedling survival. Strips 5-feet wide were mowed down the center of the 10-foot-wide area between rows annually for the first 5 years after plantation establishment and again after the 10th year. This operation left an unmowed strip about 5-feet wide centered on each row. Naturally regenerated trees of various species became established in the unmowed strips and grew in direct competition with direct-seeded Nuttall oaks.

Treatments

Seven release treatments were applied in early 1980 when the plantation was 11 years old:

1. Cut all non-oaks more than 15 feet tall (C15+)
2. Deaden all non-oaks more than 15 feet tall (D15+)
3. Cut all non-oaks more than 20 feet tall (C20+)
4. Deaden all non-oaks more than 20 feet tall (D20+)
5. Cut all non-oaks more than 25 feet tall (C25+)
6. Deaden all non-oaks more than 25 feet tall (D25+)
7. Unreleased control - no cutting or deadening

Direct-seeded Nuttall oaks in the dominant or codominant crown classes averaged 15 feet in height at the time release treatments were applied. Consequently, treatments approximated the removal, through either cutting or deadening, of all non-oaks as tall as (C15+ and D15+), 1⅓ times as tall as (C20+ and D20+), and 1⅔ times as tall as (C25+ and D25+) direct-seeded Nuttall oaks in the dominant or codominant crown classes. Cutting of non-oaks was performed with a chainsaw between January and March 1980. Deadening of non-oaks was accomplished through injection with a mixture of 2,4-D and picloram in May 1980.

Study Design

Individual treatments were applied to three-row treatment plots in early 1980. However, only the center row of each treatment plot was measured. With 54 seed spots per row, the measurement plot consists of 0.062 acres.

Total height, diameter at breast height (d.b.h.), and crown class of every direct-seeded Nuttall oak greater than 4.5-feet tall were measured annually for the first 9 years and at the end of the 11th year after treatments were applied. Crown class, d.b.h., and merchantable height for either pulpwood or sawtimber were measured at the end of the 30th year after treatment. During each measurement event, data were collected for all variables on every direct-seeded oak at each seed spot within the measurement row. However, only data for the tallest tree at each seed spot were used in the statistical analyses.

Analysis of variance for a randomized complete block design with three replications of seven treatments was used to detect differences among treatments in trees per acre, basal area per acre, quadratic mean diameter, and the number of free-to-grow oaks per acre for each of the first 9 years and for the 11th and 30th years of the study. Free-to-grow oaks are defined as those direct-seeded oaks in the dominant or codominant crown classes. Pulpwood volume per acre and sawtimber volume per acre were analyzed for the 30th year only. Significance tests were conducted at the 0.05 level of probability. Treatment effects were considered fixed; block effects were considered random. Duncan's New Multiple Range Test was used to separate treatment means.

RESULTS AND DISCUSSION

Competitors

By the time the plantation was 11-years old, many direct-seeded Nuttall oaks had become overtopped by volunteer trees of other species. Before application of the release treatments in 1980, there were 283 free-to-grow oaks per acre, averaged across the plantation. In contrast, there were 937 competitors per acre that were taller than the average free-to-grow oak (nearly 15 feet). Green ash accounted for nearly half of all competitors more than 15-feet tall. Other common competitors included American elm, water hickory [*Carya aquatica* (Michx. f.) Nutt.], honeylocust (*Gleditsia triacanthos* L.), and sugarberry. The number of competitors removed during treatment decreased substantially with decreasing intensity of release (fig. 1). By implication, the cost required to apply the release treatments also decreased with decreasing intensity of release.

Response to Release Treatments

We found no significant differences among treatments in any of the response variables during any of the first 9 years after application of release treatments. Even though some means appeared to differ across treatments for some response variables during some years, high variability within the data resulted in large standard errors associated with most means and likely prevented detection of any statistical differences that may have existed among treatments.

The study site experienced a severe ice storm in February 1994 (14 years after application of release treatments). Many trees in the plantation suffered significant limb breakage in the crown. The stems of some trees snapped off at heights of 10 to 15 feet. Damage from the ice storm certainly affected growth and development of the direct-seeded Nuttall oak saplings. Consequently, beneficial effects of the release treatments that may have accrued to many trees in the plantation may have been negated by the ice storm.

Because we were unable to detect significant differences among treatments during the first 9 years of the study, the remainder of this paper describes conditions at the time of treatment and addresses responses only in the 11th and 30th years after release.

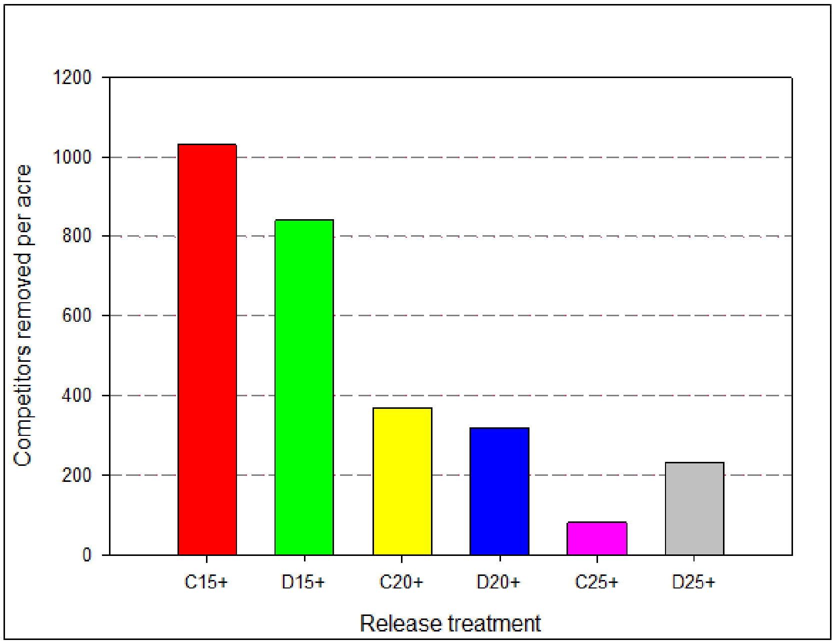

Figure 1--Number of competitors removed during application of release treatments.

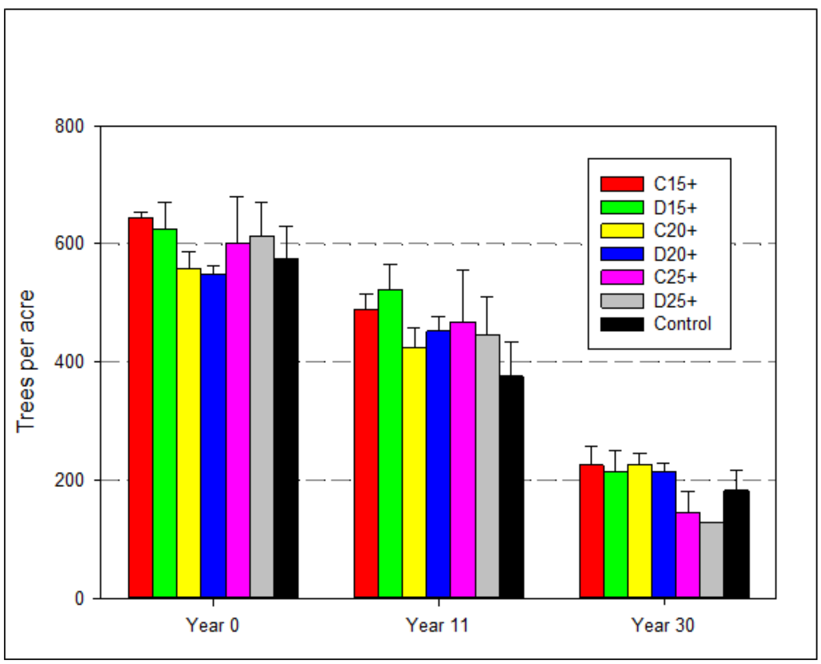

Figure 2--Number of Nuttall oak trees per acre (± SE), by treatment, immediately before release (year 0) and 11 and 30 years after application of seven release treatments. Means within each year followed by the same letter are not significantly different at the 0.05 level of probability.

Trees per acre--The plantation averaged 595 direct-seeded Nuttall oak trees per acre, across all plots, at the time the study was established. Plot means ranged from 548 to 645 trees per acre. We found no significant differences (p = 0.70) among plots prior to application of release treatments (fig. 2). Based on an initial density of 871 seed spots per acre, survival of direct-seeded oaks averaged 68 percent across all plots. Plot means ranged from 63 to 74 percent.

Through the first 11 years after release, the two most severe release treatments (C15+ and D15+) appear to have been successful at keeping oaks alive (fig. 2). Relative to the unreleased control plots, there were 30 percent more oaks in the C15+ plots and 39 percent more oaks in the D15+ plots. However, these differences among treatment means were not statistically significant (p = 0.61).

By the end of the 30th year after release, the four most severe release treatments (C15+, D15+, C20+, and D20+) seemed to have kept more oaks alive than the two least severe treatments (C25+ and D25+), with the number of oaks per acre in the unreleased control plots intermediate between these two groups (fig. 2). Number of oaks ranged from 215 to 226 per acre across the four most severe treatments and from 129 to 145 per acre across the two least severe treatments. However, these differences among treatment means were not statistically significant (p = 0.15).

Basal area per acre--Before application of release treatments, the plantation averaged 4.6 square feet of basal area per acre, in direct-seeded Nuttall oaks only. Plot means ranged from 4.2 to 5.0 square feet per acre, with no significant differences (p = 0.94) among plots at the time of treatment (fig. 3).

As the plantation developed after release, we observed a steady decline over time in the p value associated with statistical significance of differences among treatments in oak basal area per acre, from 0.94 at the time of treatment to 0.24 for 5-year treatment means to 0.11 for 9-year treatment means. This decline in p value indicates that differences among treatments have become larger over time and are approaching the threshold for statistical significance despite high variability in the data. The p value declined to 0.06 by the end of the 11th year after release. The four most severe release treatments clearly produced more oak basal area per acre than did the two least severe treatments or the unreleased control (fig. 3), but these differences were not statistically significant. In fact, the D15+ and D20+ treatment plots contained 49 and 43 square feet of oak basal area per acre, respectively, roughly double the 23 square feet of basal area per acre contained in the unreleased control plots.

Basal area per acre is the only response variable for which statistically significant differences (p = 0.02) were detected among 30-year treatment means (fig. 3). Separation of treatment means revealed a pattern similar to the one observed at the end of the 11th year after release. Specifically, the two most severe deadening treatments (D15+ and D20+) produced oak basal areas of 100 and 105 square feet per acre, respectively, which were significantly greater than the oak basal areas produced by the C25+ and D25+ treatments, 64 and 44 square feet per acre, respectively. In fact, the D20+ release treatment, with a basal area of 105 square feet per acre, was the only treatment that produced a significantly larger oak basal area than the 66 square feet per acre found in the unreleased control plots. However, even though the four most severe release treatments, with oak basal areas ranging from 84 to 105 square feet per acre, appear to have produced larger basal areas than the unreleased control, most comparisons between these treatments and the unreleased control were not statistically significant.

Quadratic mean diameter--Quadratic mean diameter of direct-seeded Nuttall oaks across the plantation at the time the study was established was 1.18 inches. Quadratic mean diameter of treatment plots ranged from 1.10 to 1.29 inches, with no significant differences (p = 0.76) among plots before application of release treatments (fig. 4).

Like the trend observed for basal area per acre, the p value associated with statistical significance of differences among treatments in oak quadratic mean diameter steadily declined as the plantation developed after release, from 0.76 at the time of treatment to 0.26 for 5-year treatment means to 0.10 for 9-year treatment

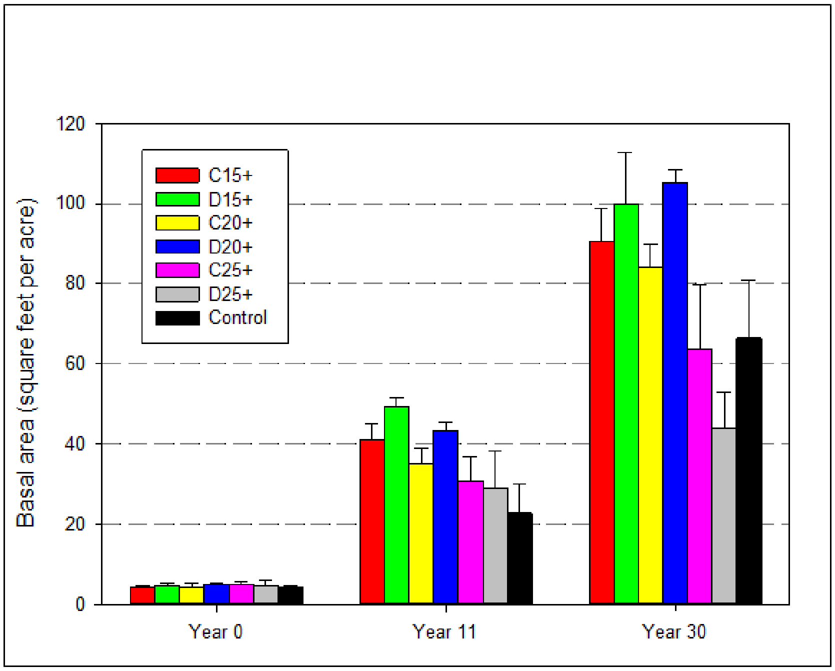

Figure 3--Basal area per acre (± SE) of Nuttall oak, by treatment, immediately before release (year 0) and 11 and 30 years after application of seven release treatments. Means within each year followed by the same letter are not significantly different at the 0.05 level of probability.

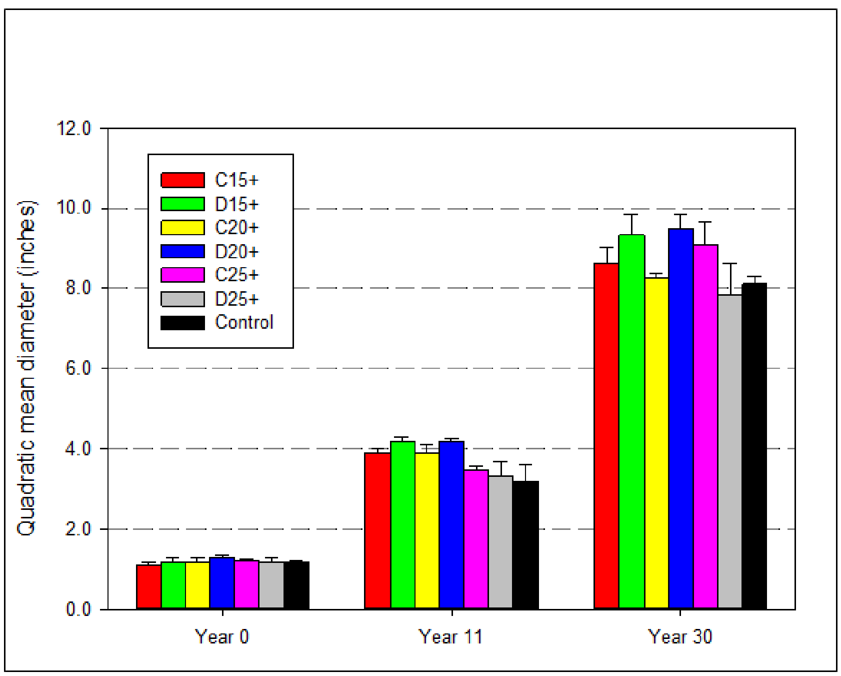

Figure 4--Quadratic mean diameter (± SE) of Nuttall oak, by treatment, immediately before release (year 0) and 11 and 30 years after application of seven release treatments. Means within each year followed by the same letter are not significantly different at the 0.05 level of probability.

means. Again, we believe that this decline in p value indicates that differences among treatments have become larger and are approaching the threshold for statistical significance despite high variability in the data.

The p value reached that threshold at the end of the 11th year after release. In fact, quadratic mean diameter is the only response variable for which statistically significant differences ($p = 0.05$) were found among 11-year treatment means (fig. 4). The two most severe deadening treatments (D15+ and D20+) produced stands with oak quadratic mean diameters of 4.18 and 4.20 inches, respectively, which were significantly larger than the oak quadratic mean diameters found in the D25+ and unreleased control plots, 3.32 and 3.19 inches, respectively. Oak quadratic mean diameters produced through the four most severe release treatments ranged from 3.89 to 4.20 inches and appeared to be larger than those produced through the C25+, D25+, and control treatments, but only some of the comparisons were statistically significant.

However, significant differences in oak quadratic mean diameter among treatments ($p = 0.21$) could not be detected at the end of the 30th year after treatment (fig. 4). All release treatments, except C20+ and D25+, produced stands with oak quadratic mean diameters seemingly larger than the quadratic mean diameter of 8.11 inches found in the unreleased control plots. The magnitude of difference, relative to the control, ranged from 0.52 to 1.38 inches.

Volume per acre--Merchantable volume per acre is perhaps the most important response variable in any evaluation of the effects of silvicultural treatments on timber production. In this study, however, we were unable to detect significant differences among treatments in either oak pulpwood volume per acre ($p = 0.23$) or oak sawtimber volume per acre ($p = 0.48$) at the end of the 30th year after release, even though differences among treatment means appeared to be large.

The D15+ and D20+ release treatments and, to a lesser extent, the C15+ and C20+ treatments, appeared to produce more oak pulpwood volume per acre by the end of the 30th year after release than did the unreleased control or the C25+ and D25+ treatments (fig. 5). Oak pulpwood volumes for the four most severe release treatments ranged from 957 to 1,295 cubic feet per acre, in contrast to only 711 cubic feet per acre in the control plots. However, these differences among treatments were not statistically significant.

Similarly, the D15+ and D20+ treatments appeared to produce more oak sawtimber volume per acre than did any of the other treatments, including the control, 30 years after release (fig. 6). Oak sawtimber volumes for the D15+ and D20+ treatments were 1,785 and 1,435 board feet per acre (Doyle scale), compared to only 661 board feet per acre in the control plots. The D25+ treatment plots contained no oak sawtimber volume at all. These differences among treatments were not statistically significant.

We found much variation among rows within treatments for both of these response variables but particularly in oak sawtimber volume per acre. This large variation within treatments probably thwarted our ability to detect statistically significant differences among treatment means, even though it appears that those differences likely exist.

Free-to-grow oaks per acre--At the time the study was established, there were 283 free-to-grow Nuttall oaks per acre, averaged across all plots. Plot means ranged from 204 to 409 free-to-grow oaks per acre, with no significant differences ($p = 0.22$) among plots before application of release treatments (fig. 7).

By the end of the 11th year after release, there were more free-to-grow oaks per acre in the most severe treatment plots than in the least severe treatment plots and in the unreleased control plots (fig. 7). The number of free-to-grow oaks in the most severe treatment plots ranged from 237 to 328 per acre, with the D15+ and D20+ treatment plots containing the largest number of free-to-grow oaks, 328 and 274 per acre, respectively. The unreleased control plots had only 145 free-to-grow oaks per acre. Free-to-grow oaks in the D15+ and D20+ treatment plots represented 62 and 61 percent of the total number of oaks per acre, respectively, whereas only 36 percent of the oaks in the unreleased control plots were free-to-grow. However, we were unable to detect significant differences ($p = 0.15$) among treatments in the number of free-to-grow oaks per acre.

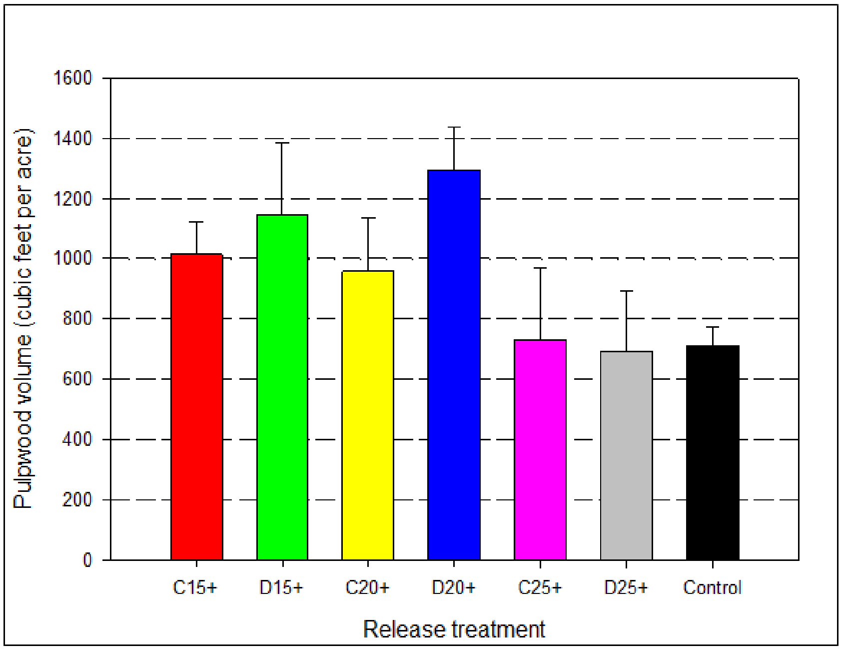

Figure 5--Pulpwood volume per acre (± SE) of Nuttall oak, by treatment, 30 years after application of seven release treatments. Means followed by the same letter are not significantly different at the 0.05 level of probability.

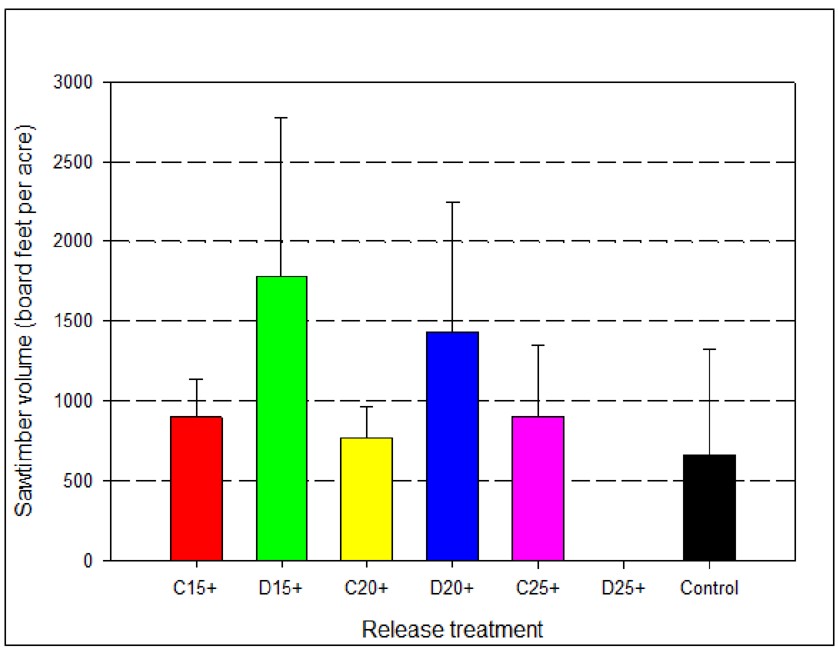

Figure 6--Sawtimber volume per acre (± SE) of Nuttall oak, by treatment, 30 years after application of seven release treatments. Means followed by the same letter are not significantly different at the 0.05 level of probability.

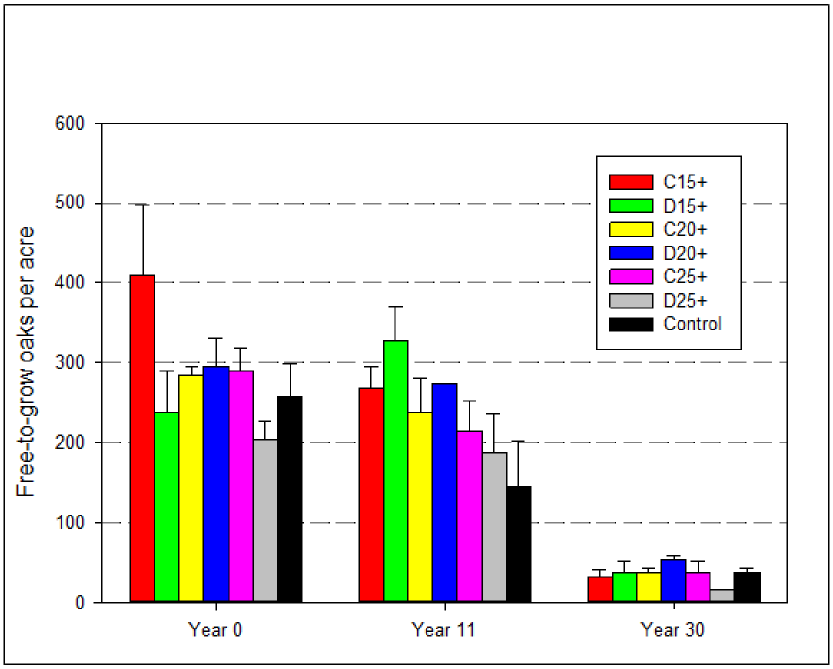

Figure 7--Number of free-to-grow Nuttall oak trees per acre (± SE), by treatment, immediately before release (year 0) and 11 and 30 years after application of seven release treatments. Means within each year followed by the same letter are not significantly different at the 0.05 level of probability.

Between the 11th and 30th years after release, the number of free-to-grow oaks per acre declined drastically across all treatments (fig. 7). There were 32 to 38 free-to-grow oaks per acre 30 years after 5 of the 7 release treatments, including the unreleased control. The D20+ treatment plots averaged 54 free-to-grow oaks per acre, while the D25+ plots averaged only 16 per acre. We found no significant differences ($p = 0.29$) among treatments in the number of free-to-grow oaks per acre.

The drastic decline in the number of free-to-grow oaks per acre across all treatments most likely was due to the destructive effects of the 1994 ice storm. Many oaks across the plantation had small, ragged crowns as a result of severe limb breakage suffered during the storm. Many of these oaks were in a dominant or codominant crown position but were assigned the intermediate crown class due to poor crown condition. Under the general definition that free-to-grow trees receive full sunlight from directly overhead, most of these trees should have been rated as free-to-grow. However, because we used crown class as a surrogate for free-to-grow status, these trees were not rated as free-to-grow. Consequently, the destructive nature of the 1994 ice storm, along with our method of assessment of free-to-grow status, likely resulted in an underestimate of the true number of free-to-grow oaks per acre across all release treatments.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

The destructive effects of the 1994 ice storm reduced our ability to detect significant differences among treatments in response to the seven release treatments. It is difficult to determine the effects of a silvicultural operation on tree growth after an ice storm destroys much of the tree's crown and limits its ability to grow. Even though all trees, including non-oak competitors, were affected, the ice storm may have damaged free-to-grow oaks more severely because they were more exposed to ice accumulation than overtopped oaks and thus more susceptible to crown damage.

Large variation within treatments also likely contributed to our inability to detect significant differences among treatments. Consequently, our statistical analyses revealed significant differences among treatments only for 11-year quadratic mean diameter and 30-year basal area per acre, even though observed differences among treatment means often were quite large.

Despite the lack of a preponderance of statistical evidence to strongly support our view, we are confident that release treatments that removed competitors as tall as the oaks (C15+ and D15+) and 1⅓ times as tall as the oaks (C20+ and D20+) improved long-term growth and development of direct-seeded Nuttall oak saplings. In fact, the moderately severe C20+ and D20+ release treatments are more attractive to landowners because they are as effective but less costly than the more severe C15+ and D15+ release treatments.

Means associated with the D20+ release treatment, in comparison with the other six treatments, consistently ranked at or near the top for all response variables evaluated in this study. In other words, the release treatment that most improved long-term growth and development of direct-seeded Nuttall oak saplings is to deaden all naturally regenerated competitors 1⅓ times as tall or taller than the direct-seeded oaks (D20+).

A comparison of the stand conditions produced 30 years after application of the D20+ release treatment with the stand conditions associated with the unreleased control in this 41-year-old Nuttall oak plantation supports our conclusion. The released stand has 215 oaks per acre, of which 54 are in a free-to-grow position. Oak basal area averages 105 square feet per acre; quadratic mean diameter of these oaks is 9.5 inches. The stand supports 1,295 cubic feet of oak pulpwood per acre and 1,435 board feet (Doyle scale) of oak sawtimber per acre. In contrast, the unreleased stand contains 183 oaks per acre, of which 38 are free-to-grow. Oak basal area is only 66 square feet per acre; quadratic mean diameter is 8.1 inches. The stand has 711 cubic feet of oak pulpwood per acre and 661 board feet (Doyle scale) of oak sawtimber per acre. The released stand contains nearly twice as much oak pulpwood volume and over twice as much oak sawtimber volume as does the unreleased stand.

The released stand is clearly better developed than the unreleased stand. It has a larger oak component than the unreleased stand. Individual oaks are larger and more numerous. Based on the stocking guide for southern bottomland hardwoods developed by Goelz (1995), oak stocking in the released stand is 96 percent, indicative of a fully stocked stand that will need to be thinned in the near future. In contrast, oak stocking in the unreleased stand is only 62 percent, indicative of a less-well-developed stand that must grow another 15 years or so before a thinning is warranted. The released stand has 54 free-to-grow oaks per acre, whereas the unreleased stand has 38 free-to-grow oaks per acre. Clatterbuck and Hodges (1988) suggested that 60 cherrybark oak crop trees per acre are ideal in sawtimber-sized, cherrybark oak-sweetgum stands. The released stand in our study is much closer to this ideal number of oak crop trees per acre than is the unreleased stand. Furthermore, thinning the released stand in the near future likely will increase the number of oak crop trees per acre. In contrast, postponement of thinning for another 15 years in the unreleased stand likely will reduce the number of oak crop trees per acre even further.

The Nuttall oak plantation produced in response to the D20+ release treatment has attained a more advanced stage of development than has the unreleased stand. It is currently more productive and has greater potential for future value growth than the unreleased stand. As such, the released stand is more attractive to landowners interested in production of valuable oak sawtimber.

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LITERATURE CITED

Broadfoot, W.M. 1976. Hardwood suitability for and properties of important midsouth soils. Res. Pap. SO-127. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 84 p.

Clatterbuck, W.K.; Hodges, J.D. 1988. Development of cherrybark oak and sweet gum in mixed, even-aged bottomland stands in central Mississippi, U.S.A. Canadian Journal of Forest Research. 18(1): 12-18.

Conover, D.F.; Ralston, R.A. 1959. Results of crop-tree thinning and pruning in northern hardwood saplings after nineteen years. Journal of Forestry. 57(8): 551-557.

Della-Bianca, L. 1975. An intensive cleaning of mixed hardwood saplings – 10-year results from the southern Appalachians. Journal of Forestry. 73(1): 25-28.

Downs, A.A. 1942. Early responses to weedings in some eastern mountain hardwoods. Journal of Forestry. 40(11): 865-872.

Goelz, J.C.G. 1995. A stocking guide for southern bottomland hardwoods. Southern Journal of Applied Forestry. 19(3): 103-104.

Lamson, N.I.; Smith, H.C. 1978. Response to crop-tree release: sugar maple, red oak, black cherry, and yellow poplar saplings in a 9-year-old stand. Res. Pap. NE-394. Broomall, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 8 p.

Johnson, R.L.; Krinard, R.M. 1988. Development of Nuttall oak following release in a sapling-sized stand. Southern Journal of Applied Forestry. 12(1): 46-49.

Oliver, C.D. 1978. The development of northern red oak in mixed stands in central New England. Bull. No. 91. New Haven, CT: Yale University, School of Forestry and Environmental Studies. 63 p.

Pettry, D.E.; Switzer, R.E. 1996. Sharkey soils in Mississippi. Bull. 1057. Mississippi State, MS: Mississippi State University, Mississippi Agricultural and Forestry Experiment Station. 37 p.

INFLUENCE OF WEATHER AND CLIMATE VARIABLES ON THE BASAL AREA GROWTH OF INDIVIDUAL SHORTLEAF PINE TREES

Pradip Saud, Thomas B. Lynch, Duncan S. Wilson, John Stewart, James M. Guldin, Bob Heinemann, Randy Holeman, Dennis Wilson, and Keith Anderson[1]

An individual-tree basal area growth model previously developed for even-aged naturally occurring shortleaf pine trees (*Pinus echinata* Mill.) in western Arkansas and southeastern Oklahoma did not include weather variables. Individual-tree growth and yield modeling of shortleaf pine has been carried out using the re-measurements of over 200 plots permanently established on the Ozark and Ouachita National Forests during the period 1985-1987. Different basal area growth models for shortleaf pine have been proposed previously, such as a model that was part of a distance-independent individual-tree simulator (Lynch and others 1999) and a model that utilized nonlinear mixed modeling of basal area growth (Budhathoki and others 2008). However, none of the previous studies incorporated the influence of the weather and climate variables in the individual-tree growth prediction models.

Change in forest productivity is the response of trees subject to varying temperature and precipitation over long time scales (Boisvenue and Running 2006). Monitoring of tree performance and response to changes in weather and climate is useful in understanding the limiting factors for forest growth on particular sites (Miller and others 2004). We used the geographic coordinates of the plots to obtain weather and climate variables in an attempt understand the influence of those climatic variables in a shortleaf pine individual-tree basal area growth model.

The GPS location of the plot was used to select the nearest weather station for each of the 129 permanent 0.2-acre fixed-radius plots that we used in this study. Maximum air temperature, average air temperature, and precipitation recorded for the weather station nearest to each plot were used as periodic climate variables. Four periodic measurements of individual trees on each plot were used to develop data for three growth periods on each plot. Lynch and others (1999) and Budhathoki and others (2008) estimated parameters in nonlinear model 1 (base model) using ordinary least squares. In this base model, average annual individual-tree basal area growth was predicted using explanatory variables, including midpoint individual-tree basal area, maximum expected individual-tree basal area for shortleaf pine trees, stand age for the plot, and the ratio of individual-tree diameter to the quadratic mean plot diameter. Weather variables including periodic average daily maximum temperature, periodic average daily mean temperature, and periodic average daily precipitation were added in the base model to construct model 2. Daily averages were for a growing season extending from March 1 to September 30.

We used nonlinear modeling methods to estimate parameters in an individual-tree basal area growth model for shortleaf pine using the Proc NLIN procedure with SAS 9.3, and nls2 package with R 2.15.2. The equation used in model 2 showed some potential for weather variables to improve predictions for the annual basal area growth of individual shortleaf pine trees. Parameter estimates for both models were significant (α = 0.05). The estimates of parameters associated with tree measurement variables were similar in both models 1 and 2. Table 1 presents means and standard

[1]Graduate Research Assistant, Professor, Assistant Professor, and Post-doctoral Research Associate, respectively, Oklahoma State University, Department of Natural Resource Ecology and Management, Stillwater, OK 74078; Supervisory Ecologist, USDA Forest Service Southern Research Station, Hot Springs, AR 71902; and Senior Superintendent, Research Specialist, Research Specialist, and Forestry Technician, respectively, Oklahoma State University Kiamichi Forest Resources Center, Idabel, OK 74745.

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Table 1--Summary of the data used to fit parameters to an individual-tree basal area growth model 1 and model 2 for Growth Period 1 (GP1), Growth Period 2 (GP2), and Growth Period 3 (GP3)

	----------GP1-----------		---------GP2-----------		----------GP3----------	
Variables	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.
Average BA[a] growth per tree (*ft^2/year*)	0.0123	0.0105	0.0138	0.0123	0.0147	0.0125
Mid-point BA for tree in period (*ft^2*)	0.3540	0.3646	0.4168	0.4013	0.5771	0.4529
Stand age (*years*)	41.681	18.724	46.677	18.726	50.777	18.221
Midpoint value of stand BA (*ft^2/acre*)	107.318	29.017	127.033	33.895	108.693	35.481
Quadratic mean diameter (*inches*)	7.451	3.230	8.223	3.203	9.426	2.935
Avg. daily max. temperature (^oF)[b]	81.657	2.684	80.187	2.818	81.947	3.465
Avg. daily mean temperature (^oF)[b]	69.865	1.915	69.041	2.117	70.378	2.343
Avg. daily precipitation (*inches*)[b]	0.1619	0.0163	0.1515	0.0153	0.1315	0.0149

[a]BA = basal area.
[b]Average over growing season (1st March to 30th September).

(a)
(b)

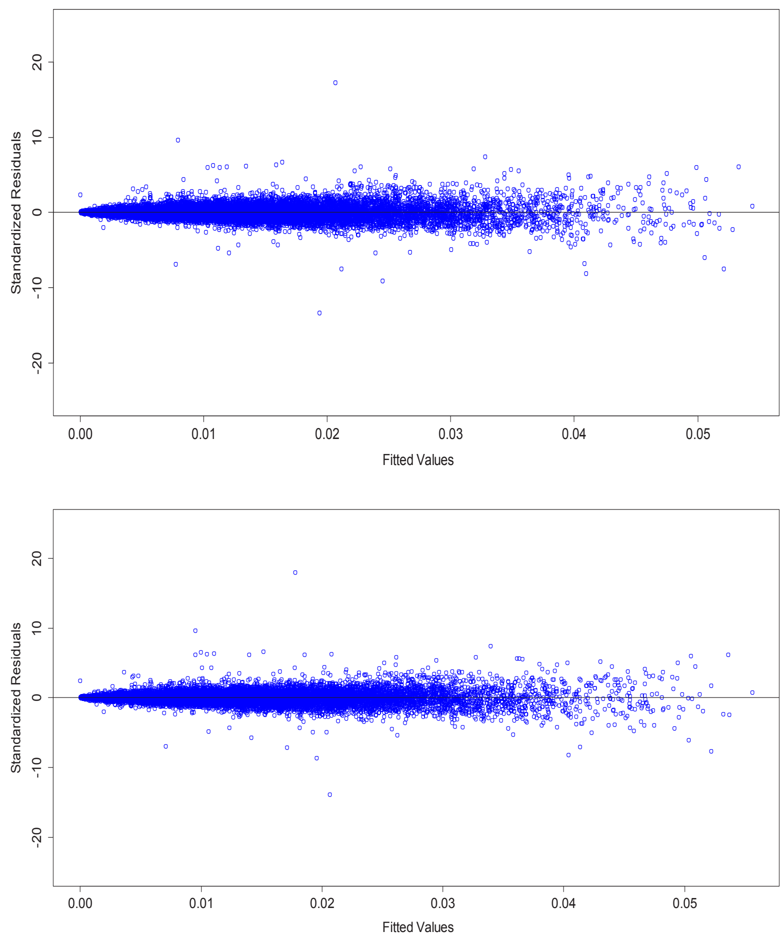

Figure 1—Standardized residuals versus fitted values of a basal area growth model for shortleaf pine using: (a) model 1 and (b) model 2.

deviations for variables used in the model by growth period for three periods. The mean square error of model 1 was 0.000058 and of model 2 was 0.000055, indicating that addition of weather and climate variables reduced the mean square error. When compared using the corrected total sum of squares, we found that the fit index (proportion of total variation explained by the model) of model 2 (0.6169) was slightly better than model 1 (0.5964). The summary of fit statistics for the models were as follows: (a)Akaike's information criterion (AIC) of model 1 was -100540.1 and of the model 2 was -101381; (b) Bayesian information criterion (BIC) of model 1 was -100479.6 and of model 2 was -101297.8; and (c) the residual standard error of model 1 was 0.007144 and of model 2 was 0.006935.

We found that non-linear least squares modeling methods with weather variables used to estimate parameters in an individual-tree basal area growth model for naturally occurring shortleaf pine indicated better fit than a model without weather variables. A small but significant influence of weather variables including precipitation and average air temperature was observed in the modified individual-tree basal area growth model of shortleaf pine.

LITERATURE CITED

Boisvenue, C.; Running, S.W. 2006. Impacts of climate change on natural forest productivity- evidence since the middle of the 20th century. Global Change Biology. 12: 862-882.

Budhathoki, C.B.; Lynch, T.B.; Guldin, J.M., 2008. Nonlinear mixed modeling of basal area growth for shortleaf pine. Forest Ecology and Management. 255: 3440-3446.

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.

Miller, S.D.; Goulden, M.L.; Menton, M.C. [and others]. 2004. Biometric and micrometeorological measurements of tropical forest carbon balance. Ecological Applications. 14(4) Supplement: S114-S126.

Conservation

Moderators:

Tom Dean
Louisiana State University
School of Natural Renewable Resources

and

Wayne Clatterbuck
University of Tennessee
Department of Forestry, Wildlife, and Fisheries

MORE THAN JUST TIMBER: SILVICULTURAL OPTIONS AND ECOSYSTEM SERVICES FROM MANAGED SOUTHERN PINE STANDS

Don C. Bragg, Jamie L. Schuler, Matthew H. Pelkki, D. Andrew Scott, and James M. Guldin[1]

Abstract--The dramatic decline of timber harvests on public lands in the western United States (U.S.) has helped intensify silviculture in the southern U.S. Today, intensive southern pine management usually involves establishment of plantations using site preparation, genetically improved seedlings, chemical fertilization and competition control, early stand density regulation, and increasingly shorter rotations. Little is known about the consequences of this intensification on the ecosystem services provided by pine-dominated forests. Using a synthesis of field studies, simulations, and literature reviews, we compared the impacts of different management options on key services such as diversity, forest productivity (including carbon sequestration), and erosion and other site-related qualities. This review suggests that naturally regenerated pine stands tend to be more structurally and compositionally diverse than plantations, especially as management intensity decreases. Currently, well-managed naturally regenerated pine stands yield only 50 to 90 percent of the wood fiber produced by plantations but in forms more conducive to long-term sequestration in structures. Carbon sequestration is largely a function of stand density, treatment timing, and what is counted as "stored carbon." Plantation establishment also typically involves more soil disturbance, thereby increasing the potential for short-term sedimentation, soil compaction, and drainage issues and may provide accelerated problems with invasive species. Because of the greater tangible cash value of timber yield as an ecosystem service, natural regeneration will not replace pine plantation silviculture for landowners focused largely on commodity production. However, since family forest landowners control 70 percent of southern forests, the combination of acceptable wood production and better ecosystem services from naturally regenerated pine-dominated stands should present opportunities for a subset of those interested in multiple resource values.

INTRODUCTION

Historically, naturally regenerated stands have dominated the silviculture of southern timberlands. Prior to widespread Euroamerican settlement, forests in this region naturally originated by seed or sprout. The first efforts to manage forests across the South depended almost exclusively on natural regeneration whether following uneven- or even-aged silviculture; it was not until the mid-20th century that plantations became a viable option. The growth of plantation-based pine management accelerated after 1980, with extensive (landscape-scale) forest type conversions by corporations and other larger private landowners (Wear and others 2007, Wear and Greis 2012). Even with this widespread conversion, planted pine stands are found on only 27 percent of the coastal plain forest area in the southeastern U.S. (Wear and Greis 2012).

Recent ownership trends have witnessed vertically integrated forest products companies being supplanted (in most instances) by various investment-related ownerships (Wear and Greis 2012). Over the last few decades, a dramatic decline of timber produced on public lands in the western U.S. coupled with other technological and silvicultural advances have helped intensify silviculture in the southern U.S. regardless of stand origin or ownership (Wear and Greis 2012). Today, southern pine management increasingly involves site preparation, genetically improved seedlings (in plantations), chemical fertilization and competition control, early stand density regulation, and shorter rotations intended to produce sawlog-sized stems of more or less uniform size (Allen and others 2005, Rousseau and others 2005). Given continued interest in increasing silvicultural intensity in southern pine forests (Allen and others 2005), it is incumbent on forest managers, researchers, and landowners to better understand the socioeconomic and ecological implications of this trend.

For example, little is known about the consequences of silvicultural intensification on ecosystem services provided by pine-dominated forests. There are numerous definitions of ecosystem services, but useful ones consider the direct and indirect contributions of ecosystems to human well-being and incorporate a range of functions, including

[1]Research Forester, USDA Forest Service, Southern Research Station, Monticello, AR 71656; Assistant Professor, West Virginia University, Division of Forestry and Natural Resources, Morgantown, WV 26506; Professor, University of Arkansas-Monticello, School of Forest Resources, Monticello, AR 71656; Research Soil Scientist, USDA Forest Service, Southern Research Station, Normal, AL 71360; and Supervisory Ecologist, USDA Forest Service, Southern Research Station, Hot Springs, AR 71902.

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“supporting services” (for example, nutrient cycling), “provisioning services” (for example, timber, food, energy), “regulating services” (for example, pollination, water purification), and “cultural services” (for example, aesthetics, recreational experiences) (Braat and de Groot 2012). In this paper, we consider a number of the implications of forest conversions from naturally regenerated pine, pine-hardwood, and hardwood-dominated forests to pine plantations in the southeastern U.S., with an emphasis on ecosystem services.

MATERIALS AND METHODS

Using a synthesis of field studies, simulations, and literature reviews, we compared the impacts of different management options on ecosystem services such as diversity, forest productivity, carbon sequestration, and erosion and other site-related qualities. This literature review and synthesis was not intended to be an exhaustive treatise on this topic but rather a brief summary of some key points related to silvicultural practices and ecosystem services in southern pine-dominated forests.

RESULTS AND DISCUSSION

Some impacts of the conversion of naturally regenerated pine-dominated forests into pine plantations are self-evident. For example, it is likely that for any given stand, converting from a complicated overstory found in a naturally regenerated multi-aged pine-hardwood forest to a tightly regulated monoculture of improved loblolly pine (*Pinus taeda* L.) may increase certain ecosystem services (for example, production of wood fiber) at the expense of others (for example, decreased biodiversity due to the loss of structural complexity). Other impacts may not be as apparent, at least not initially.

Diversity

Within naturally propagated populations, southern pines have considerable amounts of genetic diversity (Xu and others 2008) which allows these species to adapt to changing environmental conditions, forest health threats, and resource competitors. Supplanting this variability with genotypes selected for a limited number of traits (for example, early growth rate or fusiform rust resistance) may leave plantations vulnerable to other unanticipated influences, such as extreme droughts, ice storms, wind, pests, or other pathogens. By definition, monoculture plantations (as well as pure stands of natural original) have very low tree species richness. In practice, many monocultures, even intensively managed ones, have a number of other tree species present. Harvesting permits the reestablishment of early successional species that may be rare or absent in unmanaged stands of natural origin, thereby increasing community diversity at least temporarily (Jeffries and others 2010, Jones and others 2009). Though it is not unusual for hardwoods to be a notable fraction of the stocking in many stands, the use of broad spectrum herbicides and/or mechanical treatments in intensively managed southern pine forests can greatly reduce if not eliminate non-pine competitors, at least temporarily (Jones and others 2012).

Other aspects of biodiversity are also influenced at least in part by the management of southern pine-dominated forests. Locality-focused studies have often found limited impacts of site preparation and vegetation management treatments on wildlife in landscapes dominated by similar early successional communities (Hanberry and others 2012, 2013; Lane and others 2011). Thinning dense even-aged stands also permits the development of preferred forage species for many animals (Peitz and others 2001). However, most of these studies also recognize that certain taxa, especially wildlife associated with interior forest habitats, are rare or absent from these intensively managed landscapes; for instance, Lane and others (2011) reported a low abundance of interior bird species in pine plantation-dominated landscapes. Other long-term meta-analyses on certain functional groups have noted marked declines of populations and even widespread extirpations, such as seen in some forest birds in the southeastern U.S. between 1966 and 2006 (USDA Forest Service 2011) coincidental to the large-scale transition to plantation-dominated landscapes. The conversion of naturally regenerated forests to plantations probably also impacts other floral, fungal, and faunal communities, although it is hard to disentangle certain aspects of habitat requirements from others. As an example, studies of forest-dwelling bats in the southeastern U.S. have found young pine plantations did not have major impacts on foraging opportunities, especially those taking place above the canopy, but roost habitat for species requiring large trees or snags with loose bark or cavities were considerably lessened

(Elmore and others 2005, Menzel and others 2005).

Diversity can refer to more than just genetic, taxonomic, or stand compositional variability. Physical structure also plays a significant role in shaping the biotic communities associated with forest type, although not always in a clear pattern. Naturally regenerated pine stands can be more structurally diverse than plantations. Part of this may arise from the impacts of competition control activities, even though thinning of dense overstories can release understory plants (Jones and others 2012, Peitz and others 2001). For southern pine stands managed with uneven-aged silvicultural techniques, a broad range of size classes is typically present across the stand (Baker and others 1996, Guldin 2011, Reynolds 1959). Given the need for abundant sunlight to encourage the establishment of the relatively shade-intolerant southern pines, the practice of uneven-aged silviculture requires much of the stand to be kept at low density via numerous large gaps in the pine-dominated overstory (Baker and others 1996). The implementation of uneven-aged silviculture in the shade-intolerant southern pines therefore tends to produce a stand with considerable horizontal and vertical structure. Some of the impacts of structural simplification caused by converting more complex natural stands into plantations can be offset by the incorporation of structural complexity via reserves or corridors such as riparian management zones commonly incorporated in "certified" ownerships. Researchers have found that such habitats can help certain fauna persist in otherwise unfavorable stand conditions (Hein and others 2008, Lane and others 2011).

Forest Productivity and Carbon Sequestration

Our review suggests that carbon sequestration is largely a function of stand density, treatment timing, and the definition of stored carbon. Research has shown that intensively managed southern pine plantations produce significantly greater wood volume than most naturally regenerated, well-managed pine forests (Allen and others 2005, Stanturf and others 2003). Bragg and Guldin (2010), using a very simplified modeling approach for well-managed loblolly pine plantations and even- and uneven-aged loblolly and shortleaf pine (*P. echinata* Mill.)-dominated natural stands reported that plantations produced about twice the tree biomass and harvested volumes of uneven-aged stands and at least 10 percent more net biomass than the natural even-aged stand (fig. 1). Some of this disparity is directly attributable to the fact that uneven-aged southern pine stands tend to be understocked, a deliberate result of the treatments needed to ensure continuous regeneration of the shade-intolerant pines (Baker and others 1996, Reynolds 1959). In addition to cultural treatments such as site preparation, use of genetically improved seedlings, and early competition control, even-aged stands also tend to receive mid-rotation thinnings earlier in the rotation and may also be fertilized in addition to chemical and/or mechanical releases, thereby boosting their yields. The prospects for even further gains in plantation volume growth and wood quality are considerable (Allen and others 2005, Blazier and others 2004).

One possible but rarely considered productivity-related consequence of short rotations in low-density southern pine plantations is the high proportion of juvenile wood produced under these conditions (Clark and others 1994, Megraw 1985). Juvenile or "crown" wood has significantly lower specific gravity than mature wood, implying a substantially lower fraction of carbon. For stems of equal size, slower growing pines have a higher proportion of mature wood, and hence may actually sequester more carbon per unit of wood volume produced. Few studies have considered the large-scale impacts of this differential pattern in carbon accumulation, but it probably accounts for some of the difference in volume production between plantation and naturally-regenerated stands. Similarly, if fiber from plantations is converted to pulp, residual fuels, and paper products, the long-term sequestration benefit is less than for lumber products used in housing manufactured from logs commonly produced in naturally regenerated stands (Lippke and others 2011).

Plantations optimized for volume growth using simplified genetics, fertilization, competition control, and density management will have different impacts on overall ecosystem productivity than naturally regenerated forests (Tian and others 2012), but how these treatments will respond to climate change is still uncertain. Shorter rotations can permit

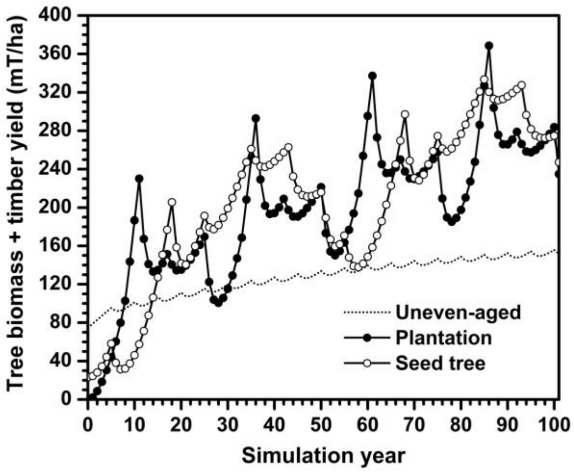

Figure 1--Projected net biomass accrual (standing live trees plus harvested yield minus decomposition or other losses) differences between various southern pine silvicultural techniques. Adapted from Bragg and Guldin (2010).

landowners to more quickly adjust their management to changing climate, but this may be less of an issue in more genetically diverse natural forests. Pine growth is typically considered to be resource-limited in the southeastern U.S., and moisture can be one of those constraining factors (Albaugh and others 2004, Allen and others 2005). Some models predict only limited climate-change-related productivity impacts due to compensating effects of longer growing seasons, CO_2 fertilization, and possible increases in overall precipitation (Sun and others 2000). However, it can be expected that greater moisture demands will be placed on the soil by faster growing trees selected for higher leaf area and net photosynthetic rate (Teskey and others 1987). Assuming it is obtainable, greater soil moisture consumption by such trees can reduce the water yield for other parts of the ecosystem and may decrease the quantity and quality available for human consumption if groundwater and overland flow are diminished (Farley and others 2005, Licata and others 2008). How moisture-limited ecosystem productivity may interact with future droughts and differences in overall stand stocking (low-density plantations versus higher-density stands) requires further study.

There are also other ecosystem services that may benefit from the continuous presence of large-diameter pines in uneven-aged stands, especially those with structural characteristics unique to large live trees. Big pines can provide opportunities for certain types of habitat usually unavailable in younger, faster growing plantations. For example, cavities in large, older, red heart [*Phellinus pini* (Thore:Fr.) Ames]-decayed pines are the nesting habitat of red-cockaded woodpeckers [*Picoides borealis* (Vieillot)]), a federally listed endangered species (Jackson and others 1979). Larger pine snags with loose attached bark are also preferred roost habitat for certain bat species (Perry and Thill 2007). Obviously, adjacent stands with large pines may be capable of providing habitat, but as the overall forest matrix changes, certain species face declines that may threaten them with extinction, as has been seen with the red-cockaded woodpecker in the changing pine forest demographics of the southeastern U.S. (Jackson and others 1979).

Erosion and Other Site Impacts

With very few exceptions, the process of logging disrupts soil surfaces. Site preparation practices across much of the southeastern U.S. often involve the use of deep ripping plows (subsoiling), disking, and/or bedding. These are often done to improve soil conditions on sites that have subsurface rooting depth limitations or those that have been rutted or compacted by past land-use practices, including timber harvest (Carter and others 2006, Fox and others 2007, Lincoln and others 2007, Miwa and others 2004). By design, these treatments penetrate the soil and disturb soil horizons, roots, existing drainage patterns, and vegetation on the surface, all of which are intended to benefit early pine growth and survival. On sites with even greater water drainage issues, ditches were sometimes dug to improve pine survival and growth. Only rarely do naturally regenerated pine-dominated stands receive such intensive site preparation treatments. In these stands, most soil impacts arise from logging, including felling and skidding which, obviously, also occur when harvesting plantations. Hence, naturally regenerated stands are less likely to experience any detrimental effects of these site preparations or, conversely, enjoy any potential benefits.

Unfortunately, while much has been reported on soil nutrient dynamics, carbon sequestration, surficial water movement, and biological diversity following intensive site preparation, the results have been inconclusive (Scott and others 2006). For example, bedding impacts the recovery of water table depth following the logging of wet sites (Xu and others 2002) but had no effect on arthropod diversity on another location (Bird and others 2000). Site disturbance is often cited as a reason why invasive species become more widespread, especially if the equipment used to conduct the treatments is contaminated with propagules (Miller and others 2010). Undoubtedly, the physical disruption of previously intact soils may also negatively impact cultural (archeological) resources and lead to localized erosion problems if not properly ameliorated. The conversion of stands of natural origin to plantations can affect a number of soil properties, including rates of forest floor accumulation, nitrogen dynamics, and moisture levels (Scott and Messina 2010), although such changes do not inherently have positive or negative consequences on ecosystem services. The large-scale transformation of landscapes into pine plantations requires further study, especially given the scope and rate of these alterations.

CONCLUSIONS

Well-tended natural-origin pine stands yield only a fraction of the wood volume produced by intensively managed plantations for a given rotation length. For this reason alone, natural regeneration will not soon replace pine plantation silviculture for landowners focused solely on commodity production. In fact, some anticipate that the development of southern pine-based bioenergy will further accelerate the rate of conversion from natural stands to plantations under the current system of pricing and incentives (Abt and others 2012). However, the combination of acceptable timber production and (in many cases) more complementary ecosystem services present opportunities for the greater than 70 percent of southern forests controlled by non-industrial landowners. For instance, hardwoods and slower-growing, larger pines sequester more carbon per unit volume of wood than fast-growing, smaller pines, so reconsidering this provisioning service in terms of total ecosystem carbon accumulated (including that stored long-term in dead wood and belowground) rather than merchantable bole volume produced may show different outcomes when comparing these silvicultural systems (Sohngen and Brown 2006). While some have inferred the retention of naturally regenerated forests on lands otherwise suited for more productive pine plantations as a measure of willingness to pay for some types of ecosystem services (Raunikar and Buongiorno 2006), this “trade-off” could also result from either a lack of knowledge or interest in silviculture rather than more altruistic motivations. However, many surveys of non-industrial private landowners in the southeastern U.S. have identified financial benefits as only one of many motivations behind land ownership and management decisions (Butler and Leatherberry 2004, Davis and Fly 2010, Kluender and Walkingstick 2000), suggesting that opportunities for ecosystem services beyond wood production exist (McIntyre and others 2010).

If other ecosystem services are considered than the production of wood volume, the values of natural- and plantation-origin pine forests may be more comparable, although not necessarily equivalent. Unfortunately, unlike the relatively straightforward determination of certain provisioning services, quantifying the influence of silvicultural regime on supporting, regulating, and cultural services is not as easy. First, we have little concrete data on the socioeconomic contributions of certain forest-based ecosystem services (such as pollination, water purification, aesthetics, recreation) or derived components such as the non-timber flora and fauna found in those forests. After all, what is the cash value of a box turtle (*Terrapene carolina*) in Arkansas (fig. 2) or a scenic view in Mississippi? Who pays for that, and how would that payment be made? Second, there is only limited documentation regarding the specific system responses of conversion from natural stands to pine plantations in terms of ecosystem services. Without further unbiased research and analysis, it is not possible to unequivocally state that one is better, worse, or even the same regarding supporting, regulating, and cultural services. Finally, evolving technological and regulatory environments will likely continue to alter the potential for ecosystem services to influence southern pine silviculture well into the future; a national or even global adoption of a compliance-based cap-and-trade system for carbon with rigorous contingencies for wood products and forest-based carbon sequestration could dramatically alter the economics of the region.

It is almost certain that further intensification of timber management is likely in southern pine-dominated landscapes. Most econometric analyses of future forest conditions in this region forecast that pine plantations will continue to displace naturally regenerated forests of all types, including pine-dominated, upland hardwood-dominated, and mixed pine-hardwood

Figure 2--The box turtle conundrum for pricing ecosystem services. Because this protected species does not currently have any value in the commercial pet trade, as a human food source, or as a sport hunting species, some would assign it a value of $0. However, many states assign a restitution value for animals that could act as the basis for valuation, and box turtles do have a potential monetary value in their role as part of the biota of an ecosystem consuming invertebrates that can be forest pests and disseminating seeds of desirable plants. Further, many (most?) people are likely willing to pay in order to ensure that this and other native taxa collectively remain. This could even be extended to the costs associated with habitat preservation and species conservation (Dalrymple and others 2012).

forests (Abt and others 2012, Sohngen and Brown 2006, Wear and Greis 2012, Zhang and Polyakov 2010). Between growing global demand for wood products, declining supplies from other regions (such as the decrease in production from beetle-killed forests in western North America), and a shrinking available forest land base in the southeastern U.S., pressures on remaining southern pine forests to continue to produce are not likely to change into the foreseeable future. Other challenges related to the impacts of silvicultural practices on ecosystem services are expected as the climate changes, new invasive species arrive and existing ones expand further, human resource demands continue to strain natural ecosystems, forestlands fragment, and land use practices change. Hopefully, some kind of ecological "tipping point" will not be reached that causes a region-wide collapse in many other tangible but underappreciated ecosystem services.

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LITERATURE CITED

Abt, K.L.; Abt, R.C.; Galik, C. 2012. Effect of bioenergy demands and supply response on markets, carbon, and land use. Forest Science. 58(5): 523-539.

Albaugh, T.J.; Allen, H.L.; Dougherty, P.M.; Johnsen, K.H. 2004. Long term growth responses of loblolly pine to optimal nutrient and water resource availability. Forest Ecology and Management. 192: 3-19.

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.

Baker, J.B.; Cain, M.D.; Guldin, J.M. [and others]. 1996. Uneven-aged silviculture for the loblolly and shortleaf pine forest cover types. Gen. Tech. Rep. SO-118. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 65 p.

Bird, S.; Coulson, R.N.; Crossley, D.A. 2000. Impacts of silvicultural practices on soil and arthropod diversity in a Texas pine plantation. Forest Ecology and Management. 131: 65-80.

Blazier, M.A.; Hennessey, T.C.; Lynch, T.B. [and others]. 2004. Productivity, crown architecture, and gas exchange of North Carolina and Oklahoma/Arkansas loblolly pine families growing on a droughty site in southeastern Oklahoma. Forest Ecology and Management. 194: 83-94.

Braat, L.C.; de Groot, R. 2012. The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosystem Services. 1: 4-15.

Bragg, D.C.; Guldin, J.M. 2010. Estimating long-term carbon sequestration patterns in even- and uneven-aged southern pine stands. In: Jain, T.B.; Graham, R.T.; Sandquist, J., tech. eds. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: proceedings of the 2009 national silviculture workshop. Proc. RMRS-P-61. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 111-123.

Butler, B.J.; Leatherberry, E.C. 2004. America's family forest owners. Journal of Forestry. 102(7): 4-9.

Carter, M.C.; Dean, T.J.; Wang, Z.; Newbold, R.A. 2006. Impacts of harvesting and postharvest treatments on soil bulk density, soil strength, and early growth of *Pinus taeda* in the Gulf Coastal Plain: a Long-Term Soil Productivity affiliated study. Canadian Journal of Forest Research. 36: 601-614.

Clark, A.; Saucier, J.R.; Baldwin, V.C.; Bower, D.R. 1994. Effect of initial spacing and thinning on lumber grade, yield, and strength of loblolly pine. Forest Products Journal. 44(11/12): 14-20.

Dalrymple, C.J.; Peterson, M.N.; Cobb, D.T. [and others]. 2012. Estimating public willingness to fund nongame conservation through state tax initiatives. Wildlife Society Bulletin. 36(3): 483-491.

Davis, M.L.E.S.; Fly, M.J. 2010. Do you hear what I hear: better understanding how forest management is conceptualized and practiced by private forest landowners. Journal of Forestry. 108(7): 321-328.

Elmore, L.W.; Miller, D.A.; Vilella, F.J. 2005. Foraging area size and habitat use by red bats (*Lasiurus borealis*) in an intensively managed pine landscape in Mississippi. American Midland Naturalist. 153(2): 405-417.

Farley, K.A.; Jobbagy, E.G.; Jackson, R.B. 2005. Effects of afforestation on water yield: a global synthesis with implications for policy. Global Change Biology. 11: 1565-1576.

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.

Guldin, J.M. 2011. Experience with the selection method in pine stands in the southern United States, with implications for future application. Forestry. 84(5): 539-546.

Hanberry, B.B.; Edwards, S.L.; Demarais, S.; Jones, J. 2013. Small mammal responses to intensively established pine plantations in coastal plain Mississippi. Southern Journal of Applied Forestry. 37(1): 53-58.

Hanberry, B.B.; Hanberry, P.; Riffell, S.K. [and others]. 2012. Bird assemblages of intensively established pine plantations in coastal plain Mississippi. Journal of Wildlife Management. 76(6): 1205-1214.

Hein, C.D.; Castleberry, S.B.; Miller, K.V. 2008. Sex-specific summer roost-site selection by Seminole bats in response to landscape-level forest management. Journal of Mammalogy. 89(4): 964-972.

Jackson, J.A.; Lennartz, M.R.; Hooper, R.G. 1979. Tree age and cavity initiation by red-cockaded woodpeckers. Journal of Forestry. 77(2): 102-103.

Jeffries, S.B.; Wentworth, T.R.; Allen, H.L. 2010. Long-term effects of establishment practices on plant communities across successive rotations in a loblolly pine (*Pinus taeda*) plantation. Forest Ecology and Management. 260: 1548-1556.

Jones, P.D.; Demarais, S.; Ezell, A.W. 2012. Successional trajectory of loblolly pine (*Pinus taeda*) plantations established using intensive management in southern Mississippi, USA. Forest Ecology and Management. 265: 116-123.

Jones, P.D.; Edwards, S.L.; Demarais, S. 2009. White-tailed deer foraging habitat in intensively established loblolly pine plantations. Journal of Wildlife Management. 73(4): 488-496.

Kluender, R.A.; Walkingstick, T.L. 2000. Rethinking how nonindustrial landowners view their lands. Southern Journal of Applied Forestry. 24(3): 150-158.

Lane, V.R.; Miller, K.V.; Castleberry, S.B. [and others]. 2011. Bird community responses to a gradient of site preparation intensities in pine plantations in the coastal plain of North Carolina. Forest Ecology and Management. 262: 1668-1678.

Licata, J.A.; Gyenge, J.E.; Fernández, M.E. [and others]. 2008. Increased water use by ponderosa pine plantations in northwestern Patagonia, Argentina compared with native forest vegetation. Forest Ecology and Management. 255: 753-764.

Lincoln, M.C.; Will, R.E.; Morris, L.A. [and others]. 2007. Soil change and loblolly pine (*Pinus taeda*) seedling growth following site preparation tillage in the Upper Coastal Plain of the southeastern United States. Forest Ecology and Management. 242(2-3): 558-568.

Lippke, B.; Oneil, E.; Harrison, R. [and others]. 2011. Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. Carbon Management. 2(3): 303-333.

McIntyre, R.K.; Jack, S.B.; McCall, B.B.; Mitchell, R.J. 2010. Financial feasibility of selection-based multiple-value management on private lands in the South: a heuristic case study approach. Journal of Forestry. 108(5): 230-237.

Megraw, R.A. 1985. Wood quality factors in loblolly pine. Atlanta, GA: TAPPI Press. 88 p.

Menzel, J.M.; Menzel, M.A.; Kilgo, J.C. [and others]. 2005. Effect of habitat and foraging height on bat activity in the coastal plain of South Carolina. Journal of Wildlife Management. 69(1): 235-245.

Miller, J.H.; Manning, S.T.; Enloe, S.E. 2010. A management guide for invasive plants in southern forests. Gen. Tech. Rep. SRS-131. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 120 p.

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.

Peitz, D.G.; Shelton, M.G.; Tappe, P.A. 2001. Forage production after thinning a natural loblolly pine-hardwood stand to different basal areas. Wildlife Society Bulletin. 29(2): 697-705.

Perry, R.W.; Thill, R.E. 2007. Roost selection by male and female northern long-eared bats in a pine-dominated landscape. Forest Ecology and Management. 247(1-3): 220-226.

Raunikar, R.; Buongiorno, J. 2006. Willingness to pay for forest amenities: the case of non-industrial owners in the south central United States. Ecological Economics. 56: 132-143.

Reynolds, R.R. 1959. Eighteen years of selection timber management on the Crossett Experimental Forest. Tech. Bull. 1206. Washington, DC: U.S. Department of Agriculture Forest Service. 68 p.

Rousseau, R.; Kaczmarek, D.; Martin, J. 2005. A review of the biological, social, and regulatory constraints to intensive plantation culture. Southern Journal of Applied Forestry. 29(2): 105-109.

Scott, D.A.; Burger, J.A.; Crane, B. 2006. Expanding site productivity research to sustain non-timber forest functions. Forest Ecology and Management. 227(1-2): 185-192.

Scott, D.A.; Messina, M.G. 2010. Soil properties in 35 y old pine and hardwood plantations after conversion from mixed pine-hardwood forest. American Midland Naturalist. 163: 197-211.

Sohngen, B.; Brown, S. 2006. The influence of conversion of forest types on carbon sequestration and other ecosystem services in the south central United States. Ecological Economics. 57: 698-708.

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.

Sun, G.; Amatya, D.M.; McNulty, S.G. [and others]. 2000. Climate change impacts on the hydrology and productivity of a pine plantation. Journal of the American Water Resources Association. 36(2): 367-374.

Teskey, R.O.; Bongarten, B.C.; Cregg, B.M. [and others]. 1987. Physiology and genetics of tree growth response to moisture and temperature stress: an examination of the characteristics of loblolly pine (*Pinus taeda* L.). Tree Physiology. 3: 41-61.

Tian, S.; Youssef, M.A.; Skaggs, R.W. [and others]. 2012. Modeling water, carbon, and nitrogen dynamics for two drained pine plantations under intensive management practices. Forest Ecology and Management. 264: 20-36.

USDA [U.S. Department of Agriculture] Forest Service. 2011. National report on sustainable forests - 2010. FS-979. Washington, DC: U.S. Department of Agriculture Forest Service. 214 p.

Wear, D.N.; Carter, D.R.; Prestemon, J. 2007. The U.S. South's timber sector in 2005: a prospective analysis of recent change. Gen. Tech. Rep. SRS-99. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 29 p.

Wear, D.N.; Greis, J.G. 2012. The Southern Forest Futures Project: summary report. Gen. Tech. Rep. SRS-168. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 54 p.

Xu, S.; Tauer, C.G.; Nelson, C.D. 2008. Genetic diversity within and among populations of shortleaf pine (*Pinus echinata* Mill.) and loblolly pine (*Pinus taeda* L.). Tree Genetics & Genomes. 4: 859-868.

Xu, Y-J.; Burger, J.A.; Aust, W.M. [and others]. 2002. Changes in surface water table depth and soil physical properties after harvest and establishment of loblolly pine (*Pinus taeda* L.) in Atlantic coastal plain wetlands of South Carolina. Soil & Tillage Research. 63: 109-121.

Zhang, D.; Polyakov, M. 2010. The geographical distribution of plantation forests and land resources potentially available for pine plantations in the U.S. South. Biomass and Bioenergy. 34: 1643-1654.

INFLUENCES OF SHELTERWOOD PRESCRIPTIONS TO ABOVE-GROUND CARBON STORAGE AND HERPETOFAUNAL AND SMALL MAMMAL COMMUNITIES

Padraic Conner, Yong Wang, and Callie Jo Schweitzer[1]

Abstract--We examined the response of herpetofaunal and small mammal communities to silvicultural treatments. In addition, differences between silvicultural treatments of carbon storage ratios in trees, shrubs, vines, herbaceous material, course woody debris, and fine woody debris were studied. A complete randomized design with multiple replications, 20 experimental stands of approximately 5 ha each, was used to test three prescriptions: (1) control (no alteration); (2) shelterwood (SW, 30 to 40 percent basal area retention); and (3) oak shelterwood (OSW, herbicide midstory treatment with triclopyr). Drift fences with pitfall and box funnel traps and Sherman live traps were used to assess herpetofaunal and small mammal communities. Above-ground materials for carbon quantification were either collected, dried, and weighed or derived using estimating equations to ascertain carbon ratios. We found more carbon in control and oak shelterwood compared to shelterwood. Overstory trees contained the majority of the carbon within all treatment types and accounted for most of the differences in total carbon between treatment types. In comparison between treatment types, OSW and SW had the highest herbaceous carbon content. Over both years, the most abundant reptile was eastern fence lizard (*Sceloporus undulatus*), the most abundant amphibian was American toad (*Anaxyrus americanus*), and the most abundant small mammal was the white-footed mouse (*Peromyscus leucopus*). Lizards were more abundant in the shelterwood stands in 2011 and 2012 compared to the other treatments.

INTRODUCTION

There has been a growing public awareness and political interest in the limited supply and ecological effects of using fossil fuels to generate energy. This increased awareness has raised interest in renewable bioenergy resources. Forest and agricultural residues are showing promise as a source of biofuels (Winandy and others 2008). Studies have estimated that fossil fuel consumption can be significantly offset with the use of wood product biofuels (De Vries and others 2007, Field and others 2008). This may increase demand for intensive biomass production. Much of this production in the United States could be centered in the Southeast, as it has been estimated that over one half of the nation's recoverable forestry residues are found in the Southeast and south-central regions (Gan and Smith 2006). Forests store carbon as they accumulate biomass, and active management of forests can influence carbon storage in several ways. Applied silviculture prescriptions germane to forested ecosystems result in varying ratios of carbon stocks stored in hardwoods, softwoods, woody shrubs, herbaceous growth, woody debris, and in the soil (Tilman and others 2006).

Forest above-ground structures, or forest biomass, provides habitat for a diverse array of organisms that interact with each other and the environment resulting in a number of important ecosystem functions and services (Ferris and Humphrey 1999). Total above-ground forest biomass is a complex structure that provides habitat and foraging sources for many wildlife species (Lanham and Guynn 1996). Changes in these forest features can affect the density and species composition of wildlife communities as well as individual species (Wang and others 2006). The presence and continued input of dead wood or woody debris in various states of decay are of key significance for biodiversity in managed forest systems (Hansen and others 1991). A variety of studies indicates that changes in woody debris supplies due to forest management can have strong impacts on forest biodiversity (Cromer and others 2007, Nordén and others 2004, Verschuyl and others 2011). Horner and others (2010) also found that moderate disturbance (thinning) resulted in the highest carbon standing stocks and that the lowest carbon storage was found in untreated stands. The sustainability of any short-term gain in biomass or carbon will be impacted by the age, diameter, and species distribution of the residual trees, which may or may not continue to respond over time (D'Amato and others 2011, Hoover and Stout 2007,).

The USDA Forest Service Southern Research Station, Upland Hardwood Ecology and

[1]Master's Student and Professor, respectively, Alabama A&M University, Department of Biological and Environmental Science, Normal, AL 35762; and Research Forester, USDA Forest Service, Southern Research Station, Huntsville, AL 35801.

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Management Research Work Unit implemented a study with partners to address how three treatments affect oak and other hardwood species regeneration and wildlife communities. Effects of the following forest management treatments are currently being examined: (1) shelterwood with prescribed fire (SW); (2) oak shelterwood (OSW); and (3) untreated control. All three treatment types will have all residual trees cut 11 years after initial implementation.

Study Site Description

The study site is located in Grundy County, TN, situated on the Mid-Cumberland Plateau. This property is owned and managed by Stevenson Land Company. The treatment stands are located on the eastern escarpment of Burrow's Cove. The site is just east of the Eastern Highland rim (Smalley 1982). The site is classified as being a true plateau with strongly dissected margins (Smalley 1982), and additionally described by Braun (1950) as being in the Cliff section of Mixed Mesophytic Forest region. Stands were situated on the escarpment. Upland oak site index (SI) is 23 to 24 m, and yellow poplar SI is 30 m (Smalley 1982). Soil classification is Bouldin Stony Loam (NRCS 2007), which is deep and well drained, consisting of 30 percent rocky slopes. The hardwood forest within the stands is comprised of 27 different species with yellow poplar (*Liriodendron tulipfera* L.), sugar maple (*Acer saccharum* Marsh.), white oak (*Quercus alba* L.), pignut hickory [*Carya glabra* (Mill.) Sweet], and northern red oak (*Quercus rubra* L.) as the dominant overstory trees (Cantrell and others 2013). For stems over 25 cm in diameter, the stands have a basal area (BA) of 22.5 m^2/ha and 1,000 stems per ha (SPHA). Following treatment, the basal area (for stems > 25 cm in diameter) of the control stands was 22.8 m^2/ha with 993 SPHA. The OSW had a basal area of 24.0 m^2/ha and 1,060 SPHA, and the SW had 9.7 m^2/ha of basal area and 613 SPHA. For the OSW, the average treated stem diameter was 10.5 cm, and 1,500 SPHA were treated. (Personal communication. 2013. Callie Schweitzer, Research Forester, USDA Forest Service, Southern Research Station, 730-D Cook Avenue, Huntsville, AL 35801).

Experimental Design

The field experiment was originally designed targeting three regeneration treatments and one control replicated five times resulting in 20 experiment units or stands (approximately 5 ha each). Treatment stands were selected by the Forest Service researchers so as to have mature closed canopy stands with trees > 70 years old and without any major anthropogenic or natural disturbances within the last 15 to 20 years.

Modifications were made to the original experimental design. The prescribed burn was not implemented therefore we incorporated these unaltered stands as control stands. One of the oak shelterwood stands and two of the controls were partially harvested on the bottom slope and were not used due to the impact on the animal sampling protocols. One prescribed burn stand was harvested at the same level as the other shelterwoods and was considered a shelterwood stand. These modifications resulted in 17 experimental units: 6 shelterwood stands (SW), 4 oak shelterwood stands (OSW), and 7 control stands.

Shelterwood

The shelterwood method is an even-aged forest management practice that allows the regeneration of a new tree cohort under a partial over-story (Spetich and Graney 2003) due to increased light availability to the understory. The shelterwood harvest prescription used the guidelines of Brose and others (1999). The treatment entailed harvesting of timber with 30 to 40 percent of the original BA retained. Residual trees were retained based on species, diameter, and quality. Trees were harvested by chainsaw felling and grapple skidding. Many dominant and co-dominant oak species were retained in the residual stand. Trees harvested had their crown, limbs, and branches removed on site, leaving the majority of slash within the stand (Cantrell and others 2013).

Oak Shelterwood

The oak shelterwood method is a modified silviculture practice that is used specifically for the purpose of regenerating oak (Loftis 1990). This method reduces the midstory structure via the use of herbicides, with a goal of reducing competition from shade-tolerant trees and increasing light to the understory. In this process, the herbicide treatment is followed by overstory removal once the regenerating oaks reach a height which will allow them to compete with other species upon release (Cantrell and others 2013).

The OSW treatment followed the guidelines of Loftis (1990). The treatment used triclopyr herbicide (Garlon 3A, Dow AgroSciences, LLC) applied to the trunk via the hack and squirt method. Herbicide was applied to competing mid-story trees with > 5 cm and < 25 cm in diameter at breast height (d.b.h.). Initial treatment was repeated in fall/winter of 2009 due to defective active ingredient in the original chemical used.

Herpetofaunal Sampling
The herpetofaunal community was sampled using drift fences with box funnel traps. Each stand contained four drift fences made of 7.6-m lengths of aluminum flashing with funnel traps placed centrally on both sides of the fence. In addition, 19 L buckets were dug in flush with the ground surface at each end of the fence. Each pitfall had drainage holes to minimize mortality (Cantrell and others 2013).

Trapping was conducted from mid-May until the end of September in 2011 and 2012. Traps were open continuously except for 1 week in August. Captured animals were measured, identified, and released. Captured herpetofauna were marked using toe clips. Clip corresponded to treatment type and year captured. All animals were released a few meters away from their capture site.

Small Mammal Sampling
The small mammal community was surveyed using Sherman live traps (7.7 x 9.0 x 23.3 cm) (Cantrell 2010). In addition, mammals found in the drift fence arrays were recorded. Sherman live trapping was conducted from June and through August with each stand sampled twice. Each sampled stand had 60 Sherman live traps placed 10 m apart within a 50- by 90-m grid. All traps were baited with peanut butter and re-baited each day. To avoid possible sample time bias, four stands were sampled concurrently for each trapping period, one stand of each treatment type (control, SW, OSW), as well as one additional stand of one of the treatment types. All traps were opened continuously for five nights and checked each morning. After the five-night trapping period, the traps were moved to the next set of stands. After all stands have been sampled once, the process was repeated resulting in 10-trap nights of sampling per stand. While the stands that were sampled concurrently remained the same, the order in which the groups of stands were chosen for sampling was randomized during both sampling periods to minimize time bias. Captured mammals were measured, identified, and released. All animals were released a few meters away from their capture site. Captured mammals were marked using toe clips. Clip corresponded to treatment type and year captured.

Carbon Sampling
In the summer of 2012, field samples were collected to determine the amount of dried mass and to model the carbon stored within trees (d.b.h. > 1 cm.); woody vegetation (d.b.h. ≤ 1cm at 1 to 30 cm above ground level); herbaceous vegetation; coarse woody debris [small end diameter > 7.62 cm (Woodall and Williams 2005)]; fine woody debris (small end diameter ≤ 7.62 cm); and vines.

Vines, woody vegetation, and herbaceous vegetation were harvested from 1- by 1-m quadrats by clipping at ground level at 10 m from the center of each drift fence at two randomly generated azimuths. If the quadrat fell within a permanent vegetation sample plot or directly on a large standing tree then the next azimuth was used. The quadrat was constructed using 3.8-cm-diameter PVC pipe and was measured from the inside edge. Sampling at the four traps resulted in eight total samples per stand. All vegetation with d.b.h. > 1 cm within the plot was excluded. Collected material was bagged and oven dried at 105 ºC until a consistent weight was achieved before being weighed to determine biomass (Davies and others 2011). Biomass was converted to carbon weight by multiplying by 0.5 (Namayanga 2002).

Coarse and fine woody debris were sampled in the same 1- by 1-m quadrats at 10 m from the center of each drift fence at two randomly generated azimuths. Samples were collected, brought into the laboratory, dried, and weighed. Materials too large to weigh (logs) had their volume calculated using measurement of diameter at both ends as well as length of the log. Using the average diameter, volume was calculated assuming log shape as a cylinder. A section of the log was collected using a handsaw. This section was brought back to the lab for analysis. Wet volume was determined by wrapping the section in plastic and submerging in water to test displacement. The section was dried and weighed to determine density which was used to estimate total mass of the log (Scott and others 2004).

Statistical Analysis

General linear model (GLM) analysis of variance (ANOVA) for a completely randomized split plot design (CRSPD) was used to test the treatment difference and interaction on study species. Species with < five captures in each year were excluded from the analysis to avoid the effect of small sample bias. Post-hoc Tukey multiple range test (HSD) was used to identify differences between specific treatments if ANOVA tests were significant. Statistical tests were declared significant when $p < 0.1$ (SPSS.v20.0).

RESULTS AND DISCUSSION

Data were collected for the 2011 and 2012 field seasons. Data presented here were preliminary results. Sampling will continue in 2013. Data for midstory trees have not yet been analyzed.

Total aboveground carbon storage (not including midstory) for control and OSW were similar (table 1). SW treatments had approximately half the stored carbon compared to control and OSW. Overstory trees contained the majority of the carbon within all treatment types and accounted for most of differences in total carbon between treatment types. Herbaceous biomass/carbon among treatment types showed that control had the lowest amount while OSW and SW had similar carbon content within the herbaceous layer. Control stands had the lowest amount of carbon within woody vegetation while SW had the highest amount. Control had the highest amount of carbon stored in CWD while OSW had the lowest.

Control and OSW stands both had nearly twice the amount of carbon stored above ground (excluding midstory) compared to SW stands. This mirrors the differences seen in carbon stored within the overstory trees, with control and OSW having more than twice the amount of stored carbon in this component compared to SW. The most rapid change in the carbon pool has been found in the above-ground tree biomass (Fahey and others 2010). These initial assessments are subject to change, as others have found that carbon storage rates are higher in disturbed stands compared to untreated stands (Horner and others 2010). Because the SW treatment removed 60 to 70 percent of the overstory trees while OSW retained the overstory, we expect that the amount of carbon stored will remain higher in the OSW compared to the SW, although the rate of storage may not follow that same pattern. Increased light and growing space created by tree removal created conditions within the SW stands that favored growth of herbaceous and small woody plants.

Table 1--Carbon sampling results showing amounts of carbon stored in 2012 within stands treated with three different forest management practices at Burrow Cove in Grundy County, TN

Treatment	Classes[a]	Mean	Total
		------*Mg/ha carbon*------	
Control	CWD	11.62	114.0911
	FWD	4.58	
	HERB	0.01	
	LL	3.84	
	VINE	0.40	
	WOODY	0.08	
	Overstory	93.56	
Oak shelterwood	CWD	3.71	115.8632
	FWD	5.99	
	HERB	0.17	
	LL	4.87	
	VINE	1.40	
	WOODY	0.15	
	Overstory	99.57	
Shelterwood	CWD	6.29	58.41753
	FWD	5.84	
	HERB	0.14	
	LL	3.53	
	VINE	0.12	
	WOODY	0.31	
	Overstory	42.18	

[a]Classes described are: coarse woody debris (CWD), fine woody debris (FWD), herbaceous vegetation (HERB), leaf litter (LL), vines (VINE), woody vegetation (WOODY), and overstory trees (overstory).

Table 2--Means ± standard deviations of herpetofaunal response to three different forest management practices at Burrow Cove in Grundy County, TN, 2011. ANOVA (F) test was followed with post-hoc Tukey tests. Different letters in columns indicate significant difference (Tukey p < 0.1)

Species	Scientific name	Control	SW	OSW	F	P
Eastern five-lined skink	*Plestiodon fasciatus*	1.00±1.65a	4.33±1.50b	2.0±1.58a	7.928	0.004
Eastern fence lizard	*Sceloporus undulatus*	0.56±0.52a	6.50±4.41b	1.60±2.30a	6.703	0.007
Broadhead skink	*Plestiodon laticeps*	1.22±0.67a	5.33±2.50b	3.20±3.27ab	9.411	0.002

Table 3--Means ± standard deviations of herpetofaunal and small mammal response to three different forest management practices at Burrow Cove in Grundy County, TN, 2012. ANOVA (F) test was followed with post-hoc Tukey tests. Different letters in columns indicate significant difference (Tukey p < 0.1)

Species	Scientific name	Control	SW	OSW	F	P
Eastern five-lined skink	*Plestiodon fasciatus*	0.71±0.76a	2.67±1.75b	1.25±1.50ab	3.491	0.059
Eastern fence lizard	*Sceloporus undulatus*	0.14±0.38a	3.67±2.88b	1.00±1.15a	6.354	0.011
Masked shrew	*Sorex cinereus*	0.43±0.54a	1.67±1.03b	0.75±0.96ab	3.675	0.052

In the 2011 field season we captured a total of 2,469 herpetofauna individuals encompassing 26 species (13 amphibian and 13 reptile). We also had 347 mammal captures comprised of 15 species. During the 2012 field season we captured a total of 1,170 individuals. Reptile captures yielded 12 different species and 111 individuals. Amphibian captures yielded 15 different species and 906 individuals. Mammal captures yielded 10 species and 153 captures. In both years American toad (*Anaxyrus americanus*) was the most abundant species overall. The most abundant mammal species was the white footed mouse (*Peromyscus leucopus*) in both years. The most abundant reptile in 2011 was the broadhead skink (*Plestiodon laticeps*). In 2012 the most abundant reptile was the eastern fence lizard (*Sceloporus undulatus*). Data for both years combined showed the most abundant reptile was the eastern fence lizard.

In 2011, eastern five-lined skink (*Plestiodon fasciatus*), broadhead skink, and eastern fence lizard all showed greater abundance in SW stands (table 2). In 2012, eastern five-lined skink, eastern fence lizard, and masked shrew (*Sorex cinereus*) all showed greater abundance in SW stands (table 3).

The most commonly captured species was the American toad. This is a common and ubiquitous species in Tennessee and the large numbers can likely be attributed to breeding pools created by road ruts in the logging road that the treatment stands all border (Cantrell and others 2013). American toad showed no response to treatment, but the numbers did fluctuate. In 2011 there were 1,316 captures of this species while in 2012 there were only 610 captures. This difference may be due to a prolonged drought during breeding season as well as logging activity that destroyed many road rut pools.

There were four total lizard species captured, but one species, Ground Skink (*Scincus lateralis*), had too few captures (< 3 captures per year) to warrant analysis. Three species, five-lined skink, broadhead skink, and eastern fence lizard, showed response to treatment. All of these lizard species were more abundant in SW stands compared to OSW and control stands. This is due to the removal of canopy cover and subsequent changes in the understory environment. The opening of the forest canopy increased the amount of light that reached the forest floor, creating basking sites for thermoregulation. Slash piles and CWD created habitat as well. This is supported by findings by Greenberg (2001) and Cantrell and others (2013) who also found an overall increase in reptilian species richness and abundance in response to removal of canopy.

Abundance changed for only one mammal species in response to the treatments. Masked shrew showed higher abundance in SW stands compared to control, but there was no difference in abundance between SW and OSW. This could be due to the increased light associated with the removal of canopy allowing herbaceous and small woody plants to increase. A flush of herbaceous and woody growth created habitat for the shrews as well as habitat for insects on which the shrew feeds. The carbon data show that both the SW and OSW treatments created conditions which increased the herbaceous and woody plants.

Preliminary results show that the SW treatments created conditions favorable to certain lizard species, as they showed consistently greater abundance in SW stands compared to the other treated stands. Masked shrew also responded with higher abundance in the SW compared to other treatments in 2012.

These findings give forest resource managers and private land owners in the region better understanding of how herpetofauna and small mammals respond to these three forest management decisions 2- to 4- years post-harvest. Results suggest that the active management treatments do not adversely affect reptiles or amphibian populations and may provide some benefit to a few species. Conversely, the control, or the decision to 'do nothing' may not be an optimal choice for some shrew and lizard species.

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LITERATURE CITED

Braun, E.L. 1950. Deciduous forests of eastern North America. Caldwell, NJ: The Blackburn Press. 596 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: 158-163.

Cantrell, A. 2010. Herpetofaunal and small mammal response to oak regeneration treatments on the Mid-Cumberland Plateau of southern Tennessee. Normal, AL: Alabama A&M University.139 p. M.S. thesis.

Cantrell, A.; Wang, Y.; Schweitzer, C.; Greenberg, C. 2013. Short-term response of herpetofaunal to oak regeneration treatments on the Mid-Cumberland Plateau of southern Tennessee. Forest Ecology and Management. 295: 239-247.

Cromer, R.B.; Gresham, C.A.; Goddard, M. [and others]. 2007. Associations between two bottomland hardwood forest shrew species and hurricane-generated woody debris. Southeastern Naturalist. 6(2): 235-246.

D'Amato, A.W.; Bradford, J.B.; Fraver, S.; Palik, B.J. 2011. Forest management for mitigation and adaption to climate change: insights from long-term silviculture experiments. Forest Ecology and Management. 262: 803-816.

Davies, Z.G., Edmondson, J.L.; Heinemeyer, A. [and others]. 2011. Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city wide scale. Journal of Applied Ecology. 48(5): 1125–1134.

De Vries, B.J.; van Vuuren, D.P.; Hoogwijk, M.M. 2007. Renewable energy sources: their global potential for the first-half of the 21st century at a global level: an integrated approach. Energy Policy. 35(4): 2590-2610.

Fahey, T.J.; Woodbury, P.B.; Battles, J.J. [and others]. 2010. Forest carbon storage: ecology, management, and policy. Frontiers in Ecology and the Environment. 8(5): 245-252.

Ferris, R.; Humphrey, J.W. 1999. A review of potential biodiversity indicators for application in British forests. Forestry. 72(4): 313-328.

Field, C.B.; Campbell, J.E.; Lobell, D.B. 2008. Biomass energy: the scale of the potential resource. Trends in Ecology & Evolution. 23(2): 65-72.

Gan, J.; Smith, C.T. 2006. Availability of logging residues and potential for electricity production and carbon displacement in the USA. Biomass and Bioenergy. 30(12): 1011-1020.

Greenberg, C.H. 2001. Response of reptile and amphibian communities to canopy gaps created by wind disturbance in the southern Appalachians. Forest Ecology and Management. 148(1): 135-144.

Hansen, A.J.; Spies, T.A.; Swanson, F.J.; Ohmann, J.L. 1991. Conserving biodiversity in managed forests. BioScience. 41(6): 382-392.

Hoover, C.; Stout, S. 2007. The carbon consequences of thinning techniques: stand structure makes a difference. Journal of Forestry. 105: 266-270.

Horner, G.J.; Baker, P.J.; Mac Nally, R. [and others]. 2010. Forest structure, habitat and carbon benefits from thinning floodplain forests: managing early stand density makes a difference. Forest Ecology and Management. 259: 286-293.

Lanham, J.D.; Guynn D.C., Jr. 1996. Influences of coarse woody debris on birds in southern forests. Gen. Tech Rep. SE-94. In: McMinn, J.W.; Crossley, D.A., Jr., eds. Proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 101-107.

Loftis, D.L. 1990. A shelterwood method for regenerating red oak in the southern Appalachians. Forest Science. 36(4): 917-929.

Namayanga, L.N. 2002. Estimating terrestrial carbon sequestered in above ground woody biomass from remotely sensed data. Enschede, The Netherlands: International Institute for Geo-information Science and Earth Observation: 10-12.

NRCS [Natural Resources Conservation Service]. 2007. Soil Survey Geographic (SSURGO) database for Grundy County, Tennessee. http://soildatamart.nrcs.usda.gov. [Date accessed: December 19, 2012].

Nordén, B.; Götmark, F.; Tönnberg, M.; Ryberg, M. 2004. Dead wood in semi-natural temperate broadleaved woodland: contribution of coarse and fine dead wood, attached dead wood and stumps. Forest Ecology and Management. 194(1): 235-248.

Scott, N.A.; Rodrigues, C.A.; Hughes, H. [and others]. 2004. Changes in carbon storage and net carbon exchange one year after an initial shelterwood harvest at Howland Forest, ME. Environmental Management. 33: 9-22.

Smalley, G.W. 1982. Classification and evaluation of forest sites on the Mid-Cumberland Plateau. Gen. Tech. Rep. SO-38. New Orleans: U.S. Department of Agriculture Forest Service, Southern Research Experiment Station. 58 p.

Spetich, M.A.; Graney, D.L. 2003. Effect of preharvest understory treatment and group opening size on four-year survival of advance reproduction in the Boston Mountains of Arkansas. In: Van Sambeek, J.W.; Dawson, J.O.; Ponder, F., Jr. [and others], eds. Proceedings of the 13th central hardwood forest conference. Gen. Tech. Rep. NC-234. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Research Station: 259-263.

Tilman, D.J.; Hill, J.; Lehman, C. 2006. Carbon-negative biofuels from low-input high diversity grassland biomass. Science. 314(5805): 1598-1600.

Verschuyl, J.; Riffell, S.; Miller, D.; Wigley, T.B. 2011. Biodiversity response to intensive biomass production from forest thinning in North American forests–a meta-analysis. Forest Ecology and Management. 261(2): 221-232.

Wang, Y.; Lesak, A.A.; Felix, Z.; Schweitzer C.J. 2006. Initial response of an avian community to silvicultural treatments in the southern Cumberland Plateau, Alabama, USA. Integrative Zoology. 3: 126-129.

Winandy, J.E.; Rudie, A.W.; Williams, R.S.; Wegner. T.H. 2008. Integrated biomass technologies: a future vision for optimally using wood and biomass. Forest Products Journal. 58(6): 6-16.

Woodall, C.W.; Williams. M.S. 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, Northern Research Station. 47 p.

NON-TIMBER FOREST PRODUCTS: RAMPS IN THE WAYNESVILLE, NC WATERSHED

Kristina Connor, Jim Chamberlain III, Hilliard Gibbs Jr., and Matt Winn[1]

Abstract--The potential of forest farming was noted as far back as 1929, but the recognition of its importance dates back only 20 to 30 years. The U.S. market for harvested foods and medicinal plants from forests now exceeds $4 billion annually. Ramps (*Allium tricoccum* Aiton), or wild leeks, grow in patches in the rich moist forests of the eastern United States. They are harvested during the spring, and their use is becoming more popular. This increased harvesting pressure is forcing national forests and local municipalities to consider the long-term sustainability of plant populations. In 2010, a study was initiated in the Waynesville, NC watershed to examine the reproductive biology of ramps and to assess the field survival of their seeds. Flower stalk survival and seed production of individual ramp plants were tracked, length of time ramp seeds remain viable in the seed bank was determined, and the basic germination requirements of seeds were studied. We found that seeds will not germinate without some form of stratification and that no seeds remained ungerminated in the sample buried for 20 months in the field.

INTRODUCTION

We have little information about the flowering cycle and seed development for some understory plant species. Also lacking are data on how long seeds of certain understory species survive naturally in the field or how long they can be stored at low temperature in the laboratory. This information is not only critical for assessing potential of a species to thrive in the field but also can be used to evaluate its storage potential in artificial germplasm reserves. Ramps (*Allium tricoccum* Aiton; fig. 1) grow in patches in the rich deciduous forests of eastern North America from Canada south to Tennessee and North Carolina (Davis and Greenfield 2002). A member of the Liliaceae family, the plants have broad, smooth leaves that appear in March or April and send up a shoot of white flowers in summer. Ramp plants also reproduce vegetatively from underground rhizomes or when a bulb is large enough to split. The plants were the first ‘greens’ of spring for early settlers in the eastern United States and are now harvested during annual spring festivals which are growing in popularity, as evidenced at http://www.richwooders.com/ramp/ramps.htm [Date accessed: April 17, 2013]. Typically, both the leaves and bulbs are harvested as well as any attached underground rhizomes (Rock and others 2004).

Figure 1--Ramp leaves emerging (early March).

Ramp flower stalks appear after leaves have senesced. Each stalk has a head of small (approximately 6-mm long) flowers, each of which contains one ovary. The ovary matures into a three-celled seed capsule; each cell can produce one seed, thus there is a maximum production of three seeds per flower. The seeds, which mature in late August or early fall, are shiny black spheres approximately 3 to 3.5 mm in diameter (fig. 2). Because harvesting occurs before flower stalks are readily apparent and

[1]Research Plant Physiologist, Southern Research Station, USDA Forest Service, Auburn, AL 36849; Research Forest Products Technician, Southern Research Station, USDA Forest Service, Blacksburg, VA 24060; Physical Scientist, Southern Research Station, USDA Forest Service, Auburn, AL 36849; and Forestry Technician, USDA Forest Service, Southern Research Station, Blacksburg, VA 24060.

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long before seeds mature, genetic diversity in wild populations may be devastated if harvesters fail to nurture individual patches by allowing some mature plants capable of flowering to remain *in situ* and by leaving rhizomes undamaged [Davis and Greenfield 2002; Edgar and others (no date); Rock and others 2004].

Figure 2--Ramp mature seed head (September).

OBJECTIVES

Our study objectives were to: (1) determine flower stalk survival and seed production of individual ramp plants; (2) observe ramp seed longevity and viability in the seed bank; and (3) examine the basic germination requirements (stratification procedures, light requirements) of ramp seeds.

MATERIALS AND METHODS

The Study Site

The field study was installed and seed heads harvested on the 8,400-acre Waynesville Watershed, located in Haywood County, southwest of Waynesville, NC [35 29' 19" N, 82 59' 20" W (fig. 3)]. The 50-acre man-made reservoir and surrounding watershed are classified by the state of North Carolina as WS-1, the state's most stringent classification, under which development is forbidden. The watershed is one of the largest tracts of undeveloped, non-federal forest land remaining in western North Carolina. Detailed information on soils and forest cover can be found at http://www.townofwaynesville.org/content/view/374/347/ [Date accessed: July 14, 2010].

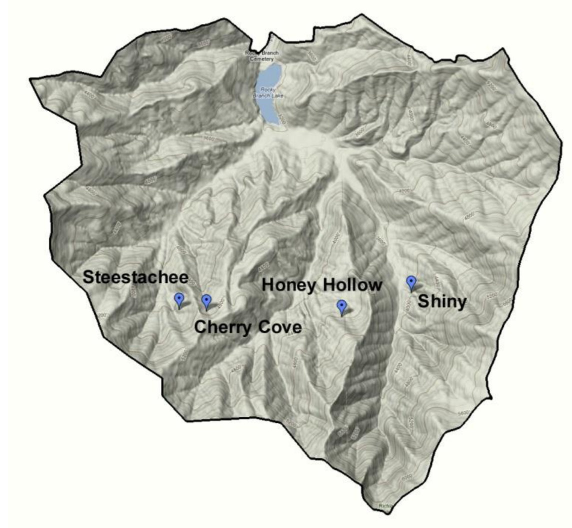

Figure 3--Map of the Waynesville Watershed and study site locations.

Flower Stalk Survival and Seed Production

Fifty individual ramp plants with flower stalks forming were flagged in May 2010 on each of four sites in the watershed: Honey Hollow (HH), Shiny Gulch (SH), Cherry Cove (CC), and Steestachee (ST). Two months after flagging, an average of 37.5 percent of the flower stalks had aborted or died. Flags were moved to plants with fully formed flower stalks to maintain data collection on 50 plants per site. The number of flowers per stalk, the number of seeds developing per stalk, the number of dominant seeds forming, and the number of mature seeds produced by each stalk were counted and recorded. A dominant seed is defined as one which, at the time of observation, is visually larger than other seeds formed on the same flower (fig. 4).

Seed Bank Study

Ramp seeds for the seed bank study and laboratory study were collected from plants transplanted from western North Carolina (Waynesville Watershed; Franklin, NC; and Robbinsville, NC) to raised beds in Blacksburg, VA (37˚ 16' 29.63" N, 80˚ 25' 15.40" W, elevation 2,024 feet). Seeds were harvested in 2009 and cold-stored prior to field study establishment.

Figure 4--Example of dominant seed (arrow).

Screen mesh bags measuring 7.6- by 10.2-cm (3- by 4-inches) and sewn with plastic thread were made to enclose the seeds for the seed bank study. Each bag contained 100 seeds. Four bags were harvested at each sampling time throughout the year. Bags were buried 1.27-cm (0.5-inch) deep and covered with mineral soil in September 2010. A wire mesh 'sandwich' enclosed each set of 4 bags (fig. 5). Sites were marked with numbered flags to facilitate recovery, with 4 bags anchored to each flag. The flag to harvest on each collection date was randomly determined at the start of the experiment. Seeds that were harvested but had not germinated in the field bags were used to determine the laboratory germination requirements of ramp seeds.

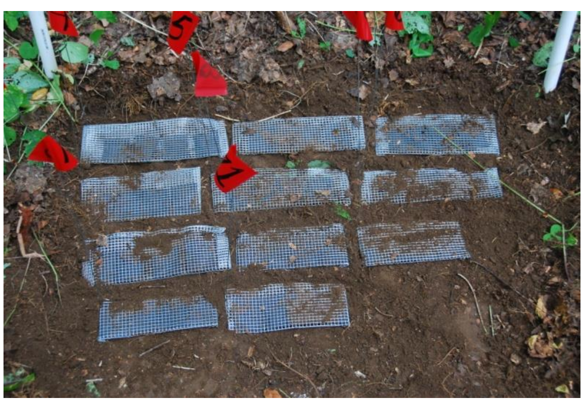

Figure 5--Bags enclosed in a wire mesh 'sandwich' and buried in contact with mineral soil. Bags are attached to flags to facilitate retrieval.

Laboratory Study

Treatments--In initial germinations tests, we found that placing boxes of ramp seeds straight into a Precision® low-temperature illuminated incubation chamber set at a uniform temperature of 30 ˚C without prior cold stratification resulted in no germination. In addition, cold stratifying seeds for 12 weeks prior to placing them in the germination chamber at a uniform 30 ˚C resulted in no germination. Cold stratification was accomplished by rolling the seeds in a moist paper towel, sealing the towel in a plastic bag, and storing in the cold room at 5 ˚C. We abandoned the uniform temperature approach and reset the Precision cabinet to register 25 ˚C for 16 hours with no light and 30 ˚C for 8 hours with light.

Four replications of 100 seeds each were placed in clear plastic boxes layered with moist sand. In addition to the lid, the top of the box was sealed with a layer of clear plastic wrap. The boxes were put in the low-temperature illuminated incubation chamber using the fluctuating temperature regime (fig. 6). Seeds remained in the germinator for 12 weeks. They were then cold stratified for 12 weeks by placing the boxes in the cold room at 5 ˚C. After stratification, the boxes were put back in the germinator for an additional 12 weeks.

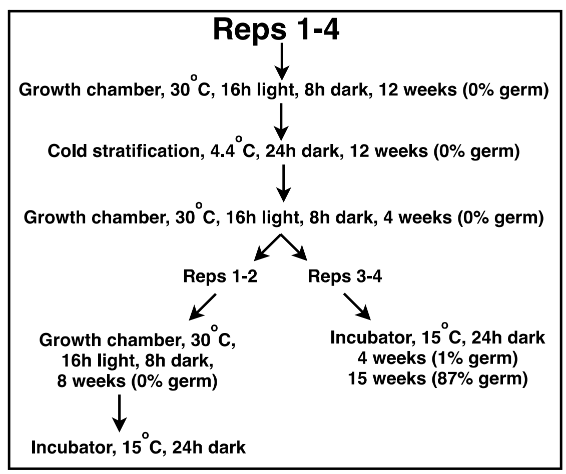

Figure 6--Ramp seed laboratory germination flow diagram.

Although it was not part of our original study plan, we acquired a non-illuminated, refrigerated Revco® incubator well after the study was well

Table 1--Ramp flower and seed production information for four sites within the Waynesville Watershed, NC

Site Code	Average flower stalk length	Average flowers per stalk	Total seed sites (T)[a]	Total dominant seeds (D)[b]	Total seeds collected (C)	C/T	C/D
	mm	*number*				*percent*	
ST	317.4	11.2	1,594	355	337	21.1	94.9
CC	290.7	10.6	1,584	574	343	21.7	59.8
SH	256.9	12.5	1,762	691	453	39.2	65.6
HH	279.5	17.9	2,680	1,036	857	32.0	82.7

[a] There are typically three seeds (a cluster) formed in each flower. Some of these seed sites, however, were necrotic or unformed and were not included in the total.

[b] Dominant seeds were those in each seed cluster that were developing normally and that could reasonably be expected to form mature, viable seeds. They were visibly larger than other seeds formed from the same flower (fig. 4). These seeds were counted in August and collected in October.

underway and utilized it to expand our experiment (fig. 6). Replications 1 and 2 remained in the illuminated incubation chamber for an additional 8 weeks. Replications 3 and 4 were placed in the non-illuminated incubator, set at 15 ˚C, for 12 weeks.

RESULTS AND DISCUSSION

Flower Stalk Survival and Seed Production

Seeds were only collected once during the fall, and some may have shed before collection. Because of this, it is said with qualifications that an average of 10 seeds were produced per flower stalk (table 1). Like all members of the Liliaceae, each ramp flower can produce three seeds. Of the total potential seed sites on each stalk, 74 percent did not form or were aborted, resulting in an average seed yield of 26 percent. We also counted the number of dominant seeds (fig. 4) that were being formed in each ramp flower in August 2010 and found that an average of 75 percent of these seeds survived to maturity when collected in October 2010, again with the qualifier that only one collection was made during the fall. Insect predation was apparent on 14.3 percent of collected seeds (fig. 7). Eighty percent of these damaged seeds were found at the HH site.

Seed Bank Study

Ramp seeds did not germinate in the field until the collection in August 2011, 11 months after the bags were buried. Then, only roots had emerged. We did not observe shoot emergence in the field until May 2012, 20 months after study initiation (fig. 8). We found no viable, ungerminated seeds in that collection or in the July 2012 collection. From these data, we hypothesize that ramp seeds will not remain in the seed bank longer than 2 years. If mature plants are removed from a site and rhizomes harvested or damaged, it is doubtful that populations will recover.

Figure 7--Seed pods damaged by insect activity prior to harvest.

Figure 8--Ramp seeds with both root and shoot emergence in bags collected from the field (May 2012).

Laboratory Study

Laboratory seed germination proved difficult. Alternating light and dark regimes at uniform temperature in the chamber proved ineffective, as did a cold stratification treatment, in accelerating germination.

It wasn't until we acquired an incubator and put seeds into it for 24 hours without light and at a cooler temperature (15 ˚C) that roots emerged. Determining if it was the absence of light or the cooler temperature that affected root emergence will be the subject of another study, when more seeds can be acquired from the study sites. Shoot emergence in the laboratory tests was not achieved.

ACKNOWLEDGMENTS

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LITERATURE CITED

Davis, J.M.; Greenfield, J. 2002. Cultivating ramps: wild leeks of Appalachia. In: Janick, J.; Whipkey, A., eds. Trends in new crops and new uses. Alexandria, VA: ASHS Press: 449-452. http://www.hort.purdue.edu/newcrop/ncnu02/v5-449.html [Date accessed: July 14, 2010].

Edgar, B.; Brubaker, H.; Tuminelli, K. [n.d.]. Plugging the leak on wild leeks: the threat of over-harvesting wild leek populations in northern New York. 54 p. http://www.stlawu.edu/sites/default/files/resource/wild_leek_conservation.pdf [Date accessed: June 6, 2013].

Rock, J.; Beckage, B.; Gross, L.J. 2004. Population recovery following differential harvesting of *Allium tricoccum* Ait. in the southern Appalachians. Biological Conservation. 116: 227-235.

SURVIVAL AND GROWTH OF RESTORED PIEDMONT RIPARIAN FORESTS AS AFFECTED BY SITE PREPARATION, PLANTING STOCK, AND PLANTING AIDS

Chelsea M. Curtis, W. Michael Aust, John R. Seiler, and Brian D. Strahm[1]

Abstract--Forest mitigation sites may have poor survival and growth of planted trees due to poor drainage, compacted soils, and lack of microtopography. The effects of five replications of five forestry mechanical site preparation techniques (Flat, Rip, Bed, Pit, and Mound), four regeneration sources (Direct seed, Bare root, Tubelings, and Gallon), and three planting aids (None, Mat, Tubes) on American sycamore (*Platanus occidentalis* L.) and willow oak (*Quercus phellos* L.) were examined for 2 years following establishment in order to evaluate the treatment potential for enhancing survival and growth. After 2 years, Mounding and Gallon seedlings were found to be the most beneficial treatments for American sycamore survival and growth. Bedding also proved beneficial. For willow oak, Mound and Bed were also beneficial, particularly with Bare root seedlings Gallons. The positive responses of the species to mounding and bedding were due to treatment effects on elevation on poorly drained sites, reduction of competition, and reduction of compaction.

INTRODUCTION

The Federal Water Pollution Control Act of 1972 and subsequent amendments and interpretations have resulted in policies which require wetland restoration or creation to offset wetland losses caused by activities such as urbanization (Stolt and others 2000). Wetland creation projects have a relatively poor track record for success, thus it is common to have wetland mitigation ratios of 2:1 or 3:1 (Brown and Lant 1999). The relatively poor success rates are caused by a variety of problems, including: poor recognition of site conditions which results in poor species selection; sites with compacted soil conditions, which inhibit soil water movement and root penetration; excessively wet sites that may kill or suppress growth of desired tree species; and lack of topography which may limit the survival and growth of planted tree species (Bailey and others 2007).

Forest managers have been facing similar regeneration problems on such sites. Harvested sites are commonly compacted and poorly drained, yet silviculturalists have overcome these limitations with a variety of mechanical site preparation techniques (Aust and others 1998). For example, both mounding and bedding have been widely used across the eastern United States since the 1950s and 1960s to overcome lack of relief and soil compaction on wet sites (Lof and others 2012, Miwa and others 2004). Similarly, riparian restoration efforts often have site limitations that are overcome by using alternative planting sources or planting aids. Interestingly, there has been little technical transfer between forest managers and the wetland restoration community.

The literature indicates that wetland restoration efforts could be enhanced with increased use of silvicultural tools. Therefore, the objective of this research project is to quantify the effects of mechanical site preparation, regeneration source, and planting aids on the survival and growth of two species commonly used on mitigation sites: the early successional species American sycamore (*Platanus occidentalis* L.) and the later successional species willow oak (*Quercus phellos* L.).

MATERIALS AND METHODS

Study Site

The study site is located in the Piedmont physiographic province on the Virginia Tech R.J. Reynolds Homestead Forest Research Extension Center near Critz, VA. Much of this 280-ha area was converted to tobacco plantations during the 1800s, and the specific riparian area was subjected to agriculture and excessive compaction by a recent soil compaction research project. Thus the area has compacted soils, lack of relief, and is excessively wet during winter and spring. Soil series in the study include Augusta (fine-loamy, mixed semiactive, thermic Aeric Endoaquults), French (fine loamy over sandy, mixed, active

[1]Graduate Research Assistant, Professor, Professor, and Assistant Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061.

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mesic Fluvaquentic Dystrudepts), and Roanoke (fine, mixed, semiactive thermic Typic Endoaquults). The site is located in the floodplain of a first-order perennial stream with a 500-ha watershed. Flooding to a depth of approximately 25 cm occurs during most spring seasons.

Treatments

Five site preparation treatments were established (Flat, Rip, Bed, Pit, Mound). The Flat treatment consisted of surface tillage with a disk harrow to reduce herbaceous competition. The Rip treatment consisted of subsoiling with a 30-cm ripping shank underneath the planting zones. The Bed treatments were made with the blade on a bulldozer. The Pit and Mound treatments were created in the same area: a tractor-mounted backhoe was used to excavate pit material (approximately 40 cm) to create an adjacent mound of approximate 40 cm. Four regeneration sources (Seed, Bare root, Tubeling, Gallon) were superimposed across all site preparation treatments. For the Seed treatment, three acorns or a finger pinch of American sycamore seeds were planted. The seeds were collected from piedmont seed sources approximately 3 months prior to planting. The bare-root seedlings were purchased from commercial nurseries and planted with dibble bars. Tubeling and Gallon containers were purchased from a commercial nursery and planted with spades. Three levels of planting aids were applied to all combinations of site preparation and regeneration sources (None, Tubes, and Mats). Tubes consisted of 1-m planting tubes, and mats were 50- by 50-cm geotextile fabric. Seeds and seedlings were planted in May 2011, and planting aids were installed in June 2012. Minimal herbaceous control was conducted during summer 2011 and 2012.

Survival and Growth Parameters

Survival and growth indices were measured after one and two growing seasons (2011, 2012) simply recording if the individual had survived. Growth parameters included average ground line diameter based on caliper measurements from two directions and total tree height measured to the nearest 1 cm with a height poles. For trees taller than 1.3 m, diameter at breast height (d.b.h., cm) was also recorded. The diameter and height measures were subsequently converted to a biomass index in cm^3 based on d^2h geometry.

Statistical Design and Analysis

The study is arranged as a split-split plot within a Randomized Complete Block Design. Five blocks were established for each of the two species. The main effects are five site preparation treatments (Flat, Rip, Bed, Pit, and Mound). The Pit and Mound treatments were established together but were analyzed as two treatments. The split plot was comprised of the four regeneration sources. The second split consisted of the three planting aids. For each experimental unit, four units (seed or seedling) were established. Thus, for both American sycamore and willow oak , approximately 1,200 trees or seeds were planted (5 blocks x 5 site preparation treatments x 4 regeneration sources x 3 planting aids x 4 trees per seeds = 1,200). Survival and growth parameters were analyzed via analysis of variance (ANOVA) and statistically different means were separated with a Tukeys HSD test.

RESULTS AND DISCUSSION

American sycamore survival was consistently between 62 and 66 percent across the site preparation treatments during years 1 and 2 with the exception of the Pit treatment, which had significantly lower survival (table 1). American sycamore's biomass index was significantly reduced by the Pit treatment during both years and was significantly increased by the Mound site preparation treatment (table 1). Not unexpectedly, American sycamore survival percentages were favored by the Tubeling and Gallon regeneration sources and were very low for direct seeding (table 2). Biomass values followed the same general trends: lowest for Direct seeding, followed by Bare root and Tubeling, and greatest in the Gallon regeneration sources (table 2). Examination of the effects of planting aids on American sycamore survival and growth indicated that the

Table 1--Effects of site preparation on survival and biomass indices for American sycamore on a Piedmont forest restoration site during years 1 and 2. Mean values within a column followed by a different letter are significantly different at α ≤ 0.10

Site preparation treatment	--------Survival--------		---Biomass index---	
	Year 1	Year 2	Year 1	Year 2
	--------*percent*---------		-----------*cm*^{3}------------	
Flat	68b	68b	496b	1487b
Rip	68b	68b	549b	1900b
Bed	66b	66b	649b	2497b
Pit	62a	59a	333a	1232a
Mound	68b	68b	913c	3811c
p	≤0.0001	≤0.0561	≤0.0001	≤0.0001

Table 2--Effects of regeneration sources on survival and biomass indices for American sycamore on a piedmont forest restoration site during years 1 and 2. Mean values within a column followed by a different letter are significantly different at α ≤ 0.10

Regeneration source treatment	--------Survival-------		---Biomass index---	
	Year 1	Year 2	Year 1	Year 2
	--------*percent*--------		----------*cm*^{3}------------	
Direct seed	36a	16a	10a	10a
Bare root	72b	58b	196a	933b
Tubling	85c	85c	445b	1896c
Gallon	88c	88c	1023c	3728d
p	≤0.0001	≤0.0001	≤0.0001	≤0.0001

planting Mats and Tubes slightly increase survival, and the Mat increased biomass (table 3).

Willow oak survival was also reduced by the Pit treatment (table 4). Mound treatment positively increased biomass for willow oak during year 1. During year 2, the pattern continued, but the differences were not significant. Bare root and Gallon both provided good survival for willow oak (table 5), and the Direct seeding survivals were lower. However, survival of the relatively larger-seeded willow oak was better with direct seeding than for American sycamore. During year 1, the gallon regeneration source offered the best biomass growth for willow oak, but by year 2, regeneration sources effects on willow oak biomass were not significant (table 5). The Mat planting aid provided both enhanced survival and biomass for willow oak (table 6).

The efficacy of any given treatment will be due to a combination of the treatment effects on both survival and growth. Therefore, we created a unitless performance index which is the product of survival and biomass for both American sycamore (table 7) and willow oak (table 8). We then examined the top 25 percent of all treatments to select the “best” treatments. For American sycamore, the Mound treatment followed by the Bed site preparation treatment combined with large Gallon containerized seedlings clearly performed the best. There was no clear pattern with planting aids for American sycamore. For willow oak the Mound and Bed treatments worked well for the Gallon containers, but the Bare root seedlings also performed well. For both species, the Mound and Bed treatments elevated the seedlings in the poorly drained soils and favored tree survival. The Mound and Bed treatments also alleviated soil compaction. The Mound treatment

Table 3--Effects of planting aids on survival, diameters, heights, and biomass indices for American sycamore on a piedmont forest restoration site during years 1 and 2. Mean values within a column followed by a different letter are significantly different at α ≤ 0.10. Year 1 biomass indices were not significantly different

Planting aid treatment	-------Survival------- Year 1	Year 2	--Biomass index-- Year 1	Year 2
	------*percent*--------		----------*cm*^{3}----------	
None	71a	63a	600	2230ab
Tube	76b	66ab	533	1694a
Mat	77b	70b	649	2632b
p	≤0.006	≤0.0001	≤0.6370	≤0.0144

Table 4--Effects of site preparation on survival and biomass indices for willow oak on a piedmont forest restoration site during years 1 and 2. Mean values within a column followed by a different letter are significantly different at α ≤ 0.10. Year 2 biomass indices were not significantly different

Site preparation treatment	-------Survival------- Year 1	Year 2	---Biomass index--- Year 1	Year 2
	--------*percent*-------		--------*cm*^{3}------------	
Flat	77b	56b	53a	2193
Rip	79b	60b	67b	1471
Bed	80b	80b	69b	1435
Pit	67a	67a	70b	440
Mound	83b	83b	84c	2001
p	≤0.0001	≤0.0001	≤0.0001	≤0.1507

Table 5--Effects of regeneration sources on survival and biomass indices for willow oak on a piedmont forest restoration site during years 1 and 2. Mean values within a column followed by a different letter are significantly different at α ≤ 0.10. Year 2 biomass indices were not significantly different

Regeneration source treatment	--------Survival------- Year 1	Year 2	---Biomass index--- Year 1	Year 2
	----------*percent*----------		------------*cm*^{3}------------	
Direct seed	61a	31a	0.5a	1240
Bare root	86c	86c	65c	2278
Tubling	74b	65b	5b	1435
Gallon	84c	84c	127d	1104
p	≤0.0001	≤0.0001	≤0.0001	≤0.2038

Table 6--Effects of planting aids on survival, diameters, heights, and biomass indices for willow oak on a piedmont forest restoration site during years 1 and 2. Mean values within a column followed by a different letter are significantly different at α ≤ 0.10. Year 1 biomass indices were not significantly different

Planting aid treatment	-------Survival------- Year 1	Year 2	--Biomass index-- Year 1	Year 2
	------*percent*-------		---------*cm*^{3}----------	
None	74a	51a	67	718a
Tube	78ab	60ab	67	1620b
Mat	82b	69b	67	2154c
p	≤0.0001	≤0.0001	≤0.3323	≤0.0011

Table 7--American sycamore performance index (biomass x survival) at 2 years. Numbers with asterisk represent the top 25 percent of all treatment combinations for performance

Regeneration source	Planting aid	-----------------Site preparation treatment------------- Flat	Rip	Bed	Pit	Mound
Direct seed	None	1	156	17	1	337
	Tube	1	438	112	76	76
	Mat	2	3	25	7	11
Bare root	None	557	550	670	138	2023*
	Tube	426	530	770	257	1370
	Mat	402	451	250	92	852
Tubling	None	645	1523	2238*	382	2234*
	Tube	721	831	1084	443	874
	Mat	893	1616	1799	875	3119*
Gallon	None	2193*	2208*	1923*	1803	3113*
	Tube	1684	2038*	2735*	1393	3456*
	Mat	1592	1905*	3532*	2042*	6234*

Table 8--Willow oak performance index (biomass x survival) at 2 years. Numbers with asterisk represent the top 25 percent of all treatment combinations for performance

Regeneration source	Planting aid	-----------------Site preparation treatment------------- Flat	Rip	Bed	Pit	Mound
Direct seed	None	111	4	4	1	5
	Tube	20	86	10	1	31
	Mat	1	10	15	1	39
Bare root	None	398	748	1541*	145	1624*
	Tube	2173*	516	1015*	76	1025*
	Mat	669	674	787	237	1424*
Tubling	None	52	127	33	1	153
	Tube	101	108	38	45	118
	Mat	116	53	13	8	390
Gallon	None	727	1067*	987*	287	1480*
	Tube	676	985*	1157*	518	1354*
	Mat	888	972*	1168*	446	1201*

provided some competition control by burying the seeds of competitors deeper than they could survive. Currently, the use of large seedlings is common practice on mitigation sites, but these data indicate that Mound and Bed also offer significant potential for improving wetland mitigation. The wetland mitigation community typically has close working ties with equipment contractors, thus locating excavator operators should be relatively straightforward. The use of Mounds offers significant potential to overcome the typical problems encountered on Piedmont mitigation sites.

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LITERATURE CITED
Aust, W.M.; Burger, J.A.; McKee, W.H., Jr. [and others]. 1998. Bedding and fertilization ameliorate effects of designated wet-weather skid trails after four years for loblolly pine (*Pinus taeda*) plantations. Southern Journal of Applied Forestry. 22(4): 222-226.

Bailey, D.E.; Perry, J.E.; Daniels, W.L. 2007. Vegetation dynamics in response to organic matter loading rates in a created freshwater wetland in southeastern Virginia. Wetlands. 27(4): 936-950.

Brown, P.H.; Lant, C.L. 1999. The effect of wetland mitigation banking on the achievement of no-net-loss. Environmental Management. 23(3): 333-345.

Lof, M.; Dey, D.C.; Navarro, R.M.; Jacobs, D.F. 2012. Mechanical site preparation for forest restoration. New Forests. 48: 825-848.

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: 137-151.

Stolt, M.H.; Genthner, M.H.; Daniels, W.L. [and others]. 2000. Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. Wetlands. 20(4): 671-683.

VEGETATIVE COMPOSITION IN FORESTED AREAS FOLLOWING APPLICATION OF DESIRED FOREST CONDITION TREATMENTS

Trent A. Danley, Andrew W. Ezell, Emily B. Schultz, and John D. Hodges[1]

Abstract--Desired forest conditions, or DFCs, are recently created parameters which strive to create diverse stands of hardwoods of various species and age classes, along with varying densities and canopy gaps, through the use of uneven-aged silvicultural methods and repeated stand entries. Little research has been conducted to examine residual stand composition and hardwood regeneration after DFC installment. The objectives of this study were to characterize forest overstory and midstory conditions after DFC treatments and to assess natural regeneration. Residual stand conditions after application of DFC harvest guidelines indicate that shade-tolerant species will be future site occupants, and oaks will diminish or disappear over time. This documented initial forest response to DFC treatments can be used by forest and wildlife habitat managers when assessing the potential outcomes of DFC management.

INTRODUCTION

Managers of southern bottomland hardwood forests have recently been introduced to a new forest management approach. Desired forest conditions, or DFCs, are parameters which strive to create diverse stands of hardwoods of various species and age classes, along with varying densities and canopy gaps, through the use of uneven-aged silvicultural methods and repeated stand entries. DFCs are frequently promoted as enhancing habitat conditions for many threatened neotropical migratory bird populations and other wildlife species (LMVJV 2011). However, little research has been conducted to examine residual stand composition and hardwood regeneration after DFC application. From a silvicultural standpoint, it is very important to characterize regeneration to determine what long term effects DFCs could have on hardwood forest composition.

Proponents of DFCs claim that this management practice will not shift forest composition toward shade-tolerant species (LMVJV 2011). Although oak or other commercially desirable trees are significant timber species on many southern bottomland hardwood sites and simultaneously provide valuable food and cover resources to many wildlife species, regeneration of these species is not a top priority of DFC management. Regardless of the objectives given by the creators of DFCs, it is responsible stewardship to consider the outcome of any land management practice prior to application. Since DFC parameters have been created in the last decade, it is imperative to begin to characterize and document early forest response in order to understand long-term forest impacts. These DFCs have recently been heavily promoted for use on nonindustrial private bottomland hardwood forest land (LMVJV 2011). It is the purpose of this study to provide critical information to land managers and other decision makers regarding the application of these practices.

The first objective of this study was to characterize residual forest conditions after DFC application, including species composition and basal area within the overstory and midstory. The second objective was to evaluate natural regeneration by species composition and density.

MATERIALS AND METHODS

Site Description

The study site was comprised of two adjacent tracts within a private landholding owned by Catfish Point Land and Timber Company LLC. It is located in Bolivar County, MS, (latitude 33.68° N, longitude 91.16° W), adjacent to the Mississippi River. Catfish Point is approximately 15 miles northwest of Greenville, MS. The Catfish Point Air Strip stand was approximately 84 acres in size, and the Catfish Point Main Road stand was 52 acres. DFC treatments were applied to the Air Strip stand in February 2010 and to the Main Road stand in February 2011. These soils were of alluvial origin and included very fine sandy loams, silt loams, and silty clay loams. These stands were behind the Mississippi River levee, and occasionally flood in late spring. The overstory was comprised of sugarberry (*Celtis laevigata* Willd.), boxelder

[1]Graduate Research Assistant, Professor, and Professor, respectively, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; and Professor Emeritus, Ashland, MS, 39576.

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(*Acer negundo* L.), pecan [*Carya illinoinensis* (Wangenh.) K. Koch], and sycamore (*Platanus occidentalis* L.). The midstory is dominated by sugarberry, boxelder, and American elm (*Ulmus americana* L.), and the understory is also primarily shade-tolerant regeneration.

Sampling Design

A nested-plot sampling method was used; therefore three concentric plots at a sample point shared a common plot center. The total number of plots sampled was based on the acreage of each tract with a sampling intensity of one set of plots per 2 acres, resulting in 68 plots to sample 136 acres. Plots were spaced on a systematic 4- by 5-chain line-plot grid. This inventory was conducted from June 25 to July 5, 2012 and represented a 10 percent overstory cruise.

According to DFC specifications, the intended goals for basal area and stand density were to be attained through the use of uneven-aged silviculture. Therefore, site treatments were applied with alternation of two variable retention cutting regimes, light-cut and heavy-cut, along with untreated control areas throughout the tract. Areas harvested using single-tree selection were classified as light-cut treatment areas, and areas harvested using group selection were called heavy-cut treatment areas. Attempts were made to equally distribute nested plots throughout all three treatment types. This was performed by offsetting plots perpendicular to the direction of travel, if necessary, to ensure that each entire set of plots was centered within the opening of either a light-cut or heavy-cut treatment. Plot centers were recorded by GPS and physically demarcated using a 36-inch pin flag.

The overstory was defined as stems within a dominant, co-dominant, or intermediate crown class and assessed using 0.2-acre (52.7-foot radius) plots. Species and diameter at breast height (d.b.h.) in 0.1-inch classes were recorded for each overstory tree in the plot. Midstory trees were defined as stems between 10-feet tall and the base of the general canopy and measured in 0.025-acre (18.6-foot radius) plots. Midstory information included species and d.b.h. in 0.1-inch classes for each tree in the plot. Understory species were defined as woody regeneration < 10-feet tall and sampled within 0.01-acre (11.7-foot radius) plots (Stanturf and Meadows 1994). Within the understory plots, all species were identified and assigned into height classes. The height classes were <1 foot, 1 to 2 feet, 2 to 3 feet, 3 to 6 feet, and 6 to 10 feet.

RESULTS AND DISCUSSION

Catfish Point - Air Strip Stand

Overstory--Average overstory basal area in 17 heavy-treatment plots within the Catfish Point Air Strip stand was 65.92 square feet per acre, with 25.59 trees per acre (table 1). Residual species composition (table 2) following the heavy treatment was comprised of sugarberry, sycamore, American elm, boxelder, hickory, pecan, and sweetgum (*Liquidambar styraciflua* L.) with sugarberry being the predominant species. Average overstory basal area among 12 light-treatment plots was 105.04 square feet per acre, with an average of 44.17 trees per acre. This treatment area had a residual overstory made up of 57.5 percent sugarberry. There were also American elm, boxelder, green ash (*Fraxinus pennsylvanica* Marsh.), hickory (*Carya* spp.), pecan, sweetgum, and sycamore throughout the stand. Based on 13 control plots, mean basal area was 93.85 square feet per acre with 48.46 trees per acre. Overstory of the control area was 67.5 percent sugarberry. Miscellaneous other species comprising minor portions of the stand were American elm, boxelder, cedar elm (*U. crassifolia* Nutt.), hickory, pecan, sweetgum, and sycamore.

Though this stand is now privately owned by Catfish Point Land and Timber Company, LLC, it was formerly owned and managed by Chicago Mill and Lumber Company. This particular stand was previously “high graded” by selection cutting for decades. This method of harvesting repeatedly removed the most commercially desirable species from the stand and failed to create conditions favorable for the regeneration of oaks. Thus, some portions of the stand naturally became devoid of desirable species.

According to Walker and Watterson (1972), less desirable species such as sugarberry and boxelder retained during high-grading timber harvests will continue to grow and regenerate to the detriment of other more desirable species.

Table 1--Average basal area and trees per acre in treatment areas at Catfish Point

	Basal area ($feet^2/acre$)				Trees per acre			
	Air Strip		Main Road		Air Strip		Main Road	
Treatment	Overstory	Midstory	Overstory	Midstory	Overstory	Midstory	Overstory	Midstory
Control	93.85	21.19	75.46	20.52	48.46	129.23	46.25	215.00
Heavy	65.92	10.94	44.41	3.96	25.59	32.94	21.36	50.91
Light	105.04	13.86	58.67	11.15	44.17	103.33	29.29	108.57

Table 2--Average overstory species percent composition in treatment areas at Catfish Point

	Air Strip			Main Road		
Species	Control	Heavy	Light	Control	Heavy	Light
	percent					
American elm	11.1	11.5	8.5	2.7	8.2	4.9
Boxelder	4.8	2.3	3.8	4.1	--	--
Cedar elm	0.8	--	--	--	--	--
Green ash	--	--	1.9	1.4	--	--
Hickory	0.8	1.1	0.9	--	4.1	--
Pecan	3.2	8.0	5.7	8.1	18.1	24.4
Persimmon	--	--	--	5.4	--	--
Sugarberry	67.5	52.9	57.7	67.6	67.3	61.0
Sweetgum	4.8	9.2	15.1	2.6	--	--
Sycamore	7.1	14.9	6.6	8.1	--	9.7
Water oak	--	--	--	--	2.0	--

Based on the species composition in table 2, sugarberry was the primary component of both treatment areas and the control. Control areas had the highest level of sugarberry, which indicates that sugarberry was selected against when the stand was marked for DFC installment. In this case, the removal of sugarberry would also be helpful to promote the growth and regeneration of other more desirable trees both for wildlife and commercial timber interests. In fact, there were higher percentages of pecan, sweetgum, and sycamore in the heavy- and light-treatment areas compared to the control. Though these species are not the most beneficial trees to wildlife or the most valuable commercial species for timber production, they are an improvement over American elm, boxelder, and sugarberry.

Midstory--Average midstory basal area in 17 heavy-treatment plots was 10.94 square feet per acre with a mean of 32.94 trees per acre (table 1). Flowering dogwood was the largest component of the midstory in heavy-treatment areas and comprised 41.7 percent of this layer. Boxelder comprised 29.2 percent of the layer, and sweetgum made up 12.5 percent. Other less-abundant heavy-treatment area midstory species included American elm, pecan, and sugarberry (table 3). Average midstory basal area in 12 light-treatment plots was 13.86 square feet per acre with 103.33 trees per acre. Species composition of the light-treatment midstory plots varied from the heavy treatment. In these areas, sugarberry made up exactly 50 percent of the midstory species. Flowering dogwood (*Cornus florida* L.) was the second most common species with 20.6 percent. Other species present in these treatment areas were American elm, boxelder, and sycamore. Based on the 13 midstory plots in control areas, mean basal area was 21.19 square feet per acre with 129.23 trees per acre. The most common midstory species was sugarberry, which made up 70.5 percent of the layer. American elm, boxelder, flowering dogwood, and pecan were

Table 3--Average midstory species percent composition in treatment areas at Catfish Point

Species	Air Strip Control	Air Strip Heavy	Air Strip Light	Main Road Control	Main Road Heavy	Main Road Light
	percent					
American elm	9.1	4.2	5.9	6.8	4.8	4.8
Boxelder	13.6	29.2	17.6	45.5	--	47.6
Deciduous holly	--	--	--	--	4.8	--
Flowering dogwood	4.5	41.7	20.6	2.3	71.4	23.8
Pecan	2.3	4.2	--	--	--	--
Persimmon	--	--	--	2.3	--	--
Sugarberry	70.5	8.3	50.0	43.2	14.3	23.8
Swamp chestnut oak	--	--	--	--	4.8	--
Sweetgum	--	12.5	--	--	--	--
Sycamore	--	--	5.9	--	--	--

other species that comprised the remainder of the control area midstory. The midstory of the control and light-treatment areas was composed heavily of sugarberry. This is logical since the overstory in these areas is primarily sugarberry, and it is a shade-tolerant species (Putnam 1951). Based on the midstory species composition presented in table 3, the future overstory stand will likely be heavily comprised of sugarberry as well, followed by boxelder, American elm, and pecan.

Regeneration--Abundance of regeneration in the heavy-treatment areas is presented in table 4. Sugarberry had the highest abundance with 3.06 stems per plot in the smallest height class. In comparison, green ash had the greatest abundance in the 1- to 2- and 2- to 3-foot class with 3.94 and 4.35 stems per plot, respectively. However, green ash was among the least frequently occurring species across all plots (table 7). This indicates that green ash regeneration was likely very abundant in the few plots where it was observed. Sugarberry was by far the most prevalent species in approximately 82 percent of the plots, followed by boxelder (76.47 percent) and American elm (64.71 percent). Other species in low abundance and comprising < 12 percent each of the regeneration strata were deciduous holly (*Ilex decidua* Walt.), flowering dogwood, hickory, honeylocust (*Gleditsia triacanthos* L.) , sycamore, sweetgum, and water oak (*Quercus nigra* L.). Although 0.06 stems per plot of water oak were presented in table 4, the raw data indicates that this number is representative of one water oak seedling in one height class found on one plot out of 17 sample plots in heavy-treatment areas.

The most abundant regeneration species in the light-treatment areas was sugarberry (table 4). There was an average of more than six stems of sugarberry per plot in the smallest height class, with 0.92 stems in the 1- to 2-foot category and even fewer stems in the three remaining categories. Sugarberry regeneration was present on all light-treatment plots. Boxelder was also present on 58 percent of plots at an average rate of 2.00 stems per plot under 1 foot and fewer stems in all other height classes. American elm grew on exactly 50 percent of the plots, and its greatest abundance was 1.42 stems per plot in the smallest height class. There were minor amounts of cedar elm, flowering dogwood, hickory, and sycamore present in the regeneration strata as well.

Boxelder was the most abundant species in the control areas (table 4). It occurred on 53.85 percent of the plots at a rate of 2.46 stems per plot in the smallest height class. American elm and sugarberry were found in 30.77 percent of sample plots. American elm was more abundant

Table 4--Average regeneration species abundance by treatment and by height class at the Catfish Point Air Strip stand

	Heavy					Light					Control				
	Height class in feet														
Species	<1	1-2	2-3	3-6	6-10	<1	1-2	2-3	3-6	6-10	<1	1-2	2-3	3-6	6-10
	stems per plot														
American elm	1.76	2.94	2.65	0.65	--	1.42	0.5	0.25	--	--	0.08	0.15	0.23	--	--
Boxelder	0.71	1.41	1.47	0.94	0.24	2	0.5	0.83	0.25	0.08	2.46	0.62	0.23	--	0.08
Cedar elm	--	--	--	--	--	--	0.08	--	--	--	--	--	--	0.08	--
Deciduous holly	--	--	0.06	--	--	--	--	--	--	--	--	--	--	--	--
Flowering dogwood	0.12	--	--	--	0.06	0.67	0.17	0.08	--	--	--	--	--	--	--
Green ash	0.53	3.94	4.35	0.06	--	--	--	--	--	--	--	0.08	--	--	--
Hickory	--	0.06	--	0.06	--	0.08	--	--	--	--	--	--	--	--	--
Honeylocust	0.06	--	0.06	--	0.12	--	--	--	--	--	--	--	--	--	--
Pecan	--	--	--	--	--	--	--	--	--	--	--	--	0.08	--	--
Sugarberry	3.06	3.12	1.06	0.94	0.06	6.42	0.92	0.17	0.42	0.17	0.31	0.15	--	--	--
Sweetgum	--	--	0.06	0.35	0.18	--	--	--	--	--	--	--	--	--	--
Sycamore	--	--	0.06	0.88	--	--	--	--	0.08	--	--	--	--	--	--
Water Oak	--	0.06	--	--	--	--	--	--	--	--	--	--	--	--	--
Willow oak	--	--	--	--	--	--	--	--	--	--	0.31	--	--	--	--

in the 2- to 3- and 1- to 2-foot categories, yet sugarberry was most abundant in the smallest height class. There were also minor amounts of cedar elm, green ash, pecan, and willow oak (*Q. phellos* L.) regeneration scattered throughout the site. Willow oak was present on 15.38 percent of control plots in the < 1 foot height class but in comparatively low abundance (0.31 stems per plot). Raw data show that two willow oak seedlings were found on 2 plots out of 13 sample plots. These seedlings likely germinated from the previous year's acorn crop but will have little chance for survival in the low light conditions found in control areas of this stand.

Species composition of the regeneration strata at the Catfish Point Air Strip Stand is presented in table 6. Heavy-treatment areas were composed of 27.71 percent green ash, 25.69 percent sugarberry, and 24.95 percent American elm. Other less-common species were boxelder, deciduous holly, flowering dogwood, hickory, honeylocust, sweetgum, sycamore, and water oak. Light-treatment areas contained 53.59 percent sugarberry, 24.31 percent boxelder, and 14.36 percent American elm. Remaining species of minor occurrence were cedar elm, flowering dogwood, hickory, and sycamore. In contrast to the previous treatment areas, the control was dominated by 69.84 percent boxelder in the regeneration strata, followed by only 9.52 percent of both American elm and sugarberry. Cedar elm, green ash, pecan, and willow oak were the remaining species of regeneration found throughout the control. As previously mentioned, this relatively minor abundance of water oak and willow oak will not likely mature into advanced regeneration due a lack of available light resulting from future canopy closure and midstory development.

Catfish Point - Main Road Stand

Overstory--Average overstory basal area in 11 heavy-treatment plots was 44.41 square feet per acre with 21.36 trees per acre (table 1). Sugarberry was the dominant species which comprised 67.3 percent of the overstory (table 2). Pecan (18.4 percent) and American elm (8.2 percent) were also present throughout the stand. The two remaining species which comprised the overstory in heavy-treatment areas were hickory and water oak. Average overstory basal area in seven light-treatment plots was 58.67 square feet per acre with 29.29 trees per acre (table 1). Similar to the heavy-treatment areas, sugarberry comprised the vast majority (61.0 percent) of the light-treatment area overstory. Pecan was the second most common species in the stand but only made up 24.4 percent of the overstory species. American elm (4.9 percent) and sycamore (9.7 percent) were the other components of these areas (table 2). Average overstory among eight control plots was characterized by a basal area of 75.46 square feet per acre and a mean of 46.25 trees per acre. Sugarberry comprised 67.6 percent of the overstory species. All other species comprised < 10 percent each of the stand. These species included American elm, boxelder, green ash, pecan, persimmon (*Diospyros virginiana* L.), sweetgum, and sycamore.

Basal area and number of overstory trees per acre were slightly lower in the Main Road stand. Upon visual examination of the Main Road stand, the overstory was obviously much less dense than the overstory of the Air Strip stand. Overstory species composition in the Catfish Point Main Road stand was very similar to the species composition of the Catfish Point Air Strip stand. Although these were separate stands, they were close in proximity and were subjected to the same harvesting practices of the past.

Sugarberry comprised over 60 percent of the stand across all treatment and control areas. As shown in table 2, heavy-treatment areas contained almost the exact same percentage of sugarberry as the control. Even though heavy-treatment areas were subjected to more harvesting, they contained slightly more sugarberry than light-treatment areas. In this stand, pecan appears to have been the preferred species for retention. All other species were relatively low in abundance across all treatment and control areas. Sycamore comprised 8 percent of the control overstory and 9 percent of the light-treatment areas, but was not found in heavy-treatment areas.

Midstory--Average midstory basal area in 11 heavy-treatment plots was 3.96 square feet per acre with a mean of 50.91 trees per acre (table 1). Flowering dogwood was the predominant species in these areas, comprising 71.4 percent of the midstory (table 3). Other minor species included sugarberry (14.3 percent), American elm (4.8 percent), deciduous holly (4.8 percent), and swamp chestnut oak (*Q. michauxii* Nutt.; 4.8 percent). Average basal area in seven light-treatment midstory plots was 11.15 square feet per acre with 108.57 trees per acre. In contrast with the midstory in heavy-treatment areas, midstory of the light-treatment areas was dominated by 47.6 percent boxelder. Flowering dogwood and sugarberry each composed 23.8 percent of the midstory, and the remaining midstory stems were American elm (4.8 percent). Average midstory basal area in eight control plots was 20.52 square feet per acre with 215 trees per acre. Boxelder made up 45.5 percent of this stand, and sugarberry comprised 43.2 percent. Remaining species were American elm (6.8 percent), flowering dogwood (2.3 percent), and persimmon (2.3 percent) (table 3).

Sugarberry and boxelder were the most prevalent midstory species in the Main Road stand. Heavy-treatment areas had the lowest proportion of sugarberry, and light-treatment and control areas had successively more stems. Control and light-treatment areas had high levels of boxelder, yet the heavy-treatment areas had no boxelder within the sample plots. In contrast, the heavy-treatment areas had a high level (71.4 percent) of flowering dogwood, whereas the light-treatment areas had 23 percent and the control areas had < 3 percent. The relatively low basal area (3.96 square feet per acre) and number of trees per acre (50.91) support the fact that flowering dogwood and the minor abundance of other species in this midstory were of small diameter and sporadic occurrence.

It is interesting to note that there was a small (4.8 percent) component of swamp chestnut oak in the midstory which equated to one stem sampled in one plot. Oaks typically do not grow well with the competition and low light levels found in the shaded midstory typical of bottomland hardwood forests. With this fact in mind, the importance of midstory control must not be underestimated prior to harvest activities to stimulate the germination and growth of oak seedlings into advanced regeneration before complete overstory removal. Lowery and others (1998) observed an increase in the growth and survival of oak seedlings in midstory control plots in a minor Mississippi bottomland. Their study determined that midstory injection during the early dormant season had the best control and ultimately promoted the greatest level of available light for oak seedlings.

Regeneration--Sugarberry was very common in approximately 90 percent of sample plots in heavy-treatment areas. Stems < 1-foot tall occurred at an average rate of 3.73 stems per plot, whereas stems between 1- and 2-feet and 2- and 3-feet tall were found at 1.91 and 1.27 stems per plot, respectively (table 5). There

Table 5--Average regeneration species abundance by treatment and by height class areas at the Catfish Point Main Road stand

	Heavy					Light					Control				
	Height classes in feet														
Species	<1	1-2	2-3	3-6	6-10	<1	1-2	2-3	3-6	6-10	<1	1-2	2-3	3-6	6-10
	stems per plot														
American elm	--	0.09	0.18	0.36	--	0.29	0.14	1.14	0.29	0.14	--	0.25	0.13	0.13	--
Boxelder	--	0.18	--	--	--	0.29	--	0.14	0.43	--	2.25	0.38	--	--	--
Cedar elm	--	--	--	--	--	0.14	--	--	--	--	--	--	--	--	--
Chinese privet	--	--	--	--	--	--	--	--	--	--	0.13	--	--	--	--
Eastern cottonwood	--	--	--	--	--	--	--	--	0.29	--	--	--	--	--	--
Flowering dogwood	0.90	--	--	0.36	0.18	--	--	--	0.14	--	--	--	--	--	--
Green ash	--	--	--	--	--	--	--	0.14	--	--	0.13	--	--	--	--
Hickory	--	0.18	--	--	--	0.14	--	--	--	--	--	0.13	--	--	--
Sugarberry	3.73	1.91	1.27	0.55	0.09	1.29	1.71	0.71	0.14	0.14	4.25	0.75	0.25	0.25	--
Swamp chestnut oak	--	--	--	--	--	--	--	-	--	--	--	--	0.25	--	--
Sweetgum	--	--	--	--	--	--	--	--	--	--	--	0.13	--	0.13	--
Sycamore	--	--	--	0.27	--	--	0.29	--	--	--	--	--	--	--	--
Water Oak	0.18	0.09	--	0.09	--	0.14	--	--	0.14	--	0.13	0.25	--	--	--
Willow oak	0.09	--	--	--	--	--	0.14	--	--	--	--	--	--	--	--

were also 0.55 sugarberry stems present per plot in the 3- to 6-foot height class. This heavy abundance of sugarberry regeneration across all height classes clearly expressed the ability of this species to flourish under low light conditions that prohibit the establishment of all but the most shade-tolerant species. This means a very low composition (2.0 percent) of water oak seed sources in the overstory of heavy-treatment areas (table 2), coupled with factors such as cyclical acorn crops and low levels of available light, have inhibited the establishment and growth of advanced oak regeneration in this stand. The combined total of five stems of oak regeneration in 9 percent of sample plots (table 5) confirms this hypothesis. When the height of these five stems is considered, only two stems were tall enough to be considered advanced regeneration. Unfortunately, survival of this advanced regeneration will be impeded by low light conditions.

Flowering dogwood was present on 27 percent of plots, and boxelder was found on 18 percent of plots. Flowering dogwood stems were present in the height class of < 1 foot, and in the 3- to 6- and 6- to 10-foot classes. Boxelder was only present in the 1- to 2-foot height class in 18.18 percent of sample plots. Other miscellaneous species including American elm, hickory, and sycamore between 1- and 6-feet tall were found on 9 percent of the plots. Though American elm and hickory regeneration are shade tolerant and likely developed under closed canopy conditions, the presence of sycamore indicates an area where substantial light is reaching the forest floor. It is unclear whether the sycamore originated from advanced regeneration or germinated following harvest, but it will likely suffer from the same decrease in light availability as the oak regeneration.

Sugarberry was the most common species in the regeneration strata across all treatment areas. It comprised 76.1 percent of the regeneration in heavy-treatment areas, 47.5 percent in light-treatment areas, and 55.7 percent in control areas (table 6). Other miscellaneous regeneration species in heavy-treatment areas included American elm, boxelder, flowering dogwood, hickory, sycamore, water oak, and willow oak. These species each comprised < 10 percent of the overall regeneration. Sugarberry was the primary species in light-treatment plots and comprised 47.5 percent of the regeneration. American elm made up 23.7 percent of regeneration and boxelder was 10.2 percent. Remaining species were cedar elm, eastern cottonwood, flowering dogwood, green ash, hickory, sycamore, water oak, and willow oak, and each comprised < 4 percent of the regeneration present. Approximately 55 percent of regeneration in the control areas was

Table 6--Species percent composition in the regeneration layer by treatment at Catfish Point

	Air Strip			Main Road		
Species	Heavy	Light	Control	Heavy	Light	Control
	percent					
American elm	24.95	14.36	9.52	6.4	23.7	5.1
Boxelder	14.86	24.31	69.84	1.8	10.2	26.6
Cedar elm	--	0.55	1.59	--	1.7	--
Chinese privet	--	--	--	--	--	1.3
Deciduous holly	0.18	--	--	--	--	--
Eastern cottonwood	--	--	--	--	3.4	--
Flowering dogwood	0.55	6.08	--	6.4	1.7	--
Green ash	27.71	--	1.59	--	1.7	1.3
Hickory	0.37	0.55	--	1.8	1.7	1.3
Honeylocust	0.73	--	--	--	--	--
Pecan	--	--	1.59	--	--	--
Sugarberry	25.69	53.59	9.52	76.1	47.5	55.7
Swamp chestnut oak	--	--	--	--	--	2.5
Sweetgum	1.83	--	--	--	--	2.5
Sycamore	2.94	0.55	--	2.8	3.4	--
Water Oak	0.18	--	--	3.7	3.4	3.8
Willow oak	--	--	6.35	0.9	1.7	--

sugarberry. Boxelder was the second most-common species and comprised 26.6 percent of the regeneration. Other species which occurred in control areas were American elm, green ash, hickory, Chinese privet (*Ligustrum sinense* Lour.), swamp chestnut oak, and sweetgum. Each of these species was < 6 percent of the overall composition.

Similar to the Catfish Point Air Strip stand, sugarberry was highly abundant across all treatment and control areas. American elm and boxelder also composed notable percentages of regeneration species across all areas. As previously stated, this regeneration is a strong indication of the primary overstory species of this stand prior to harvest. Though water oak, willow oak, and a few other somewhat desirable species were present, none of these species comprised over 4 percent of the total regeneration species in any treatment or control area.

In summary, regeneration across all treatment and control areas in this stand indicate that the future overstory will be comprised of sugarberry, American elm, boxelder, and other commercially undesirable species. The very low presence of oak regeneration and overall lack of environmental conditions conducive to the recruitment of future oak regeneration virtually ensure the perpetual loss of oaks and other commercially desirable shade-intolerant species in this stand.

CONCLUSION

This study was the first characterization of residual bottomland hardwood forest conditions of DFC applications in Mississippi. As such, all sampling methods were originally conceived and implemented to fully document the dynamic conditions across a forest following DFC application. It is important to note that this study simply provides an overview of conditions that were present in mid-summer of 2012. As of 2012, these findings are the sole documentation of early stages of DFC management in Mississippi.

Catfish Point DFC installation across 2 subsequent years was led and monitored by personnel experienced with marking bottomland hardwood timber to meet the forest parameters set forth by DFC objectives using uneven-aged management. Although the overstory of Catfish Point was heavily composed of less-desirable shade-tolerant species in all treatment areas, there was a notable decrease in sugarberry composition in the heavy treatment. This was

Table 7--Average percent occurrence of regeneration species by treatment at Catfish Point

	---------------Air Strip---------------			--------------Main Road---------------		
Species	Heavy	Light	Control	Heavy	Light	Control
	--*percent*--					
American elm	64.71	50	30.77	9.09	57.14	37.5
Boxelder	76.47	58.33	53.85	18.18	57.14	75
Cedar elm	--	8.33	7.69	--	14.29	--
Chinese privet	--	--	--	--	--	12.5
Deciduous holly	5.88	--	--	--	--	--
Eastern cottonwood	--	--	--	--	14.29	--
Flowering dogwood	11.76	16.67	--	27.27	14.29	--
Green ash	5.88	--	7.69	--	14.29	12.5
Hickory	11.76	8.33	--	9.09	14.29	12.5
Honeylocust	11.76	--	--	--	--	--
Pecan	--	--	7.69	--	--	--
Sugarberry	82.35	100	30.77	90.91	85.71	87.5
Swamp chestnut oak	--	--	--	--	--	12.5
Sweetgum	5.88	--	--	--	--	12.5
Sycamore	11.76	8.33	--	9.09	14.29	--
Water Oak	5.88	--	--	9.09	28.57	37.5
Willow oak	--	--	15.38	9.09	14.29	--

likely the intended result due to the high percent of sugarberry in the stand. A decrease in sugarberry could lead to an increase of other species, which could lead to greater species diversity.

Attaining species diversity is a goal of DFC management. Species diversity is undoubtedly beneficial to any naturally regenerated stand because it promotes stand development from interspecific competition, increased resilience to insects and disease, and many other ecological functions important to forest sustainability. However, the LMVJV (2007) currently suggests that species should not be a determining factor when marking timber prior to group and single tree selection harvests. Rather, the selection of harvest trees is based on the promotion of multi-strata canopy development, sustaining potentially low vigor trees for the recruitment of snags and coarse woody debris, and creating shrub-scrub habitat to support wildlife of special conservation concern.

These DFC objectives should be carefully weighed and considered against the goals of forest ownership by any manager considering DFC application. The initial forest response to DFC treatments is summarized below. It is also important to note that no observable differences in species composition were noted between the Catfish Point stands 1 and 2 years after harvest.

The Catfish Point Air Strip stand had a very high average basal area in light-treatment and control areas but a notably lower basal area in heavy-treatment areas. The Main Road stand at Catfish Point exhibited lower densities. Catfish Point had a high overall level of sugarberry, boxelder, American elm, and other species that are less desirable for commercial timber production and wildlife use. Both stands were characterized by a dense midstory of shade-tolerant species, primarily in control areas and light-treatment areas. Regeneration was variable in terms of abundance and species across both stands but was representative of the overstory on each site. The understory was primarily characterized by sugarberry and other shade-tolerant regeneration. Because DFC guidelines only allow for a maximum of 20 percent of the forest to be harvested using group selection, only a small area would be capable of developing advanced regeneration. It is therefore unlikely that shade-intolerant regeneration will develop into the overstory due to competition from other shade-tolerant species (Oliver and others 2005).

Future management activities at Catfish Point will depend on the goals of ownership. If desired, the shift of species composition of these stands into more desirable species would require harvesting the vast majority of the current stands. The oak component of these stands is too low to ensure natural regeneration for stocking. Though costly, some form of artificial regeneration is the only way to increase the oak component of these stands in the foreseeable future.

In summary, forest managers should note the high tree densities of these post-treatment areas when considering uneven-aged forest management proposed through DFC implementation when managing commercially desirable shade-intolerant species. Data in this study indicate that DFC treatments will continue to promote recruitment and growth of shade-tolerant species into the future overstory at both Catfish Point stands. Future forest inventories of these and other DFC areas will be crucial to the understanding of the implications of DFC management. The review and consideration of these results should be evaluated alongside DFC goals just like any other management scheme in order to fully realize and promote the wise, ethical, and sustained use of the bottomland hardwood forest wildlife and resources in Mississippi.

LITERATURE CITED

LMVJV [Lower Mississippi Valley Joint Venture] 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. Baton Rouge, LA: LSU Printing Services: 20-28.

LMVJV [Lower Mississippi Valley Joint Venture] Forest Resource Conservation Working Group. 2011. Wildlife forestry in bottomland hardwoods: desired forest conditions for wild turkey, white-tailed deer, and other wildlife. www.lmvjv.org/bookshelf.htm. [Date accessed: July 13, 2011].

Lowery, J.W.; Ezell, A.W.; Hodges, J.D. [and others]. 1998. Effects of imazapyr injection on competing understory and midstory stems in oak regeneration areas: first year results. Mississippi State, MS: Mississippi State University, Department of Forestry. 114 p.

Oliver, C.D.; Burkhardt, E.C.; Skojac, D.A. 2005. The increasing scarcity of red oaks in Mississippi River floodplain forests: influence of the residual overstory. Forest Ecology and Management. 210: 393-414.

Putnam, J.A. 1951. Management of bottomland hardwoods. Occas. Pap. 116. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 60 p.

Stanturf, J.A.; Meadows, J.S. 1994. Natural regeneration of southern bottomland hardwoods. In: Egan, A.F., ed. Proceedings of the Southern Regional Council on Forest Engineering. Mississippi State, MS: Office of Agricultural Communications, Division of Agriculture, Forestry, and Veterinary Medicine: 6-11

Walker, L.C; Watterston, K.G. 1972. Silviculture of southern bottomland hardwoods. Bull. 25. Nacogdoches, TX: Stephen F. Austin State University, School of Forestry. 78 p.

INTENSIVE LONGLEAF PINE MANAGEMENT FOR HURRICANE RECOVERY: FOURTH-YEAR RESULTS

David S. Dyson and Dale G. Brockway[1]

Abstract--The frequency and intensity of hurricanes affecting the United States has been projected to increase during coming decades, and this rising level of cyclonic storm activity is expected to substantially damage southeastern forests. Although hurricane damage to forests in this region is not new, recent emphasis on longleaf pine (*Pinus palustris* Mill.) restoration and the increasing number of longleaf pine plantations resulting from such efforts raise questions about both tropical storm effects on this species and suitable strategies and practices for facilitating its recovery from such storms. This study was established to evaluate different methods of quickly returning damaged stands to productive longleaf pine forests following Hurricane Ivan in 2004. After salvage operations cleared the study areas, three herbicides (hexazinone, imazapyr, triclopyr) versus an untreated control were tested for their effects on stand development using artificially regenerated longleaf pine. A fertilizer treatment was also applied on half of the plots. Four years following planting, developing trends show the possible benefits of chemical site preparation on longleaf pine seedling height and ground-line diameter, whereas fertilization has shown no significant effect.

INTRODUCTION

Hurricanes are a regular source of natural disturbance that cause both widespread and localized damage to forests on the Gulf and Atlantic Coastal Plains. Although these tropical cyclones may seem to occur irregularly during a human lifetime, if the activity from the mid-20th century is extended backwards through the Holocene, over 40,000 tropical storms are estimated to have affected the northern Gulf coast (Conner and others 1989). Furthermore, although likely due to multidecadal variability, Atlantic hurricane frequency and intensity have increased since 1995 (Pielke and others 2005); some models predict that this trend may continue because of global warming (Smith and others 2010). Thus, although southern ecosystems have developed in concert with these disturbance patterns, contemporary values and needs may not allow for the natural, unassisted rate of forest recovery or for potential increases in hurricane frequency and intensity.

Hurricanes represent a complex conundrum for southern forests, as they can both destroy and rejuvenate forests at the same time. Whereas cyclonic winds often greater than 100 miles per hour can damage or destroy hundreds or thousands of acres of forests, heavy rainfall - sometimes measured in feet - can break long-term droughts and provide immature forests the moisture needed to secure survival and establishment. Some data show that older stands are more severely damaged by hurricanes (Kush and Gilbert 2010), providing growing space for new cohorts that regenerate ecologically mature stands. By preventing succession to “climax” or steady-state conditions, hurricanes can actually increase ecosystem productivity and structural diversity (Conner and others 1989), which may benefit overall long-term ecosystem health.

In the early morning hours of September 16, 2004, Hurricane Ivan crossed Brewton, AL with winds as high as 120 miles per hour and dropped over 8 inches of rainfall. The eye-wall passed 15 miles west of the USDA Forest Service’s Escambia Experimental Forest (EEF), damaging the forest to the extent that almost 700,000 cubic feet of timber were salvaged from the property during the ensuing 6 months. Within the state of Alabama, an estimated $610 million worth of timber was damaged on 2.7 million acres.

This cooperative study with Cedar Creek Land and Timber Company of Brewton, AL was initiated in 2007 to identify the most effective approaches for restoring longleaf pine (*Pinus palustris* Mill.) on sites impacted by Hurricane Ivan. Ivan caused overstory losses exceeding 90 percent on six 22-acre units at the EEF that had been recently thinned to a basal area of 25 square feet per acre for regeneration with the shelterwood method. Because of extensive damage and the disruption caused by salvage operations, artificial regeneration remained the only viable option for achieving restoration goals. The resulting restoration project provided

[1]Forestry Technician, USDA Forest Service, Southern Research Station, Brewton, AL 36426; and Research Ecologist, USDA Forest Service, Southern Research Station, Auburn, AL 36849.

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an opportunity to evaluate various approaches for quickly reestablishing the longleaf pine forest and possibly decreasing rotation length for even-aged stands through intensive management practices. Given the predictions for increased frequency and intensity of disturbance in forests of the southern Coastal Plain, the objective of this operational-scale experiment is to test the effects of chemical site preparation and fertilization on reestablishment of productive longleaf pine forests.

METHODS

Study Site

This project is located at the EEF (31°N, 87°W) in Escambia County, AL. The EEF is a 3,000-acre tract owned by T.R. Miller Mill Company of Brewton, AL, that has been managed by the Forest Service since 1947 for longleaf pine management research. Timber on the property ranges from newly regenerated fourth-growth to second-growth stands (120+ years old). The EEF is a mesic upland site with rolling topography and elevations ranging between 85 and 285 feet above sea level. Soils on the study areas are all Ultisols, principally the Troup and Wagram associations. The EEF is located in the Alabama Area longleaf pine site zone (Craul and others 2005). Average rainfall is over 60 inches per year, evenly distributed throughout the year but with monthly minima in April and October. Temperatures are mild, with mean annual high and low temperatures of 79 °F and 53 °F, respectively, resulting in a growing season that typically extends from the end of March through October. The EEF contains a native bluestem (*Andropogon* and *Schizachyrium* spp.) understory with a wide variety of grasses and forbs that has been maintained for decades with dormant-season prescribed fire on a 3-year interval. This burning pattern has resulted in the development of an extensive shrub layer of clonal hardwood species and gallberry [*Ilex glabra* (L.) A. Gray] 2- to 3-feet high.

Experimental Design and Sampling

Six 22-acre stands that had been extensively damaged by Hurricane Ivan in 2004 were clearcut for artificial regeneration in 2007. Twenty-four (492 by 492 feet) experimental plots were then installed (four 5.5-acre plots per stand) in a randomized complete block design of eight treatments replicated three times. Treatments include fertilized and unfertilized combinations of: (1) control, in addition to (2) hexazinone as Velpar ULW, 33 pounds per acre [2.5 pounds of active ingredient (a.i.) per acre] applied March 2008; (3) imazapyr as Chopper EC, 48 ounces per acre (0.75 pounds a.i. per acre) plus 5 ounces methylated seed oil, applied June 2008; and (4) triclopyr as Garlon XRT, 4 quarts per acre (6 pounds. a.i. per acre) applied June 2008. During August 2008, all 24 plots were burned by prescription. Plots were hand-planted by a professional crew in February 2009 at 889 trees per acre (7- by 7-foot spacing) with container-grown longleaf pine seedlings obtained from Simmons Tree Farm in Kite, GA. An initial fertilizer treatment was applied to half of the plots in March 2009 and consisted of phosphorus (P as superphosphate, 120 pounds per acre) and potassium (K as muriate of potash, 70 pounds per acre). Measurement plots consist of n = 49 seedlings arranged in seven rows. Seedling heights and ground-line diameters (GLD) were measured annually, and the number of competing pine and hardwood stems within a 3.28-foot (1 m) radius of study seedlings was also tallied. All study plots were burned by prescription during winter of 2011, prior to the second-year data collection.

Data Analysis

Statistical analyses were conducted using NCSS version 7.1.1 software (Hintze 2007). For all tests, statistical significance was determined at $\alpha = 0.05$. Repeated measures analysis of variance (ANOVA) and Tukey-Kramer tests were used to determine differences among treatments and years for the following response variables: mean seedling survival, height, GLD, and number of competing stems.

RESULTS AND DISCUSSION

Seedling Survival

Four years after plantation establishment, there were no significant treatment differences in seedling survival. However, survival rates for years 2 through 4 (2010-2012) ranged between 60 and 85 percent and are significantly lower than survival following the first growing season, which was over 90 percent ($F = 77.32$; $p < 0.0001$) (fig. 1). Although seedlings were not graded or measured at time of planting, evaluating mean seedling diameters after one growing season and later growth rates suggest that average seedlings in all treatments exceeded the current interim guidelines for container seedlings (Dumroese and others

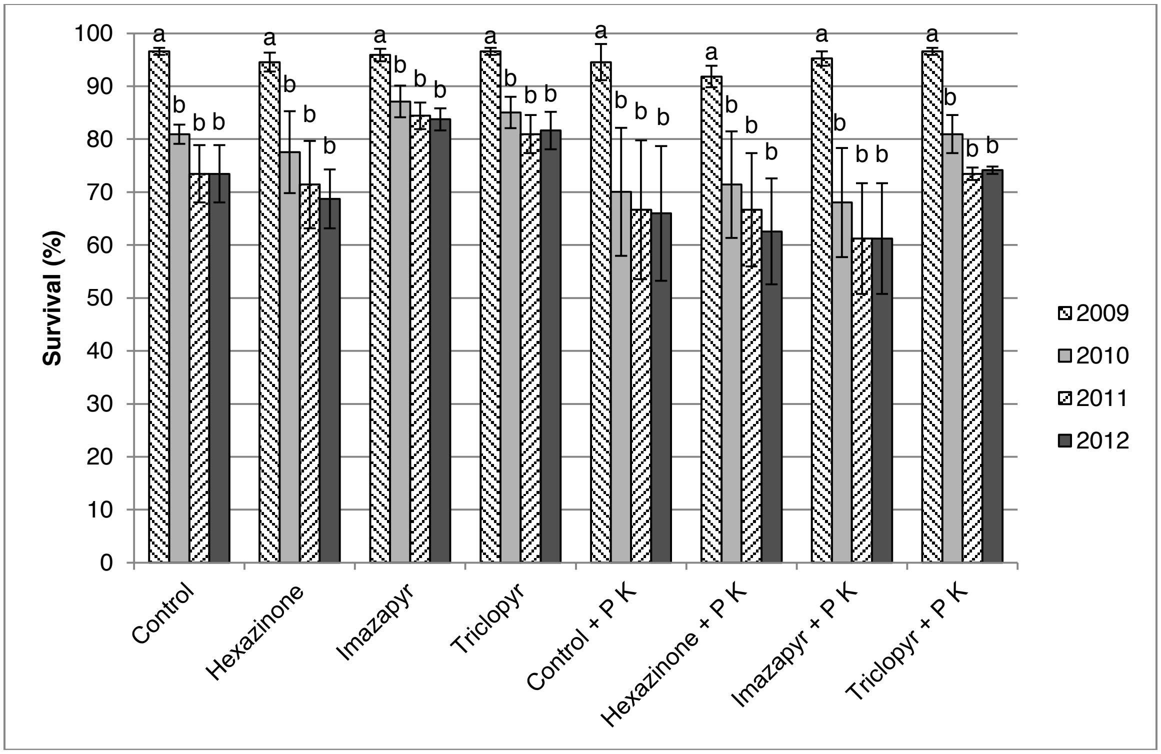

Figure 1--Mean seedling survival by treatment through four growing seasons. Bars with different letters indicate statistical significance. Winter prescribed fire occurred immediately prior to 2010 measurements.

2009). Therefore, seedling quality is unlikely to have been a factor in survival rates.

Such high initial seedling survival is not unexpected for container longleaf pine seedlings, nor is it a guarantee (Cram and others 1999; Hainds 1999; Jackson and others 2007, 2010; South and others 2005). In this case, it may be a positive result of unusually high rainfall during the year of planting (fig. 2). As expected, the survival pattern shows fewer surviving seedlings each year. The majority of the mortality occurred during the second growing season, which was an abnormally dry year that included an extended drought between August and October (fig. 2). Survival was further reduced by the 2011 prescribed fire, but this source of mortality was less than 5 percent and concentrated among smaller, lower-quality seedlings. Chemical site preparation, fertilization, and prescribed fire have been shown to produce both higher and lower survival rates 6 years after artificial regeneration (Haywood 2007) but also may not have an effect (Boyer 1988, Ramsey and Jose 2004), depending on site, climate, or other circumstances.

Seedling Height

Mean seedling heights four growing seasons after planting are presented in figure 3. Using Haywood's (2000) 4.8-inch threshold for grass stage emergence, all eight treatment means indicated active height growth. There were no significant treatment effects on seedling height, but time was again a significant factor ($F = 34.19$, $p < 0.0001$). A divergent trend is developing, however, such that the year 4 measurements were all significantly greater than the previous 3 years, whereas earlier measurements were not significantly greater than preceding ones (fig. 3). Thus, although high variation within treatments currently obscures potential differences, increasing growth rates suggest future differences in treatment effects.

Intensive vegetation management with herbicides has been documented to reduce the amount of time spent in the grass stage (Haywood 2007) and increase longleaf pine growth relative to untreated controls after 10 years (Haywood 2011). However, these effects may be transitory, as these differences can disappear over time (Boyer 1983). In one study, the level of site preparation did not affect

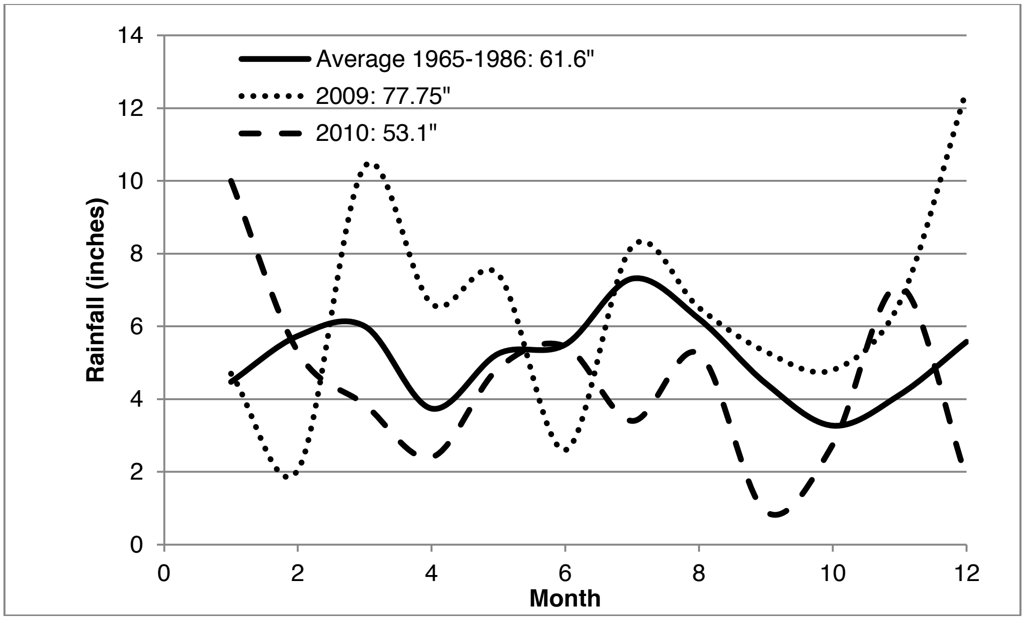

Figure 2--EEF rainfall data: long-term mean paired with totals from first (2009) and second (2010) growing seasons.

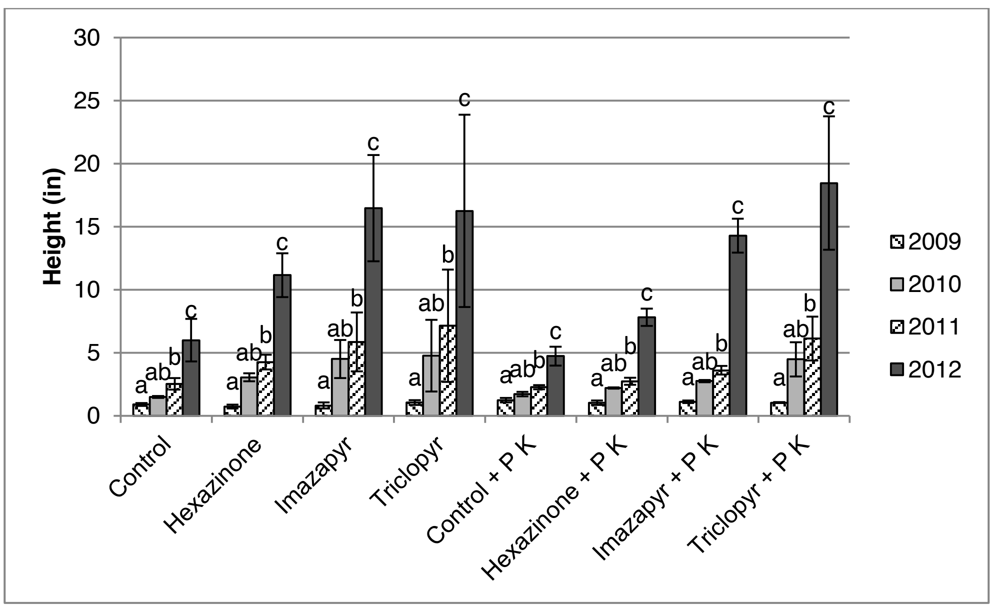

Figure 3--Mean seedling height by treatment through four growing seasons. Bars with different letters indicate statistical significance.

seedling height growth after only 3 years (Boyer 1988).

Seedling Ground-line Diameter

Seedling GLD is used as a measure of growth prior to the onset of height growth, which generally occurs as seedlings approach 1 inch (25 mm) in diameter (Boyer 1990, Wahlenberg 1946). After four growing seasons, treatment effects on mean GLD were not statistically significant (fig. 4). However, comparisons of mean seedling diameters among measurement years were significantly different in 3 of 4 years ($F = 124.42$, $p < 0.0001$). Additionally, it is worth noting that after 4 years, each treatment had surpassed or was approaching the 1-inch GLD threshold for imminent height growth. It is therefore expected that the existing numerical

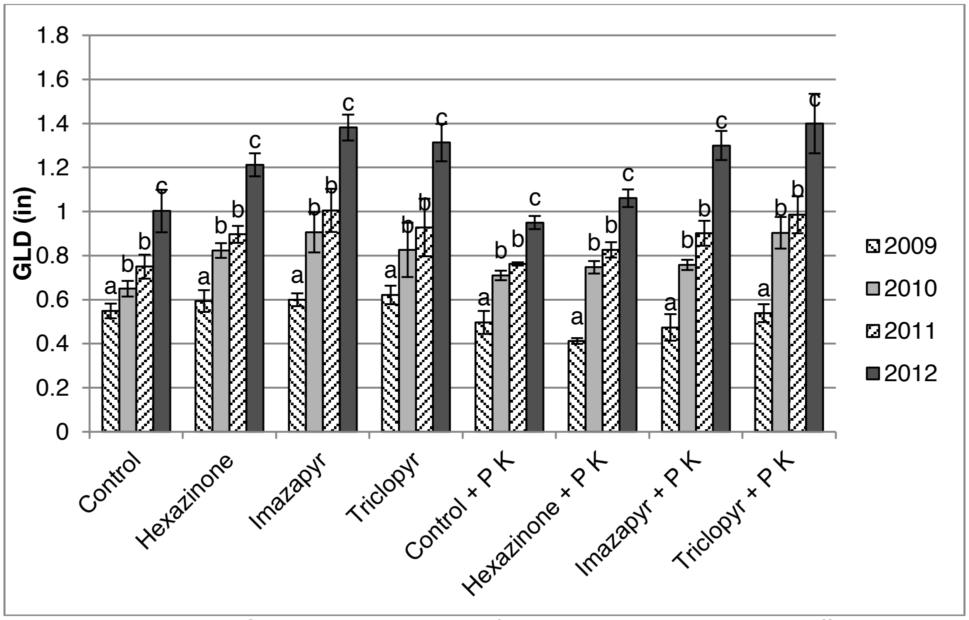

Figure 4--Mean seedling GLD by treatment through four growing seasons. Bars with different letters indicate statistical significance.

differences will be amplified and may become statistically significant in the future.

Competition

Hardwoods--Relative levels of competition were assessed annually and exhibited the same lack of significant treatment effects, but significant differences were detected among measurement years. For hardwood competition, mean number of competing stems per acre was significantly greater during the first growing season ($F = 9.51$, $p = 0.0003$) than after the following three growing seasons (table 1).

Hardwood competition data from 2009 did not show a clear pattern but suggested that the herbicide treatments were initially only marginally effective, leaving on average as few as 1,200 and as many as 22,000 stems per acre (table 1). The number of competing stems was significantly lower after the second year (2010) largely because the plots were burned by prescription following the 2010 growing season (prior to that year's measurements). In effect, fire served to remove many of the stems that had survived or only been weakened by the chemical herbicides. This indicates that herbicides alone provided a window for seedling establishment but would not generate desirable stand composition on these sites. Rather, the combination of herbicide and fire provides a longer period of reduced competition that gives artificially regenerated longleaf pine seedlings time to develop site dominance, a result similar to earlier longleaf pine restoration efforts on sandhills (Brockway and Outcalt 2000). Furthermore, the apparent "baseline" amount of hardwood competition following herbicide and fire is between 1,000 and 4,000 stems per acre, so repeated burning will be required to discourage further hardwood development. This level of remaining competition seems to support previous research showing that release treatments may be more effective in promoting height growth than intensive site preparation alone (Boyer 1988, Haywood 2007, Ramsey and others 2003).

In this study, hexazinone was relatively ineffective for hardwood control during the first growing season (table 1), which may explain why this treatment consistently showed the poorest pine growth response. Increased woody competition present in this treatment may have resulted from an application rate 20 to 40 percent lower than that commonly used on similar sites (Personal communication. 2013. W.D. Mixson, Sales and Technical Services Manager, DuPont Land Management, Pensacola, FL 32503). Other research (Haywood 2000, Ramsey and Jose 2004, Ramsey and others 2003) suggests that the rate used in this experiment is more beneficial for pine release than site preparation treatments.

Table 1--Mean number of competing hardwood and pine stems per acre through four growing seasons[a]

	Hardwood				Pine			
Treatment	2009	2010[b]	2011	2012	2009	2010*	2011	2012
Control	10,482a	2,837b	5,428b	7,401b	0c	606d	632d	629d
Hexazinone	18,309a	1,766b	1,960b	3,894b	39c	90d	142d	219d
Imazapyr	22,048a	645b	1,586b	1,702b	0c	168d	219d	232d
Triclopyr	4,165a	1,122b	2,540b	2,463b	39c	477d	593d	619d
Control + P K	16,891a	3,443b	5,854b	7,066b	90c	245d	425d	464d
Hexazinone + P K	18,696a	1,044b	632b	1,135b	168c	322d	425d	530d
Imazapyr + P K	1,251a	400b	348b	1,006b	0c	181d	232d	284d
Triclopyr + P K	1,380a	1,702b	2,514b	2,991b	0c	464d	606d	658d

[a]Values with different letters indicate statistical significance at $\alpha = 0.05$.
[b]2010 measurements occurred in winter 2011, immediately after first cycle of prescribed fire.

Similarly, the imazapyr treatment was initially ineffective at hardwood control, but its results improved with the prescribed fire after the second growing season.

Pines--Pine competition shows a pattern inverse to that of hardwood competition. The mean numbers of competing pines were significantly lower during the first growing season than in following years ($F = 27.74$, $p < 0.0001$) (table 1). The 2009 measurements show few competing pine stems, but data were collected just prior to seedfall in a "good" [> 50 cones per tree (Boyer 1996)] longleaf pine seed year at EEF (Brockway and Boyer 2010). As a result, the 2010 measurements showed a significant increase in competing pine stems. Although 2010 had a failed seed crop, 2011 produced another good seed crop at EEF (Brockway and Boyer 2011). Therefore, the net effect is a slightly positive trend in the number of competing (volunteer) longleaf pine seedlings, such that mortality among these volunteers has been more than replaced by new germinants, at levels of 200 to > 600 trees per acre (table 1). Because of the limited size of the clear-cut areas in this experiment, natural regeneration from the surrounding forest might have been adequate to restore the ecosystem without planting nursery stock. However, reliance on natural regeneration under the circumstances found in this study is risky, given the variable nature of longleaf pine seed production from year to year (Brockway and others 2006).

Effect of Fertilization

Four years after fertilizer application, there were no significant differences in mean longleaf pine height and GLD in fertilized versus unfertilized plots (figs. 3 and 4). Although fertilization can result in increased volume growth in southern pines (Dickens and others 2003), its effects on longleaf pine are not always positive (Haywood 2007). Other studies also have shown fertilizer to be an ineffective treatment in artificially-regenerated longleaf pine. In Louisiana, fertilization had no effect on longleaf pine height, basal area, or volume per tree after 10 years and even reduced stand density (Haywood 2011). Similarly, longleaf pine survival and growth were lowest on fertilized plots in a western Florida old-field study (Ramsey and others 2003).

CONCLUSIONS

Management objectives on many public and private lands now include retention of biological legacies (Franklin and others 2007) and attempt to minimize mechanical and chemical disruption of vegetation. Nevertheless, in extraordinary circumstances it may be necessary to use intensive management practices temporarily in order to rapidly restore the forest ecosystem and regain the trajectory identified in the forest management plan. Even on land managed for conservation, intensive management practices similar to those employed by production forestry can help to restore or sustain forests effectively. However, careful attention should be paid to prevent long-lasting damage from operations that disrupt ecosystem functions or impair productivity.

Prescribed fire appears vital to quickly and effectively restore damaged longleaf pine forests. Even though this experiment is located on areas burned regularly for decades, hardwood and shrub vigor exceeded the capabilities of chemical use alone, and continued burning is required for effective competition control. Given the observed importance of repeated prescribed burning in preventing hardwood succession, it is imperative that management practices be performed in a manner that does not diminish the effectiveness of surface fire. Harvest operations should be closely monitored so that equipment is either altered or relocated to prevent soil damage when necessary (Carter 2011). In this study, high-value timber was salvaged shortly after the hurricane when soils were still water-saturated, causing severe rutting in certain areas. After only 8 years, these localized microsite changes have noticeably altered understory vegetation composition, primarily by disrupting fire behavior. The impaired soil structure and altered moisture regime continue to impede the restoration process on such areas.

After four growing seasons, chemical site preparation and fertilization exhibited no significant treatment effects relative to non-treated controls. However, qualitative assessment shows that non-prepared sites result in forest structure inconsistent with both conservation and production objectives. These preliminary results suggest that some form of chemical site preparation will be required to adequately restore longleaf pine forests to full stocking and production. However, the application of phosphorus and potassium at time of planting is not justified. Rather, these results support previous findings that fertilizing longleaf pine at time of planting is either neutral or is detrimental.

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LITERATURE CITED

Boyer, W.D. 1983. Variations in height-over-age curves for young longleaf pine plantations. Forest Science. 29(1): 15-27.

Boyer, W.D. 1988. Effects of site preparation and release on the survival and growth of planted bare-root and container-grown longleaf pine. Georgia Forest Res. Pap. No. 76. Macon, GA: Georgia Forestry Commission Research Division. 8 p.

Boyer. W.D. 1990. *Pinus palustris* Mill. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America: Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 405-412. Vol. 1.

Boyer, W.D. 1996. Anticipating good longleaf pine seed crops: the key to successful natural regeneration. Alabama's Treasured Forests. 15(2): 24-26.

Brockway, D.G.; Boyer, W.D. 2010. Longleaf pine cone prospects for 2010 and 2011. [June 30, 2010, yearly report]. Available from: U.S. Department of Agriculture Forest Service, Southern Research Station, G.W. Andrews Forestry Sciences Laboratory, 521 Devall Drive, Auburn, AL 36849. 3 p.

Brockway, D.G.; Boyer, W.D. 2011. Longleaf pine cone prospects for 2011 and 2012. [May 19, 2011, yearly report]. Available from: U.S. Department of Agriculture Forest Service, Southern Research Station, G.W. Andrews Forestry Sciences Laboratory, 521 Devall Drive, Auburn, AL 36849. 3 p.

Brockway, D.G; Outcalt, K.W. 2000. Restoring longleaf pine wiregrass ecosystems: hexazinone application enhances effects of prescribed fire. Forest Ecology and Management. 137: 121-138.

Brockway, D.G.; Outcalt, K.W.; Boyer, W.D. 2006. Longleaf pine regeneration ecology and methods. In: Jose, S.; Jokela, E.J.; Miller, D.L., eds. The longleaf pine ecosystem: ecology, silviculture, and restoration. New York: Springer: 95-133.

Carter, E.A. 2011. Minimizing soil impacts from forest operations. Meeting Pap. No. 1111482. St. Joseph, MI: American Society of Agricultural and Biological Engineers. 12 p.

Conner, W.H.; Day, J.W., Jr.; Baumann, R.H.; Randall, J.M. 1989. Influence of hurricanes on coastal ecosystems along the northern Gulf of Mexico. Wetlands Ecology and Management. 1(1): 45-56.

Cram, M.M.; Mexal, J.G.; Souter, R. 1999. Successful reforestation of South Carolina sandhills is not influenced by seedling inoculation with *Pisolithus tinctorius* in the nursery. Southern Journal of Applied Forestry. 23(1): 46-52.

Craul, P.J.; Kush, J.S.; Boyer, W.D. 2005. Longleaf pine site zones. Gen. Tech. Rep. SRS-89. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 23 p.

Dickens, E.D.; Moorhead, D.J.; McElvany, B. 2003. Pine plantation fertilization. Better Crops. 90(3): 12-15.

Dumroese, R.K.; Barnett, J.P.; Jackson, D.P.; Hainds, M.J. 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. Proceedings RMRS-P-58. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 101-107.

Franklin, J.F.; Mitchell, R.J.; Palik, B.J. 2007. Natural disturbance and stand development principles for ecological forestry. Gen. Tech. Rep. NRS-19. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 44 p.

Hainds, M.J. 1999. Herbicides and longleaf pine establishment. Alabama's Treasured Forests. 18(2): 14-15.

Haywood, J.D. 2000. Mulch and hexazinone herbicide shorten the time longleaf pine seedlings are in the grass stage and increase height growth. New Forests. 19(3): 279-290.

Haywood, J.D. 2007. Influence of herbicides and felling, fertilization, and prescribed fire on longleaf pine establishment and growth through six growing seasons. New Forests. 33(3): 257-279.

Haywood, J.D. 2011. Influence of herbicides and felling, fertilization, and prescribed fire on longleaf pine growth and understory vegetation through ten growing seasons and the outcome of an ensuing wildfire. New Forests. 41(1): 55-73.

Hintze, J.L. 2007. NCSS Statistical System version 7.1.1 Quick start and self help manual. Kaysville, UT: NCSS, LLC. 104 p.

Jackson, D.P.; Dumroese, R.K.; Barnett, J.P.; Patterson, W.B. 2007. Container longleaf pine seedling morphology in response to varying rates of nitrogen fertilization in the nursery and subsequent growth after outplanting. In: Riley, L.E.; Dumroese, R.K.; Landis, T.D., tech. coords. National proceedings: forest and conservation nursery associations – 2006. Proceedings RMRS-P-50. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 114-119.

Jackson, D.P.; Dumroese, R.K.; Barnett, J.P.; Patterson, W.B. 2010. Effects of liquid fertilizer application on the morphology and outplanting success of containerized 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: 229-234.

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.

Pielke, R.A., Jr.; Landsea, C.; Mayfield, M. [and others]. 2005. Hurricanes and global warming. Bulletin of the American Meteorological Society. 86(11): 1571-1575.

Ramsey, C.L; Jose, S. 2004. Growth, survival, and physiological effects of hexazinone and sulfometuron methyl applied overtop of longleaf pine seedlings. Southern Journal of Applied Forestry. 28(1): 48-54.

Ramsey, C.L.; Jose, S.; Brecke, B.J.; Merritt, S. 2003. Growth response of longleaf pine (*Pinus palustris* Mill.) seedlings to fertilization and herbaceous weed control in an old field in southern USA. Forest Ecology and Management. 172: 281-289.

Smith, D.M.; Eade, R.; Dunstone, N.J. [and others]. 2010. Skilful multi-year predictions of Atlantic hurricane frequency. Nature Geoscience. 3(12): 846-849.

South, D.B.; Harris, S.W.; Barnett, J.P. [and others]. 2005. Effect of container type and seedling size on early height growth of *Pinus palustris* seedlings in Alabama, USA. Forest Ecology and Management. 204: 385-398.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth, and management. 1st ed. Washington, DC: Charles Lathrop Pack Forestry Foundation. 429 p. In cooperation with: The U.S. Department of Agriculture Forest Service.

THE QUEST FOR METHODS TO IDENTIFY LONGLEAF PINE STUMP RELICTS IN SOUTHEASTERN VIRGINIA

Thomas L. Eberhardt, Philip M. Sheridan, Chi-Leung So, Arvind A.R. Bhuta, and Karen G. Reed[1]

Abstract--The discovery of lightwood and turpentine stumps in southeastern Virginia raised questions about the true historical range for longleaf pine (*Pinus palustris* Mill.). Several investigative studies were therefore carried out to develop a method to determine the taxa of these relicts. Chemical approaches included the use of near infrared (NIR) spectroscopy coupled with principal component analysis and characterization of the monterpenes in stump wood extracts by gas chromatography-mass spectrometry (GC-MS). More recent efforts led to the revivification of a method involving measurements of pith and second annual ring diameters. Development of this method is still ongoing through the exploration of alternative measurements and the expansion of the data set. Results gathered thus far have been consistent with the putative range for longleaf pine in Virginia.

INTRODUCTION

Prior to colonial settlement, the vast longleaf pine (*Pinus palustris* Mill.) forest covered over 37 million ha (Croker 1987, Frost 1993, Outcalt and Sheffield 1996), ranging from southeastern Virginia to eastern Texas (Koch 1972, Wahlenberg 1946) as shown in figure 1 (inset). In Virginia, less than 323 ha remain of the estimated 607,000 ha of longleaf pine forests estimated to be present prior to colonial settlement (Frost 2006, Sheridan and others 1999). Provenance studies in Virginia have since involved the planting of longleaf pine within (Saucier and Taras 1966) and outside its putative range (Johnsen 2013).

Relicts from the harvesting of the original longleaf forest can still be found throughout the southeastern United States and have been used for tree-ring dating and the study of historical land use patterns (Grissino-Mayer and others 2001, van de Gevel and others 2009). The discovery of lightwood and turpentine stumps outside the putative range of longleaf pine in Virginia were of particular interest as possible physical evidence for justifying a northern extension of the historical range for longleaf pine (Eberhardt and Sheridan 2005). However, the presence of a stump relict alone does not provide sufficient evidence of longleaf pine. Since it is not possible to differentiate among the southern yellow pine species on the basis of anatomy (Panshin and de Zeeuw 1980), alternative methods of identification were sought. Limited utility was gleaned from spectroscopic and chemotaxonomic approaches (Eberhardt and others 2007, 2009a, 2010). Persistence in this endeavor led us to revive a method whereby measurements of the pith and the second annual ring diameters of tree disks allows longleaf pine to be distinguished from the other southern pines (Eberhardt and others 2009b, 2011). Here we provide a review and an update of our quest to develop methods that identify stump relicts, in the hope that they can provide physical evidence of the historical range of longleaf pine in Virginia.

MATERIALS AND METHODS

Method details for the characterization of stump-derived samples/specimens can be found in the publications cited in this section. Briefly, chemical analyses utilized stump relict specimens from Caroline, Prince George, Southampton, and Sussex counties in eastern Virginia. Longleaf, shortleaf (*P. echinata* Mill.) and loblolly pine (*P. taeda* L.) wood samples were included as controls. After grinding, samples were analyzed by near infrared (NIR) spectroscopy coupled with principal component analysis (PCA) (Eberhardt and others 2007) and/or GC-MS, the latter to determine the monoterpene compositions (Eberhardt and others 2009a, 2010); one of the relict stumps

[1]Research Scientist, USDA Forest Service, Southern Research Station, Pineville, LA 71360; Director, Meadowview Biological Research Station, Woodford, VA 22580; Assistant Professor, LSU Agricultural Center, School of Renewable Resources, Baton Rouge, LA 70803; Post-doctoral Fellow, Clemson University, Department of Forestry and Natural Resources, Clemson, SC 29634; and Physical Sciences Technician, USDA Forest Service, Southern Research Station, Pineville, LA 71360.

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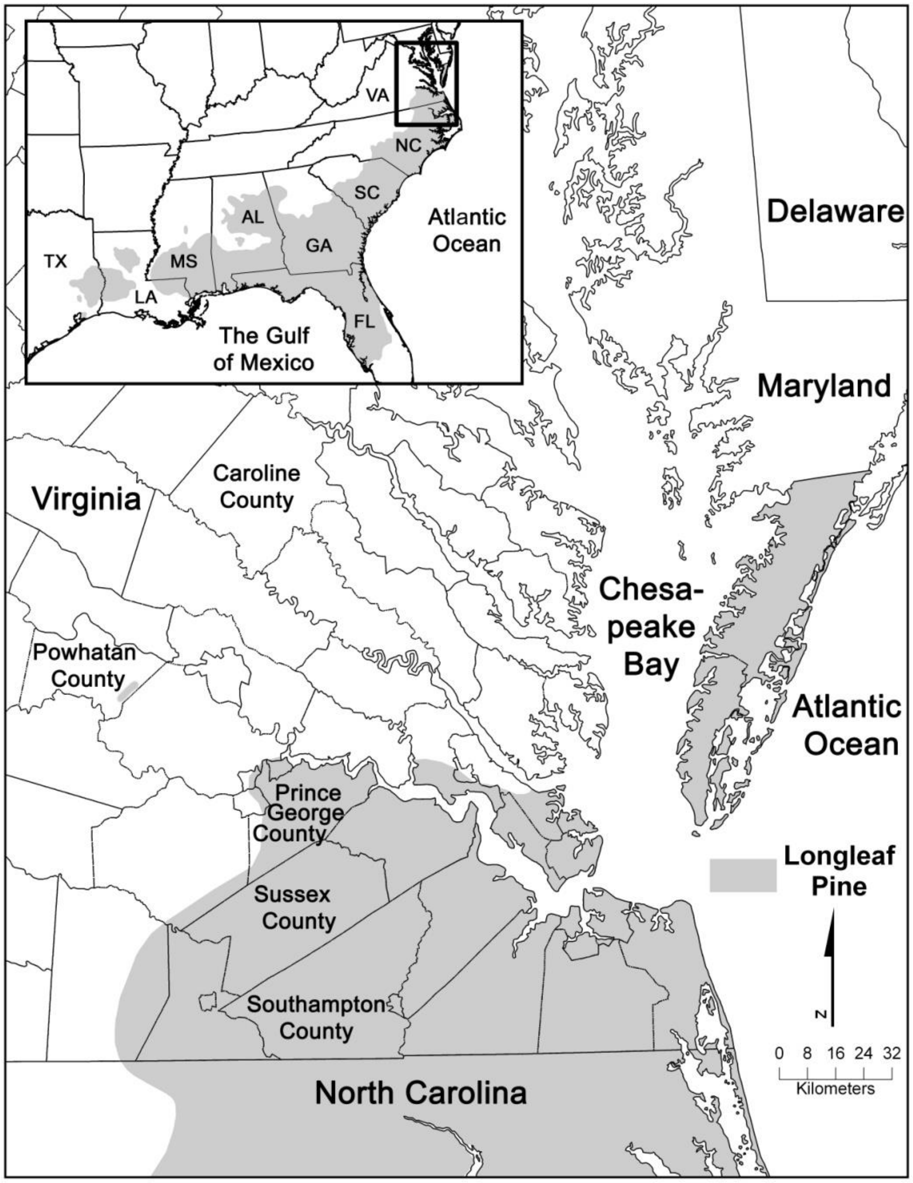

Figure 1--Map of longleaf pine ranges in Virginia and the southeastern United States (inset).

(Southampton County) subjected to gas chromatography-mass spectrometry (GC-MS) showed signs of turpentining and was therefore labeled as a turpentine stump. Validation of Koehler's method (1932) utilized southern pine tree cross-section disks from recent harvests throughout the southeastern United States. Among the relict lightwood stumps from eastern Virginia (Sussex, Prince George, Powhatan, and Caroline Counties), one from Caroline County was also labeled as being a turpentine stump. Included with the relicts was a snag from an old-growth longleaf pine tree struck by lightning (Suffolk, VA). Pith and second annual ring diameters were measured and plotted along with Koehler's (1932) delineating curve to identify those stumps belonging to longleaf pine (Eberhardt and others 2009b, 2011).

RESULTS AND DISCUSSION

Spectroscopic Identification

Our quest began after acknowledging that the stump relict samples collected in Virginia could not be identified as belonging to longleaf pine on the basis of simple wood anatomy. This led us to inquire whether our in-house expertise on the characterization of wood chemical and physical properties by NIR spectroscopy, coupled with multivariate analysis, could be applied. Stump relict samples were analyzed along with known samples of longleaf and loblolly pine wood. Application of PCA to the spectroscopic data gave the groupings shown in figure 2; a weakness in this analysis was the lack of stump relict specimens for which the species were known. Several groupings were observed with the stump relict specimens generally being separate from the longleaf and loblolly wood samples. Overall, the groupings were not sufficiently distinct to conclude that all of the stump relict specimens were of the same species or that any of them specifically belonged to longleaf pine. With these preliminary results it was concluded that even with a much larger sample set, this technique would not allow us to identify the stump relict specimens as belonging to longleaf pine.

Chemotaxonomic Identification by Monoterpene Analyses

While handling the stump relict specimens during the spectroscopic analyses, their fragrant nature suggested the presences of volatile compounds, with many likely being monoterpenes. Given the heritable nature of the monoterpenes, it seemed plausible that their characterization provided an opportunity for chemotaxonomic identification (Fäldt and others 2001, Silvestrini and others 2004, Wolff and others 1997). The most abundant monterpene among the southern pines is α-pinene, comprising 50 to 80 percent of the detected monoterpenes in oleoresin from longleaf, shortleaf, loblolly and slash pines (Hodges and others 1979, Strom and others 2002); the second most abundant monoterpene, β-pinene, typically ranges from 20 to 40 percent. Along with the pinenes, much smaller amounts of camphene, myrcene, and limonene are also often reported (Hodges and others 1979, Strom and others 2002). Whereas the most abundant monoterpenes would appear to be of little utility, it was envisioned that the minor monoterpenes might afford patterns leading to identification.

Analyses of the stump relict samples by GS-MS showed the monoterpene contents to range from 14.2 to 58.3 mg/g (Eberhardt and others 2007). The turpentine stump sampled in Southampton County was of particular interest having the highest probability of belonging to longleaf pine on the basis of a box cut, as used during turpentining operations, and historical records of the presence of longleaf pine in that particular county and at the site (Harvill and others 1986, Sheridan and others 1999). Analysis of this specimen (table 1) gave a relative amount of α-pinene (58.22 percent) that was similar to the heartwood of longleaf pine stumps (60.82 percent) (Eberhardt and others 2009a). Interestingly, the mean value for α-pinene among the remaining lightwood stump relict samples was lower (34.28 percent), offset by the presence of more oxidized monoterpenes (e.g., terpinolene, terpinen-4-ol, fenchyl alcohol). Even with access to lightwood samples for which the species were known, it became readily apparent from these data that it would be difficult to account for the changes to the monoterpene compositions over time. The only feasible utility of the technique was that it eliminated pond pine (*P. serotina* Michx.) for which limonene comprises as much as 90 percent of the detected monoterpenes of trees within the species' native range (Eberhardt and others 2010).

Measurement of Pith and Second Annual Ring Diameters

Koehler's method (1932), involving the measurement and plotting of pith and second annual ring diameters, was first validated with longleaf, shortleaf, and loblolly pine specimens taken at stump height (approximately 0.5 m); points above the delineating curve are assigned as belonging to longleaf pine. All longleaf pine specimens gave data points that when plotted could be readily assigned to longleaf pine (fig. 3). Almost all data points for loblolly and shortleaf pines were indicative of southern pines other than longleaf pine. Thus, there was no reason to suspect a dramatically different rate of false positives for currently standing timber than the second-growth timbers likely available to Koehler some 70 years ago. Given the data points shown in figure 3, it seemed that the values for the second annual ring diameter were lower than those for the other southern pines; whereas the very rapid growth of loblolly pine is manifest in the measurements of second annual ring diameters reaching 54.88 mm, the largest value for longleaf pine was 40 mm. Identification and measurement of second annual ring diameters was therefore suggested as an adaptation to the method for tentative evidence in situations where the pith is missing or decayed away (Eberhardt and others 2011).

In addition to currently growing southern pines, we were fortunate to gain access to an old-growth longleaf pine near Suffolk, VA that had been killed by lightning. Measurements from this

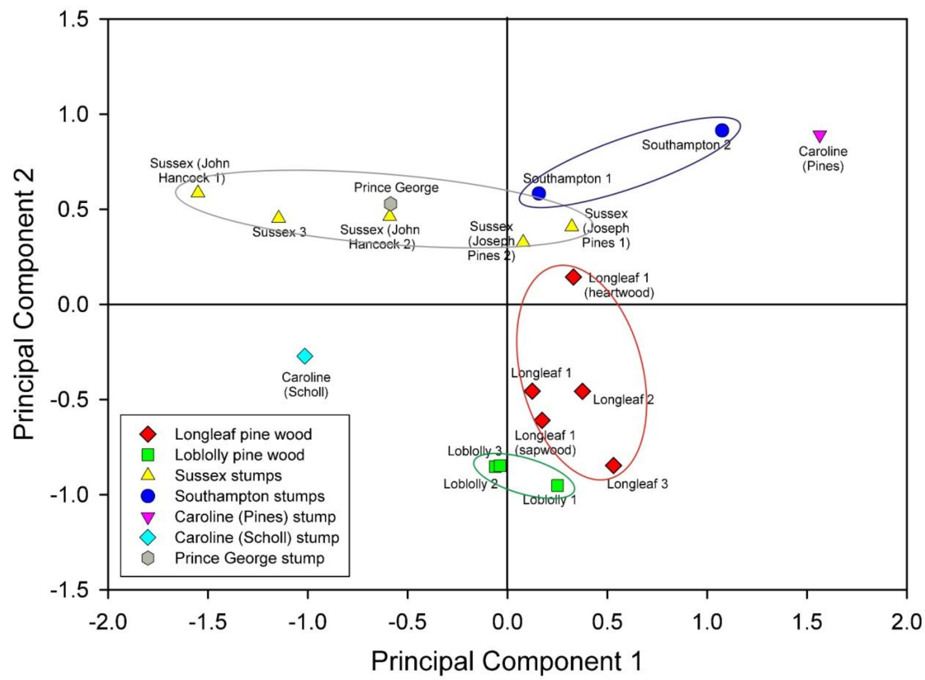

Figure 2--Principal component analysis of samples from stump relicts and controls (longleaf and loblolly pine wood). Figure reproduced from Eberhardt and others 2007.

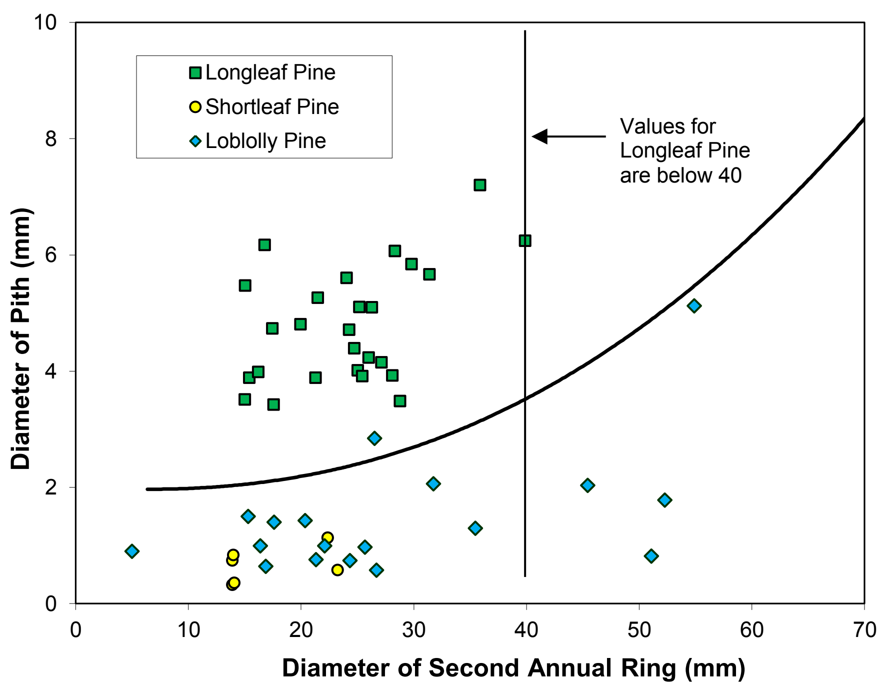

Figure 3--Plot of pith and second annual ring diameter measurements for longleaf, shortleaf and loblolly pine specimens. Figure adapted from Eberhardt and others 2011.

Table 1--Relative compositions of monoterpenes and methyl chavicol detected in stump wood samples: recently-harvested longleaf pine in central Louisiana; lightwood and turpentine stumps in eastern Virginia (ND = not detected)

Compound	Longleaf pine stumps	Unknown lightwood stumps	Southampton turpentine stump
	-------------------------*percent*	----------------------------	
α-Pinene	60.82	34.28	58.22
α-Fenchene	0.21	2.14	0.58
Camphene	1.39	3.97	3.10
β-Pinene	9.79	1.34	1.25
Myrcene	0.92	0.67	0.03
α-Phellandrene	0.03	0.73	ND
α-Terpinene	0.08	0.52	ND
Limonene	3.30	5.29	9.29
β-Phellandrene	0.36	1.38	ND
p-Cymene	0.01	13.87	0.28
Terpinolene	1.55	1.20	1.67
Fenchone	ND	3.89	0.26
Camphor	ND	6.40	0.82
Fenchyl Alcohol	2.13	1.66	1.93
Terpinen-4-ol	0.90	3.80	0.93
Methyl Chavicol	5.03	1.65	2.55
α-Terpineol	11.84	14.83	16.18
Borneol	1.64	2.28	2.91
Total	100.00	100.00	100.00

tree were well within the limits for longleaf pine (fig. 4). Lightwood stump relicts located within (Powhatan, Prince George, and Sussex counties) and outside (Caroline County) the putative range of longleaf pine in eastern Virginia gave data points on the plot that excluded longleaf pine. A few loblolly pine trees in Virginia were also sampled and gave similar results. Turpentine scars were suggested by Koehler (1932) as an indicator of longleaf pine. Contradicting this indicator was a result that showed the turpentine stump in Caroline County was not longleaf pine. This result was particularly intriguing since it could substantiate the historical report that loblolly pine was subjected to turpentining operations but without commercial success (Wahlenberg 1960).

Finally, additional samples of longleaf, shortleaf, and loblolly pine were accessed from dendrochronological studies throughout the southeastern United States (Bhuta and others 2008, 2009). These samples were independently measured, and therefore demonstrate that the technique is indeed robust. No patterns were observed in this plot (fig. 5) to suggest that the pith and second annual ring measurements are influenced by the ecological region from which they were taken.

CONCLUSIONS

It is unlikely that NIR spectroscopy coupled with PCA will provide a means to identify stump relict specimens as belonging to longleaf pine, even with a much larger sample set. Given the difficulties in accounting for the changes to the monoterpene compositions over time, the chemotaxonomic identification of longleaf pine would be difficult to achieve; this technique may only allow the exclusion of pond pine for which limonene is the most abundant monoterpene. Among the methods that we tested, the only robust technique for the identification of stump relicts as belonging to longleaf pine involves the measurement of pith and second annual ring diameters. Results collected to date do not provide physical evidence for a northern range extension for the historical range of longleaf pine in Virginia.

ACKNOWLEDGMENTS

James Scarborough, Jacob Floyd, Dan Leduc, Donna Edwards, Larry Lott, C. Dana Nelson, Don Bragg, and Rick Stagg are acknowledged for assistance in collection of control samples.

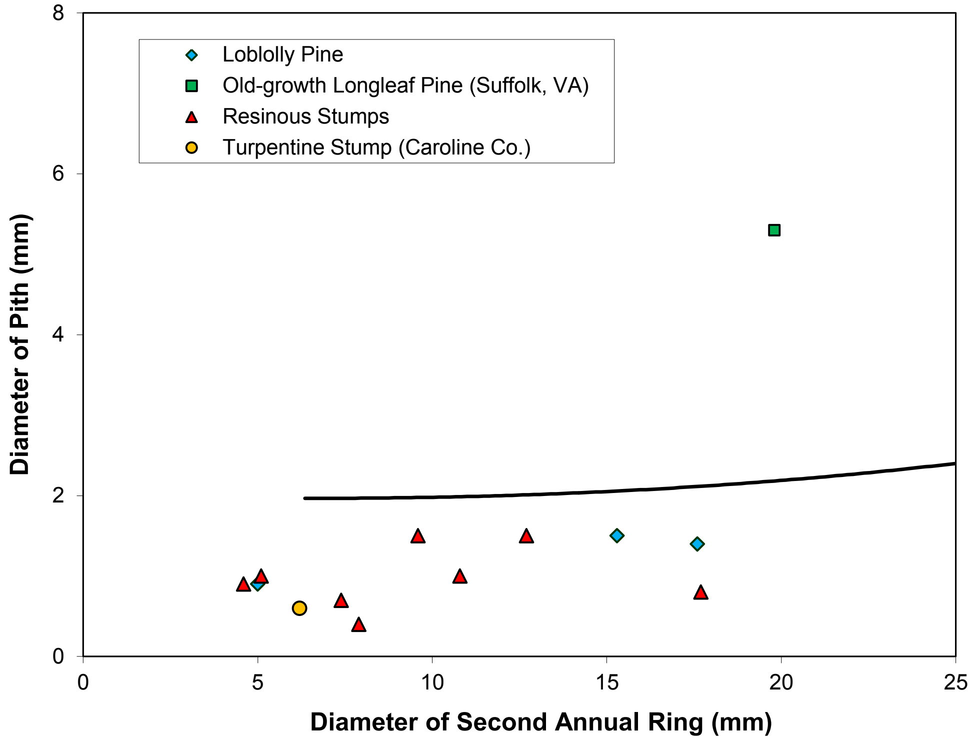

Figure 4--Plot of pith and second annual ring diameter measurements for loblolly pine tree and relict specimens collected in southeastern Virginia. Figure adapted from Eberhardt and others 2011.

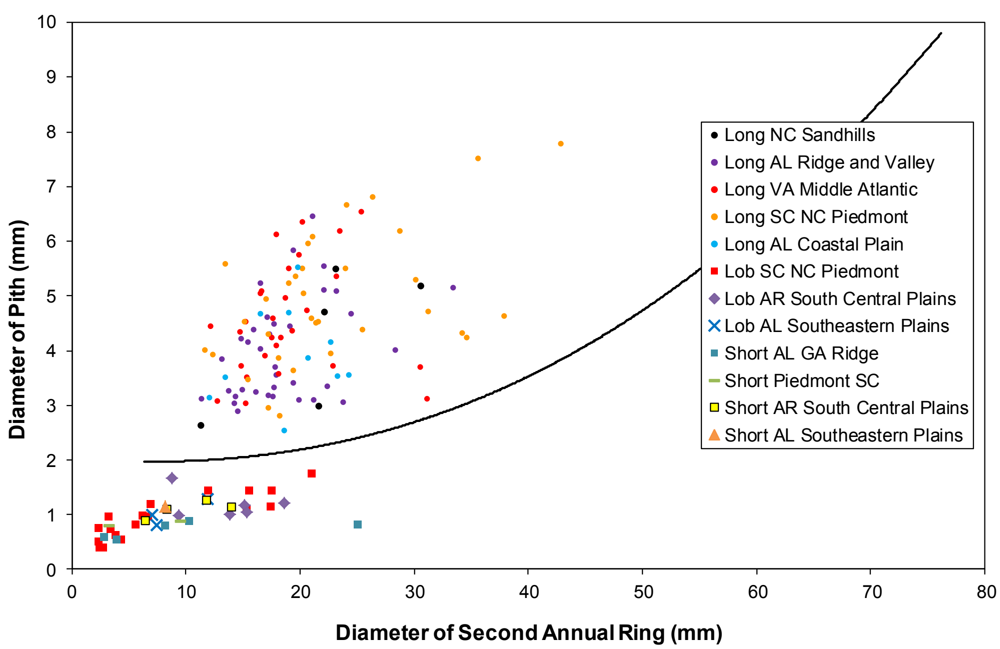

Figure 5--Plot of independently-collected pith and second annual ring diameter measurements for longleaf, shortleaf and loblolly pine specimens taken in various ecological regions across the southeastern U.S.

Jolie Mahfouz carried out all monoterpene analyses.

LITERATURE CITED

Bhuta, A.A.R.; Kennedy, L.M.; Copenheaver, C.A. [and others]. 2008. Boundary-line growth patterns to determine disturbance history of remnant longleaf pine (*Pinus palustris* P. Mill.) in mixed forests of southeastern Virginia. Journal of the Torrey Botanical Society. 135(4): 516-529.

Bhuta, A.A.R.; Kennedy, L.M.; Pederson, N. 2009. Climate-radial growth relationships of northern latitudinal range margin longleaf pine (*Pinus palustris* P. Mill.) in the Atlantic coastal plain of southeastern Virginia. Tree-Ring Research. 65(2): 105-115.

Croker, T.C. 1987. Longleaf pine. A history of man and a forest. Forestry Rep. R8-FR-7. Atlanta, GA: U.S. Department of Agriculture Forest Service. 37 p.

Eberhardt, T.; Sheridan, P. 2005. CSI—longleaf pine! In Virginia, verifying the tree's original range. Compass. 1(3): 20-21.

Eberhardt, T.L.; Sheridan, P.M.; Bhuta, A.A.R. 2011. Revivification of a method for identifying longleaf pine timber and its application to southern pine relicts in southeastern Virginia. Canadian Journal of Forest Research. 41: 2440-2447.

Eberhardt, T.L.; Sheridan, P.M.; Mahfouz, J.M. 2009a. Monoterpene persistence in the sapwood and heartwood of longleaf pine stumps: assessment of differences in composition and stability under field conditions. Canadian Journal of Forest Research. 39: 1357-1365.

Eberhardt, T.L.; Sheridan, P.M.; Mahfouz, J.M.; So, C-L. 2007. Old resinous turpentine stumps as an indicator of the range of longleaf pine in southeastern, Virginia. In: Estes, B.L.; Kush, J.S., eds. Longleaf pine: seeing the forest through the trees. Proceedings of the 6th Longleaf Alliance regional conference. Longleaf Alliance Report. 10: 79-82.

Eberhardt, T.L.; Sheridan, P.M.; Reed, K.G. 2009b. Surfing the Koehler curve: revisiting a method for the identification of pine stumps and logs. In: Bowersock, E.P.; Hermann, S.M., Kush, J.S., eds. Forestry in a changing world: new challenges and opportunities. Proceedings of the 7th Longleaf Alliance regional conference and Forest Guild annual meeting. Longleaf Alliance Report. 14: 85-86.

Eberhardt, T.L.; Sheridan, P.M.; Reed, K.G. 2010. Pondering the monoterpene composition of *Pinus Serotina* Michx.: can limonene be used as a chemotaxonomic marker for the identification of old turpentine stumps? 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: 535-537.

Fäldt, J.; Sjödin, K.; Persson, M. [and others]. 2001. Correlations between selected monoterpene hydrocarbons in the xylem of six *Pinus* (Pinaceae) species. Chemoecology. 11: 97-106.

Frost, C.C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. In: Herman, S.H., ed. The longleaf pine ecosystem: ecology, restoration and management. Proceedings of the 18th Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station. 18: 14-43.

Frost, C.C. 2006. History and future of the longleaf pine ecosystem. In: Jose, S.; Jokela, E.J.; Miller, D.L., eds. The longleaf pine ecosystem: ecology, silviculture, and restoration. New York: Springer: 9-42.

Grissino-Mayer, H.D.; Blount, H.C.; Miller, A.C. 2001. Tree-ring dating and the ethnohistory of naval stores industry in southern Georgia. Tree-Ring Research. 57: 3-13.

Harvill, A.M., Jr.; Bradley, T.R.; Stevens, C.E. [and others]. 1986. Atlas of the Virginia flora. Farmville, VA: Virginia Botanical Associates. 135 p.

Hodges, J.D.; Elam, W.W.; Watson, W.F.; Nebeker, T.E. 1979. Oleoresin characteristics and susceptibility of four southern pines to southern pine beetle (Coleoptera: Scolytidae) attacks. Canadian Entomologist. 111: 889-896.

Johnsen, K. 2013. Longleaf pine in Virginia: a provenance test [Abstract]. In: Abstracts 17th biennial southern silvicultural research conference. Louisiana Tech University. p. 17. In cooperation with: The U.S. Department of Agriculture Forest Service, Southern Research Station.

Koch, P. 1972. Utilization of the southern pines. Agric. Handb. 420. Washington, DC: U.S. Department of Agriculture. 2 vol.

Koehler, A. 1932. The identification of longleaf pine timbers. The Southern Lumberman. 145: 36-37.

Outcalt, K.W.; Sheffield, R.M. 1996. The longleaf pine forest: trends and current conditions. Resour. Bull. SRS-9. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 23 p.

Panshin, A.J.; de Zeeuw, C. 1980. Textbook of wood technology. 4th ed. New York: McGraw-Hill. 722 p.

Saucier, J.R.; Taras, M.A. 1966. Wood density variation among six longleaf pine seed sources grown in Virginia. Journal of Forestry. 64(7): 463-465.

Sheridan, P.M.; Scrivani, J.; Penick, N.; Simpson, A. 1999. A census of longleaf pine in Virginia. In: Kush, J.S., ed. Longleaf pine: a forward look. Proceedings of the 2d Longleaf Alliance conference. Longleaf Alliance Report. 4: 154-162.

Silvestrini, E.; Michelozzi, M.; Skroppa, T. [and others]. 2004. Characterization of different clones of *Picea abies* (L.) Karst using head-space sampling of cortical tissues combined with enantioselective capillary gas chromatography for the separation of chiral and non-chiral monoterpenes. Journal of Chromatography A. 1034: 183-189.

Strom, B.L.; Goyer, R.A.; Ingram, L.L., Jr. [and others]. 2002. Oleoresin characteristics of progeny of loblolly pines that escaped attack by the southern pine beetle. Forest Ecology and Management. 158: 169-178.

van de Gevel, S.L.; Hart, J.L.; Grissino-Mayer, H.D.; Robinson, K.W. 2009. Tree-ring dating of old-growth longleaf pine (*Pinus palustris* Mill.) logs from an exposed timber crib dam, Hope Mills, North Carolina, U.S.A. Tree-Ring Research. 65(1): 69-80.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth and management. 1st ed. Washington, DC: Charles Lathrop Pack Forestry

Foundation. 429 p. In cooperation with: The U.S. Department of Agriculture Forest Service.

Wahlenberg, W.G. 1960. Loblolly pine: its use, ecology, regeneration, protection, growth and management. Durham, NC: Duke University, School of Forestry. 421 p.

Wolff, R.L.; Deluc, L.G.; Marpeau, A.M.; Comps, B. 1997. Chemotaxonomic differentiation of conifer families and genera based on the seed oil fatty acid compositions: multivariate analyses. Trees. 12: 57-65.

DESTROYED VIRGIN LONGLEAF PINE STAND LIVES-ON DIGITALLY

John C. Gilbert, John S. Kush, and Rebecca J. Barlow[1]

Abstract--The Flomaton Natural Area (FNA) once stood as one of the few remnant fragments of virgin, old-growth longleaf pine stands (*Pinus palustris* Mill.) in the Southeast. This 80-acre stand contained trees over 200 years old. A restoration effort began in1994 to remove off-site trees and to reintroduce fire to the site after over 40 years of fire suppression. A geographic information system (GIS) database was created by compiling the digital data recorded for the FNA, including a stem-map of all longleaf pines ≥ 1-inch d.b.h. (diameter at breast height). The database also includes ages, heights, and crown class information, which provides opportunities for a 3-dimensional digital view of the stand structure. The GIS database contains information for over 4,000 trees. It provides a unique opportunity to spatially explore longleaf pine stand dynamics of a virgin stand and to learn more about long-term management of longleaf pine. The variations in densities, size classes, and ages across the stand will be evaluated to provide information about how longleaf pine grows and the stand dynamics of virgin, old-growth longleaf pine. Gap dynamics of openings in the stand will also be examined, including information about successful regeneration. Despite the successful restoration work and demands to save the stand, the FNA was clearcut in 2008. The stand now lives on in digital form and continues to serve as an educational tool and as a beacon for the acts of mismanagement and loss of the longleaf pine ecosystem today.

INTRODUCTION

The Flomaton Natural Area (FNA) once stood as one of the few remnant fragments of virgin, old-growth longleaf pine stands (*Pinus palustris* Mill.) in the Southeast. This 80-acre privately owned stand was located within the city limits of Flomaton, AL. The FNA was a good representation of virgin stands of longleaf pine described by early authors like Schwarz (1907) and Wahlenberg (1946) based on the size and age distribution of the trees. Varner and Kush (2004) described the FNA as one of 15 remnant old-growth sites left across the species' range, which only represented 0.00014 percent of presettlement longleaf pine extent collectively. This stand had trees over 200 years old and was burned regularly until 1950 when all burning and fuel management ceased until 1994.

A restoration effort began in1994 to remove off-site trees and to reintroduce fire to the site after over 40 years of unnatural fuel accumulations and the associated risk of wildfire. These treatments created opportunities for successful longleaf pine natural regeneration and the potential for recovery of its understory structure and associated plant and wildlife components (Kush and others 2004). During over 20 years of research and restoration work on the FNA, detailed stand information was recorded for the site and used for monitoring changes to the overstory, understory, and soils associated with the restoration efforts. The field data recorded for the FNA provide a unique opportunity to gain insight into its stand dynamics and to learn more to aid long-term management of longleaf pine.

Despite the successful restoration work and demands to save the stand, the FNA was clearcut in 2008. Kush (2009) published an obituary for the FNA describing its restoration and demise. Now, only the digital data remain as a record of the stand; it provides opportunities for spatial representations of the FNA that can be used by coming generations to learn more about virgin old-growth longleaf pine. The FNA lives on in digital form and will serve as an educational tool to future researchers and managers.

METHODS

Stem-mapped Longleaf Pine

A geographic information system (GIS) database was created by compiling the field data recorded for the FNA and converting it to digital form, including a stem-map of all longleaf pines ≥ 1 inch d.b.h. (diameter at breast height). All GIS database development and analyses were completed in ESRI's ArcGIS 10. To create the stem-map, strategic point locations were setup throughout the stand, and coordinates were recorded with a sub-meter Global Positioning System (GPS) receiver at each point. Azimuth and distance measurements were recorded from the points to the surrounding trees with a survey laser. Coordinates were then calculated for each tree and entered into ArcGIS 10 to create a point

[1]Research Associate and Research Fellow, respectively, Auburn University, School of Forestry and Wildlife Sciences, Auburn University, AL 36849; and Assistant Extension Professor, Auburn University, Alabama Cooperative Extension System, School of Forestry and Wildlife Sciences, Auburn University, AL 36849.

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shapefile. The field data recorded for the trees included d.b.h., crown classification information, ring count at breast height, heights, and litter depths. The next step was to combine the point data with the field data to create a robust GIS database for the FNA. The database provides a unique opportunity to spatially explore longleaf pine stand dynamics of a virgin stand and to learn more about long-term management of longleaf pine. The variations in densities, size classes, and ages across the stand will be evaluated to provide information about how longleaf pine grows and the stand dynamics of virgin, old-growth longleaf pine. Gap dynamics of openings in the stand will also be examined, including information about successful regeneration.

Two Dimensional (2D) Digital Representations

A layout of just a single symbol for each tree shows the spatial distribution of the trees stem-mapped on the FNA. To provide the viewer with a chance to see the spatial distribution and size of trees, the symbols for each tree were scaled relative to its measured d.b.h. Adding more detail to the 2D representation, separate symbols were added for each of the crown classifications. The symbols were then scaled relative to its measured d.b.h. for each tree. This type of layout provides the spatial distribution, size, and crown position in one layout.

Raster Interpolation

The extensive number of points where data were collected on the FNA provides opportunities to go beyond simply looking at individual points. Due to limitations of funding, time, and efficiency, measurements cannot be taken everywhere. Data for the known points are used to interpolate or predict values for the surrounding areas where data were not collected. With the good coverage of point data, the inverse distance weighted (IDW) technique was used to calculate interpolations for the FNA, within the boundary of the stand. Other methods of raster interpolation techniques like nearest neighbor were attempted, but the IDW technique provided the best representation of the conditions on the ground. ESRI's ArcGIS 10 3D Analyst and the Raster Interpolation (IDW) tool were used for the analysis with litter depths, d.b.h. measurements, and ring counts at breast height (ESRI 2010). The results of the interpolations for each analysis were then classified into pertinent categories to display the data for each metric.

Three Dimensional (3D) Digital Representations

ESRI's ArcScene 10.0 was used to create an interactive 3D model of the stem-mapped longleaf pine on the FNA (ESRI 2010). The stem-map point shapefile was added to the view in ArcScene. Within ArcScene, a user can zoom into the stand, view individual trees or gaps, turn the view in any direction, and even fly through the stand. The closest representation to a longleaf pine tree was chosen from the 3D Trees Style Reference list under the symbol selector. To show the relative size of each tree, the 3D symbols for each stem-mapped longleaf pine were scaled using the measured diameters. The 3D stem-map was compared to the 2D representations to see differences in viewing individual trees, clusters of trees, gaps, and the stand as whole. Simulations were then completed by selecting trees using the attributes in the database. When the trees are selected, ArcScene adds a blue box around each symbol selected. Various simulations were completed to show the distribution of certain size classes, such as regeneration and older trees across the FNA.

RESULTS

2D Digital Representations

The longleaf pine stem-mapped data includes data for 4,167 trees. Diameters were measured to the nearest 0.1 inch. Measurements ranged from 0.7 to 32.3 inches d.b.h. Crown classifications were recorded for trees on the FNA. Classifications in the database for each longleaf pine include dominant, co-dominant, intermediate, suppressed, and no data. Two layouts were created to show a 2D view of the FNA. The first layout with a single symbol scaled for d.b.h. shows the true distribution of size classes for the longleaf pine on the FNA. Large trees were scattered across the stand, often in clumps. Smaller symbols representing the regeneration filled in gaps across the stand and were dense in the open utility line running through the northern portion. A second 2D view of the stand was created by showing different symbols for each crown classification and then scaling the symbol by d.b.h. This layout provides a pseudo-3D view of the stand by adding a classification for crown position along with the tree size, showing a good relationship between the two.

Raster Interpolation of Litter Depth
Due to the lack of fire for over 40 years, the FNA had unnatural fuel accumulation which was a major concern of the restoration effort. Extensive litter and duff measurements were taken across the FNA. The database contains over 3,000 measurements to the nearest 0.1 inch, with depths ranging from 0.5 to 16 inches. The use of prescribed fire was very difficult, especially around large old trees. To help create a better planning tool for prescribed fire, a raster IDW interpolation surface of litter depth was created for the FNA using the point data. The litter depths were classified in five categories: 0.5 to 2.5 inches, 2.6 to 5.0 inches, 5.1 to 7.5 inches, 7.6 to 10.0 inches, and 10.1 to 16.0 inches. The raster interpolation created a surface for the FNA showing the distribution of litter depths across the stand using a light- to dark-brown color ramp to show the increasing litter accumulations. The high accumulations show up in areas with high densities and larger trees. This surface could have been very useful for prescribed burn planning and mop-up during the FNA restoration by providing opportunities to prioritize areas known to have high accumulations of litter and near older trees. It could have aided in areas to focus mop-up procedures that might have prevented the loss of trees.

Raster Interpolation of D.B.H. and Ring Count
The IDW surface for diameters shows the size distribution across the FNA that highlights dense areas and gaps. The IDW surface was classified into nine categories using breaks in the data. The categories were < 4, 4 to 7, 8 to 10, 11 to 13, 14 to 17, 18 to 20, 21 to 23, 24 to 27, and > 27 inches d.b.h. The surface showed the distribution of tree sizes across the stand using a light- to dark-brown color ramp to show the increasing value for d.b.h. The surface showed the pockets of larger trees and clumps of smaller trees in a patchy structure. Basal areas in some of these dense pockets are well over 200 square feet per acre with trees from a variety of sizes.

The ring count at breast height surface also provides information about the variation of trees in the dense areas and gaps. Recorded ring counts at breast height ranged from 3 to approaching 300 years. The IDW surface was classified into nine categories using breaks in the data. The categories were 3 to 33, 34 to 48, 49 to 60, 61 to 72, 73 to 83, 84 to 95, 96 to 109, 110 to 133, and 134 to 267 rings counted at breast height. Due to potential variations in the amount of time it may have taken trees to reach breast height and to the occurrence of red-heart fungus (*Phellinus pini*), the authors estimate trees were over 300 years old. The surface showed the distribution of ring counts at breast height across the stand using a light- to dark-brown color ramp to show the increasing number of rings counted. The surface showed a good representation of clumps of older trees and areas where regeneration was present.

3D Digital Representations
The 3D representations created for the FNA provide an interactive digital recreation that can be used to explore the stand from almost any angle. Using ArcScene 10, the stand as a whole can be rotated in any direction. The user can zoom into the stand and look at individual trees or groups of trees. Examples of the entire stand can be shown from above, the side, and a profile view. Quick simple summary data can be created by utilizing the database of individual tree metrics in the attribute table (not shown). The frequency distribution for the d.b.h. field showed a 'reverse j' distribution, common for uneven-aged stands. An example simulation was also created to show the regeneration on the FNA by selecting all trees from 0.7- to 7.5-inches d.b.h. The interactive map and summary statics showed 1,897 trees within the selection and their distribution across the FNA. Potential users with a license for ArcScene could continue to explore the stand and create a variety of simulations and examples for longleaf pine restoration and management.

DISCUSSION
Although the FNA no longer exists as a physical stand, the 2D and 3D representations of it continue to provide valuable information for future generations. The 2D representations provide a good view of the spatial distribution of the stand including d.b.h. and crown positions. These conventional representations could easily be incorporated into slide presentations, poster displays, and webpages. Since stem-maps of virgin old-growth longleaf pine stands are rare, users could learn more about these systems by viewing layouts or exploring the stand with GIS and conducting additional analyses or simulations like thinnings to test different management options. The shapefiles could also be converted to a file format for use in programs like Google Earth, which could then be used to

reach a broader audience not familiar with ArcGIS. The raster interpolations take the digital representations to the next level by showing surfaces for d.b.h., ring count at d.b.h., and litter depths. These d.b.h. and ring-count surfaces provide viewers with a chance to see the patchy nature of the uneven-aged stand by highlighting groups of similar-sized trees and also similar ring counts at d.b.h. or approximate ages. These representations help viewers see the patchy nature of an old-growth longleaf pine stand but also to show the tree groupings that are difficult to see in the 2D representations. The litter depth raster interpolation provides a surface showing areas where litter depths could be problems for prescribed fire operations. Examples like this from the FNA could continue to provide educational products to practitioners interested in longleaf pine restoration and in reintroducing fire into old-growth longleaf pine stands. The 3D representations using ArcScene bring the digital forest to life. These 3D representations provide opportunities for virtual simulations of the stand that can be used to explore stand dynamics of the FNA, from virtual tours to complex individual tree or gap dynamics. The FNA database provides many opportunities for future work. Taking the next step to customize trees and regeneration using more of the recorded individual tree metrics will make a more accurate display. These digital representations can be used to develop online educational tools for restoration and conservation of longleaf pine by providing a wide variety of interactive virtual simulations such as thinning and selection cutting, including group and individual-tree selections. These educational tools could be used to help the FNA reach natural resources professionals, students, private landowners, and other audiences.

CONCLUSIONS

There are still many unanswered questions about longleaf pine stand dynamics, especially for virgin, old-growth stands. Stands like the FNA are truly not replaceable. With the continued loss of old-growth longleaf pine stands and the scarcity of virgin stands, there are very few opportunities to visit and to learn about these ecosystems. Although we are in the midst of a resurgence of interest in longleaf pine, these efforts often concentrate on planting, while stands like the FNA are lost for future generations. Despite the physical loss of the FNA, it has been captured in digital form. The FNA will continue to serve as an educational tool, virtual demonstration, and a beacon against mismanagement and loss of longleaf pine ecosystems.

LITERATURE CITED

ESRI 2010. ArcGIS Resource Center Desktop 10 Help. URL: http://help.arcgis.com/en/arcgisdesktop/10.0/help/index.html#/What_is_ArcGIS_Desktop/00v200000005000000/ [Date accessed: May 31, 2013).

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(1): 6-10.

Kush, J.S. 2009. Obituary for Flomaton: one less virgin longleaf stand (Alabama). Ecological Restoration. 27(3): 247-248.

Schwarz, G.F. 1907. The longleaf pine in virgin forest. A silvical study. New York: John Wiley & Sons. 135 p.

Varner, J.M., III; Kush, J.S. 2004. Remnant old-growth longleaf pine (*Pinus palustris* Mill.) savannas and forests of the southeastern USA: status and threats. Natural Areas Journal. 24(2): 141-149.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth and management. 1st ed. Washington, DC: Charles Lathrop Pack Forestry Foundation. 429 p. In cooperation with: The U.S. Department of Agriculture Forest Service.

ECONOMIC ANALYSIS OF PAYMENTS REQUIRED TO ESTABLISH LONGLEAF PINE HABITAT ON PRIVATE LANDS TO FACILITATE TRAINING ON MILITARY INSTALLATIONS

J. Viola Glenn, Fred Cubbage, Ron Myers, and M. Nils Peterson[1]

Steady population growth, urbanization, increased military presence, and the 2030 completion of two significant "super highway" infrastructure projects stand to fundamentally reshape the landscape in eastern North Carolina and increase pressure on the state's land-based industries (Marstel-Day 2012). With similar trends occurring throughout rural communities in the Southeast, the question of how to address tradeoffs between land-based economic sectors will become increasingly vital to policymakers and planners. In this paper, we analyze a representative tradeoff in eastern North Carolina: the tradeoff between military training capabilities and the protection of essential habitat for the endangered red-cockaded woodpecker (RCW) (*Picoides borealis*).

The analysis valued land management options consistent with habitat requirements for the endangered RCW and the relevant rural alternatives for the region: private nonindustrial forestry for maximum timber revenue and row agriculture. The habitat scenario is essentially a lengthy longleaf pine (*Pinus palustris* Mill.) rotation while the forestry alternatives include both a short rotation loblolly pine (*P. taeda* L.) scenario and a shorter longleaf pine rotation. Row agriculture is represented by corn and soybean crops. Comparing these economic valuations provided a baseline estimate of the cost to private landowners of managing land for RCW, referred to in this paper as the opportunity cost. The opportunity cost included both: (1) the direct cost to the landowner of managing the forest for habitat (e.g., planting, prescribed burning); and (2) the income the landowner must forgo when choosing to manage for habitat rather than timber revenue.

Opportunity costs are calculated using standard capital budgeting methods with particular emphasis given to the soil expectation value (SEV). SEV is calculated as a function of net present value (NPV) and represents the value today of managing under the same regime into perpetuity. SEV is especially useful in comparing land management options of varying timeframes as it converts everything to the same time scale. A 4 percent real discount rate is assumed throughout. This is standard for longleaf pine literature and consistent with USDA Forest Service methodology.

We developed longleaf management scenarios based on those common in the literature and expert insight from the North Carolina Forest Service (NCFS). Four key components drove the longleaf pine analyses: management intensity, pine straw revenue, timber revenue, and management costs. Timber volume per acre was estimated using three longleaf pine growth and yield models: NATYIELD (Smith and Hafley 1986), Farrar (1985), and Lohrey and Bailey (1977). Each scenario was assessed with three levels of pine straw revenue representing low- to mid-range literature estimates (Dickens and others 2012) and local pine straw sales. Under the conventional scenario, the loblolly pine thinning and harvest volumes and management regime were based on prior research by Siry and others (2001) and Cubbage and others (2012), which used the TAUYIELD growth and yield model.

Results are presented in table 1. The agricultural scenarios assumed average North Carolina coastal plain crop returns for corn and soybean farms each year into perpetuity: $67.57 and $159.92 per acre, respectively (NCSU 2013). These crop assumptions were optimistic and assumed that farmers would get average yields, maintain the current high crop prices, and encounter no weather or climate issues. SEVs

[1]Agricultural/Resource Economist, RTI International, Durham, NC 27709; Professor, North Carolina State University, Department of Forestry and Environmental Resources, Raleigh, NC 27607; Staff Forester, North Carolina Forest Service, Clayton, NC 27520; and Associate Professor, North Carolina State University, Department of Forestry and Environmental Resources, Raleigh, NC 27607.

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Table 1--SEV, opportunity cost, and annual payments per acre by site index at base age 50 and pine straw revenue for forestry alternatives. Opportunity cost is the difference between the SEV of longleaf managed for habitat and the timber revenue alternatives. Annual payments assume a 10-year contract length

		SEV						
		Longleaf	Timber revenue		Opportunity costs		Annual payments	
Site Index	Pine straw	for habitat	Longleaf	Loblolly	Longleaf	Loblolly	Longleaf	Loblolly
60	None	-$497	-$140	$201	$357	$698	$42	$83
	Conservative	-$218	$104	$201	$322	$419	$38	$50
	Moderate	-$ 54	$303	$201	$357	$255	$42	$30
70	None	-$399	-$ 2	$201	$397	$600	$47	$71
	Conservative	-$125	$242	$201	$367	$326	$44	$39
	Moderate	$ 39	$440	$201	$401	$162	$48	$19
80	None	-$284	$142	$201	$426	$485	$51	$58
	Conservative	-$ 19	$386	$201	$405	$220	$48	$26
	Moderate	$145	$584	$201	$439	$ 56	$52	$ 7

calculated using these assumptions were significantly higher than any forestry scenario: $1,757 per acre for corn and $4,158 for soybeans. Marginal agricultural lands would be more attractive for conversion. An economic analysis of marginal farmland in North Carolina found that between 2007 and 2012 both corn and soybean crops generated negative returns, averaging a loss of $174 per year per acre for corn and $41 per year per acre for soybeans (Cubbage and others 2012). If these losses were repeated annually, they would lead to SEVs of -$4,524 to -$1,066 per acre.

The U.S. Fish and Wildlife and Wildlife Service (USFWS) estimates that each RCW cluster requires 75 to 100 acres of foraging range and 200 acres for breeding (USFWS 2003). We estimated a baseline range of costs to create enough habitat for a single cluster on already forested land: $24,150 to $43,900 per foraging group and $11,200 and $139,600 per breeding pair. This corresponds to the lower end of credit sales which have ranged from $100,000 to $250,000 per cluster (Bayon 2002). These estimates exclude all costs for biological monitoring though these costs may be significant. Annual payments were calculated assuming a 10-year contract. These ranged from $7 to $83 per year per acre and are consistent with payments delivered through existing State and federal forestry programs. However, significant work remains on the best mechanisms for implementation. Landowners near Camp Lejeune are more interested in short-term 10-year contracts that will not be consistent with the long-term goals of RCW habitat conservation (Rodriguez and others 2012).

Given recent concerns regarding future development in eastern North Carolina, Camp Lejeune has ample incentive to explore all options for balancing training needs with species conservation before more accessible and affordable options become unavailable. Under current timber market conditions, cooperative conservation through the direct payment to private landowners for RCW habitat creation and maintenance may provide one such option, especially given the local community's potential openness to such programs (Rodriguez and others 2012).

LITERATURE CITED

Bayon, R. 2002. A bull market in….woodpeckers? The Milken Institute Review. 1: 30-39.

Cubbage, F.W.; Glenn, J.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.

Dickens, E.D.; Moorhead, D.J.; Bargeron, C.T. [and others]. 2012. A summary of pine straw yields and economic benefits in loblolly, longleaf and slash pine stands. Agroforestry Systems. 86: 315-321.

Farrar, R. 1985. Volume and growth predictions for thinned even-aged natural longleaf pine stands in the East Gulf area. Res. Pap. SO-220. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 173 p.

Lohrey, R.E.; Bailey, R. 1977. Yield tables and stand structure for unthinned longleaf pine plantations in Louisiana and Texas. Res. Pap. SO-130. New Orleans: U.S.

Department of Agriculture Forest Service, Southern Forest Experiment Station. 52 p.

Marstel-Day, LLC. 2012. Comparative communities analysis. Task Order #10. Fredericksburg, VA: Marstel-Day. 232 p. http://www.planiteast.org/modules/evolvecms/upload/Comparative-Communities-Analysis.pdf. [Date accessed: May 12, 2013].

NCSU [North Carolina State University]. 2013. Corn-conventional till-NC, Coastal Plain, 2010. Raleigh, NC: NCSU, Department of Agricultural and Resource Economics. http://ag-econ.ncsu.edu/sites/ag-econ.ncsu.edu/files/extension/budgets/corn/Corn100-CT-CoastalPlain2010.pdf [Date accessed: November 25, 2013]

Rodriguez, S.L.; Peterson, M.N.; Cubbage, F.W. [and others]. 2012. Private landowner interest in market based conservation incentive programs for endangered species. Wildlife Society Bulletin. 36(3): 469-476.

Siry J.; Cubbage, F.W.; Malmquist, A. 2001. Growth, yield and returns for southern pine forestry investments. Forest Products Journal. 51(3): 42-48.

Smith, W.; Hafley, W. 1986. Natural stand growth and yield model (NATYIELD). Raleigh, NC: North Carolina State University. [Available only on North Carolina State University College of Natural Resources intranet server].

USFWS [U.S. Fish and Wildlife Service]. 2003. Recovery plan for the red-cockaded woodpecker (*Picoides borealis*): 2d revision. Atlanta, GA: U.S. Fish and Wildlife Service. 296 p.

FROM LOBLOLLY TO LONGLEAF: FIFTH-YEAR RESULTS OF A LONGLEAF PINE RESTORATION STUDY AT TWO ECOLOGICALLY DISTINCT SITES

Benjamin O. Knapp, G. Geoff Wang, Joan L. Walker, and Huifeng Hu[1]

Historical land-use and management practices in the southeastern United States have resulted in the widespread conversion of many upland sites from dominance of longleaf pine (*Pinus palustris* Mill.) to loblolly pine (*P. taeda* L.) in the time following European settlement. Given the ecological, economic, and cultural significance of the longleaf pine ecosystem, there is current interest in restoring longleaf pine and its associated plant communities on sites across the historical longleaf pine range. In many cases, this requires artificial regeneration to establish longleaf pine as a significant component of the regeneration cohort, particularly in stands that no longer include longleaf pine in the canopy as a seed source. However, other southern pines that commonly make up extant forest canopies can provide important ecosystem services during longleaf pine restoration, including wildlife habitat for species such as the red-cockaded woodpecker (*Picoides borealis*), needlefall for a continuous fuel bed important to fire management, and the suppression of the growth of hardwood competitors that can form an undesirable mid-story layer. Although longleaf pine is considered intolerant of competition and therefore traditional artificial regeneration practices typically include complete canopy removal, Kirkman and others (2007) proposed that retaining canopy trees in slash pine (*P. elliottii* Engelm.) stands during underplanting can help to reach restoration objectives despite an expected decrease in longleaf pine seedling growth. To date, no studies have been conducted to understand how alternative silvicultural practices may be used for longleaf pine restoration in existing loblolly pine forests. The objectives of this study were to determine: (1) the effects of canopy density on underplanted longleaf pine seedling growth and survival through five growing seasons; and (2) the effects of cultural treatments (herbicide and fertilizer) on underplanted longleaf pine seedling growth and survival through five growing seasons.

This study was replicated on two ecologically distinct study sites within the longleaf pine range: Fort Benning Military Installation in Georgia and Alabama and Marine Corps Base Camp Lejeune in North Carolina. At each site, we installed a replicated field experiment with a randomized complete block, split-plot design. Main-plot treatments included canopy manipulation to four levels of residual basal area: (1) Control [uncut, with basal area (BA) of 16 m^2/ha]; (2) medium BA (uniform thinning to BA of 9 m^2/ha); (3) low BA (uniform thinning to BA of 5 m^2/ha); and (4) Clearcut (basal area of 0 m^2/ha). Split-plot treatments included three levels of cultural treatment applied to improve longleaf pine seedling establishment: (1) NT (no treatment); (2) H (herbicide control of woody vegetation with a direct foliar application of 1 percent imazapyr to woody vegetation); and (3) H+F (the herbicide treatment combined with 280 kg/ha of 10-10-10 NPK fertilizer). Harvesting was completed in 2007, and longleaf pine seedlings were planted in January 2008. The herbicide split-plot treatment was applied in October 2008, and the fertilizer was applied in April 2009. In each 20- by 20-m split-plot measurement unit, we tagged a random selection of 30 longleaf pine seedlings in May 2008. We monitored seedling survival and measured root collar diameter and seedling height in October of 2008, 2009, 2010, and 2012. We considered seedlings to have emerged from the grass stage and entered active height growth if the terminal bud was ≥ 15 cm above the root collar.

[1]Assistant Professor, University of Missouri, School of Natural Resources, Columbia, MO 65211; Professor, Clemson University, Forestry and Natural Resources, Clemson, SC 29634; Research Plant Ecologist, USDA Forest Service, Southern Research Station, Clemson, SC 29634; and Associate Professor, Chinese Academy of Sciences, Institute of Botany, Beijing, China.

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Table 1--Effects of main-plot treatments and split-plot treatments on mean root collar diameter and percentage of seedlings in height growth five growing seasons after planting (with one standard error) at Fort Benning and Camp Lejeune. The same letter indicates no significant difference among levels within an effect for each site

		------------------Fort Benning------------------				-----------------Camp Lejeune-----------------			
		Root collar diameter		Height growth		Root collar diameter		Height growth	
Effect	Level	Mean	SE	Mean	SE	Mean	SE	Mean	SE
		----------*mm*---------		--------*percent*--------		---------*mm*---------		-------*percent*--------	
Main-plot treatment	Control	21.17C	1.08	2.97D	2.00	21.72B	1.04	13.32B	5.08
	Med BA	25.50BC	1.42	22.96C	5.92	30.01A	1.98	45.53A	9.73
	Low BA	31.78B	1.58	61.70B	7.18	30.37A	2.20	46.43A	9.76
	Clearcut	43.25A	3.05	86.61A	4.92	35.04A	4.07	56.88A	8.81
	p-value	< 0.0001		< 0.0001		0.0005		0.0002	
Split-plot treatment	NT	28.57	1.09	38.16	4.46	25.25B	1.46	28.96B	6.42
	H	31.07	1.67	46.21	4.90	32.37A	2.93	46.16A	10.36
	H+F	31.63	1.49	46.32	5.09	30.28A	1.73	47.3A	5.74
	p-value	0.0595		0.3170		< 0.0001		<0.0001	

Seedling survival displayed different patterns at the two study sites, with the lowest first-year survival in Clearcut plots at Fort Benning but the lowest survival in Control plots at Camp Lejeune (fig. 1). Although seedling survival was highest on uncut Control plots in the first year at Fort Benning, by the end of the fifth growing season the survival was similarly low on both Clearcut plots and Control plots. Results from previous studies have also suggested that early survival of planted longleaf pine seedlings is higher beneath canopy trees than in canopy openings during harsh conditions associated with drought (Gagnon and others 2003, McGuire and others 2001, Rodríguez-Trejo and others 2003). Our results suggest that facilitation effects of canopy trees on longleaf pine seedling survival may not persist over time if canopy density is high. The contrasting results of our two study sites also suggest that site-specific growing conditions can affect the impacts of canopy retention on planted longleaf pine seedling survival.

At both sites, seedling growth was lowest on the uncut Control plots, but an incremental pattern of growth increase with greater canopy removal was observed at Fort Benning but not at Camp Lejeune (table 1). In longleaf pine forests, canopy retention has been found to reduce seedling growth (Kirkman and Mitchell 2006, Palik and others 1997), and our results support this general finding in loblolly pine forests.

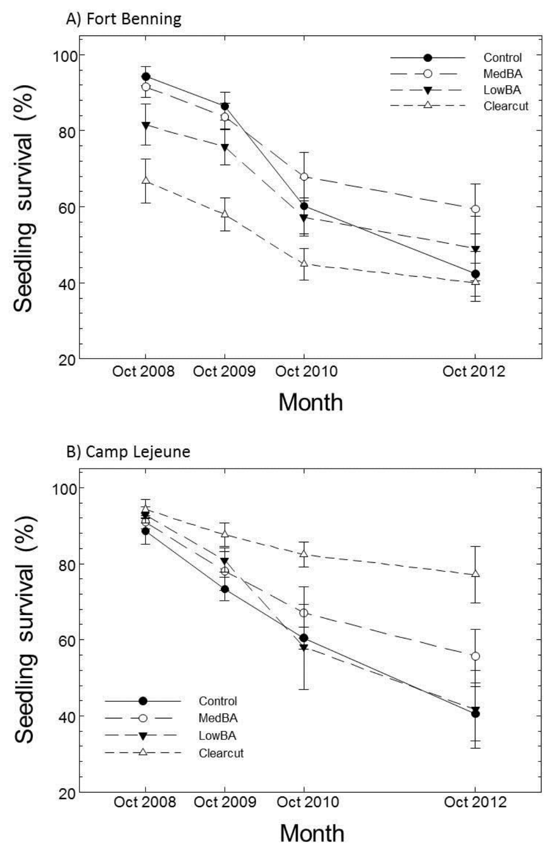

Figure 1--Planted longleaf pine seedling survival (mean ± one standard error) at the end of the first, second, third, and fifth growing seasons at: (A) Fort Benning and (B) Camp Lejeune.

However, the Clearcut plots at Fort Benning resulted in a higher percentage of seedlings out of the grass stage than that seen at Camp Lejeune. The herbicide split-plot treatments increased seedling growth at Camp Lejeune but not at Fort Benning, and Camp Lejeune generally had more abundant sub-canopy vegetation (data not shown) that our data suggest provided enough competition to limit seedling growth. We found no differences between the H and H+F treatments, suggesting that fertilizer did not improve seedling growth at either site. Generally, our results indicate that site-specific growing conditions must be considered when determining appropriate silvicultural prescriptions for restoring longleaf pine in loblolly pine stands.

LITERATURE CITED

Gagnon, J.L.; Jokela, E.J.; Moser, W.K.; Huber, D.A. 2003. Dynamics of artificial regeneration in gaps within a longleaf pine flatwoods ecosystem. Forest Ecology and Management. 172: 133-144.

Kirkman, L.K.; Mitchell, R.J. 2006. Conservation management of *Pinus palustris* ecosystems from a landscape perspective. Applied Vegetation Science. 9: 67-74.

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.

McGuire, J.P.; Mitchell, R.J.; Moser, E.B. [and others]. 2001. Gaps in a gappy forest: plant resources, longleaf pine regeneration, and understory response to tree removal in longleaf pine savannas. Canadian Journal of Forest Research. 31: 765-778.

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.

Rodríguez-Trejo, D.A.; Duryea, M.L.; White, T.L. [and others]. 2003. Artificially regenerating longleaf pine in canopy gaps: initial survival and growth during a year of drought. Forest Ecology and Management. 180: 25-36.

WHAT 45 YEARS OF RLGS DATA HAS TO SAY ABOUT LONGLEAF PINE MORTALITY – NOT MUCH

John S. Kush, John C. Gilbert, and Rebecca J. Barlow[1]

Abstract--The original longleaf pine (*Pinus palustris* Mill.) forest was self-perpetuating where seedlings always had to be present. It reproduced itself in openings in the overstory where dense young stands developed. These openings would range from a few tenths of an acre to large openings of several thousand acres. Regardless of the event size, longleaf pine was able to regenerate these openings. In 1964, the USDA Forest Service established the Regional Longleaf Pine Growth Study (RLGS) in the Gulf States. 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. The study has been expanded over the decades to examine numerous aspects of longleaf pine stand dynamics. Landowners who have stands of large/old longleaf pine trees have fears they will lose them to some type of mortality before they can harvest them. For over 4 decades, the amount of mortality and its cause have been documented for the trees in the RLGS. One of the major causes of mortality has been suppression that happens among the smallest trees or trees that have been over-topped for several years. At the other end of the tree size class, larger trees are killed by lightning strikes. These events are low level and happen at low frequencies every year. Surprisingly, despite being among the most fire-adapted tree species in nature, fire can kill longleaf pine. So what kills longleaf pine trees and how often does it happen?

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.) dominated much of the south Atlantic forest in the late 1800s to early 1900s (Ashe 1894, Harper 1928, Mohr 1897, Sargent 1884). This forest was self-perpetuating where seedlings always had to be present. It reproduced itself in openings in the overstory where young stands developed. These openings would have ranged from a few tenths of an acre due to the loss of a single tree to a lightning strike or windfall, a few acres due to insects or a larger-scale wind event, to large openings of several thousands of acres due to tornados or hurricanes. Regardless of the event size, longleaf pine trees were able to regenerate these openings. The result was a park-like, uneven-aged forest, composed of many even-aged stands of varying sizes (Schwarz 1907).

The USDA Forest Service and academia recognized the importance of longleaf pine. Reed (1905) published an inventory of large parcels of forest in central Alabama where he found longleaf pine dominating nearly 80 percent of the area. Schwarz (1907) wrote one of the first books about a U.S. trees species. H. H. Chapman a professor of forestry at Yale University, who spent summers doing research in Mississippi and Louisiana, published several papers between 1907 and the mid-1930s (Chapman 1909, 1912, 1923, 1926, 1932). Research papers discussing longleaf pine regularly appeared in the Journal of Forestry, Ecology, and Ecological Monographs.

The mid-20th century brought about industrial forestry and plantation management. Longleaf pine acreage was decreasing rapidly, and there was very little regard for regeneration. Wahlenberg (1946), in his landmark text, devoted three chapters to the topic of longleaf pine regeneration, nearly one quarter of the book. In his introduction, he stated:

> “Where formerly it had complete possession of the land, it has often failed to reproduce; this failure has resulted in deterioration of land values in many localities.”

The two major problems he identified for the frequent failure were: (1) fire, either too frequent and killing recent regeneration or too infrequent and allowing competitors to thrive; and (2) logging practices that left little or nothing on the ground or no seed trees. He summed this up by saying:

> “Mismanagement of longleaf pine has been the rule rather than the exception, due to ignorance of the unique life history and incomplete knowledge of factors

[1]Research Fellow and Research Associate, respectively, Auburn University, School of Forestry and Wildlife Sciences, Auburn University, AL 36849; and Associate Extension Professor, Auburn University, Alabama Cooperative Extension Service, School of Forestry and Wildlife Sciences, Auburn University, AL 36849.

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determining the life and death of seedlings and hence the succession of forest types."

Despite the renewed interest in longleaf pine, such as the effort of America's Longleaf Initiative (2009), we continue to lose the best-quality longleaf pine stands in structure and ground cover through the loss of natural stands on privately-held lands. Many landowners are fearful of losing their larger, older trees to lightning or some other catastrophic event. We need to maintain what existing stands are left, so it is important we get information to the people who have longleaf pine to educate them on how to maintain it and their options for the future. Do landowners and land managers need to worry about losing their longleaf pine? Some of the answers lie in a long-term study that has been documenting the cause of mortality in naturally regenerated longleaf pine stands across the Gulf Coast states. What kills longleaf pine and how often does it happen? This paper examines nearly 50 years of data to address these questions.

METHODS

In 1964, the Forest Service established the Regional Longleaf Growth Study (RLGS) in the east Gulf Region (Farrar 1978). 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 consists of 305 permanent one-tenth and one-fifth acre measurement plots located in central and southern Alabama, southern Mississippi, southwest Georgia, northern Florida, and the sandhills of North Carolina. Plot selection was based upon a rectangular distribution of cells formed by six stand age classes ranging from 20 to 120 years, five site-index classes ranging from 50 to 90 feet at 50 years, and five density classes ranging from 30 to 150 square feet per acre. Several plots have been left unthinned to follow stand development over time.

Within this distribution are five time replications of the youngest age class. All these replications are located on the Escambia Experimental Forest in Brewton, AL. As a part of the RLGS, plots in the youngest age class were first established in 1964, and new sets of plots have been added in this age-class every 10 years. Plots are located to achieve similar initial site qualities and ages and are thinned to their target basal areas.

At the time of establishment, plots are assigned a target basal area class of 30, 60, 90, 120, or 150 square feet per acre. They are left unthinned to grow into that class if they are initially below the target basal area. In subsequent re-measurements, the plot is thinned back to the previously assigned target if the plot basal area has grown 7.5 square feet per acre or more beyond the target basal area. The thinnings are generally of low intensity and from below.

Net (measurement) plots are circular and one-fifth acre (14 net plots are one-tenth acre) in size surrounded by a similar and like-treated ½-chain-wide isolation strip with both surrounded by a ½-chain-wide protective buffer strip that receives extensive management. The measurements are made during the dormant season (October through March), and it takes 3 years to complete a full re-measurement of all plots. Each tree on the net plot with a d.b.h. (diameter at breast height) > 0.5 inches is numbered by progressive azimuth from magnetic north and has its azimuth and distance from plot center recorded. A systematic sub-sample of trees from each 1-inch d.b.h. class has been permanently selected and measured for height to the live-crown base, total height, and, if the tree is dominant or co-dominant, for age from seed.

The RLGS is re-inventoried every 5 years and is now in its 45th year re-measurement. At every re-measurement, each tree has its d.b.h. recorded to the nearest 0.1 inch, and crown class, utility pole class and length determined. Height measurements are made on the selected subsample of trees. In addition to the measurements made on living trees, if a tree dies, the cause of death is recorded. The causes of death are: lightning, insects, disease, wind, fire, suppression, and unknown.

Before getting into the results, a few caveats must be presented. The 5 years between re-measurements can make it difficult at times to determine the exact cause of death. If a tree died due to suppression and the stand was burned twice between the re-measurements, we may not find a "skeleton" despite having the azimuth and distance of the tree from plot center. A second factor is the thinnings conducted to maintain the plots at their assigned basal areas. These thinnings are from below and may remove trees that would have otherwise died due to some other cause. A final factor is the density on many of the RLGS plots. The "average" RLGS plot has 386 trees per acre with a range from 15 to over 1,800 trees per acre. The higher-density plots, which contain over 900 trees per acre, are outside the range of "normal" longleaf pine management.

RESULTS

Overall Mortality

Across the timespan of the RLGS, 1.3 trees per acre per year die. The major cause of death within the RLGS is suppression. Nearly 54 percent of the trees die due to being beneath the overstory canopy. Fire is the second highest cause of death with nearly 19 percent. Wind has killed 13 percent of the trees that have died. The remaining causes are insects, just over 7 percent, unknown with nearly 5 percent, and both disease and lightning causing just over 1 percent.

Suppression

Nearly six trees per acre per year were lost to suppression making it the major cause of mortality in the RLGS. In part, this was due to the density on many plots being well above those of most managed longleaf pine stands. In addition, some of the trees were removed because the plots were thinned when they exceeded their assigned basal area. These thinnings were from below and may have removed trees that ultimately would have died due to suppression. Nearly 90 percent of those trees dying because of suppression occurred on plots < 40 years old or on plots with > 1,200 stems per acre. Older trees on low-density plots did not die of suppression. However, regardless of age or site, suppression was a cause of death on plots with more than 140 square feet per acre of basal area. For the next portion of the paper, suppression was removed from the analyses because of its overwhelming impact.

In addition to the continual loss of trees each year due to suppression, there were waves of mortality about every 30 years where nearly 10 percent of the trees classified as suppressed died. While the density for the plot dropped noticeably, the loss of basal area or volume was minimal. The loss of these trees freed growing space and resources for the remaining trees and increased their growth. The first mortality wave happened when the trees were 30 to 35 years old and waves were seen again at ages 60 to 65 and 90 to 95. The lower the site index the longer it took for the wave of suppression to occur.

Lightning

The cause of tree death feared by many landowners, lightning, resulted in mortality of < one tree per acre per year. However, those trees hit by lightning were large, with over 95 percent of them classified as dominant trees on the plot; in other words, they served as lightning rods. Nearly 82 percent of this mortality happened on plots with < 60 square feet per acre of basal area (or 45 trees per acre).

Insects

Insects killed a little more than one tree per acre per year. Nearly 95 percent of the deaths due to insects were related to a lightning strike. Frequently, *Ips* beetles were attracted to the stressed tree, and the tree died due to the insect activity. In addition to the lightning-struck tree dying, a few trees around it that may have been connected through their roots were attacked by the beetles and died.

Wind

Mortality due to wind was very episodic and quite often is large-scale event. A little more than 1.5 trees per acre per year were lost to wind damage. Kush and Gilbert (2010) provided a description of the mortality to wind in their examination of the RLGS data after Hurricane Ivan. Trees growing in open conditions, adjacent to roads and fields, or recently released from a thinning operation were more susceptible to blow-down or breakage than were trees in denser stands.

Fire

Yes, fire killed longleaf pine and did so at a rate of nearly three trees per acre per year. The RLGS plots were supposed to be prescribe-burned every 3 years with cool season (winter) burns. Every now and then weather conditions (especially wind direction) changed, and fires became hotter than intended. There was no fire-related mortality on plots with a basal area < 45 square feet per acre.

Disease

The loss due to disease is very small, < one tree per acre per year. It is very similar to the mortality due to insects because like most trees that have been listed as dying due to disease, the tree had fusiform rust, and it snapped off at the cankered location.

Unknown
With 5 years between re-measurements, it was occasionally difficult to determine the cause of death. It appeared as if much of this mortality would have been due to suppression since quite often the tree could not be found.

Age Class Mortality
The RLGS data set was separated into 40-year age groupings to examine tree mortality. For trees < 40 years old, fire and wind were the major reasons for mortality. Lightning mortality increased with tree age. Insect mortality was higher in the younger age classes due to its relationship with fire mortality.

Density Class Mortality
Density was examined by separating the RLGS into basal area classes of < 45, 45 to 90, and > 90 square feet per acre. The major cause of mortality on the lower-density plots was due to wind, secondly to lightning. At the higher density level, fire was the major cause of mortality due to higher fuel loading.

Ten-year Periods and Mortality
The RLGS was broken into 10-year periods, 1964-1973, 1973-1983, 1994-2003, and 2004-present to see if there were any trends in mortality. There was no trend with lightning; in fact, mortality has dropped very slightly over time. Insect mortality was related to stressor events of wind and fire. Wind mortality was episodic and occurred with Hurricanes Fredrick, Opal, and Ivan. Trees dying from fire were found on young plots, < 30 year old, that burned hotter than intended.

DISCUSSION and CONCLUSION

A very high percentage of the mortality occurring on the RLGS plots over the past 45 years was related to some type of episodic event. There is nothing that can be done to prevent mortality from hurricanes and tornados. At best, from a management perspective, stands should not be opened up dramatically. The sudden release from surrounding and supporting trees leave the trees more susceptible to loss from a wind event. Losses from fire can be prevented or minimized by understanding fuel and day-of-burn conditions.

Lightning was not striking all of the trees older than 80-years old. In fact, the loss due to lightning strikes decreased over time since the mid-1960s. Landowners need not fear the loss of these valuable trees.

The one area where landowners and land managers can control mortality in longleaf pine stand is to minimize the loss due to suppression. Longleaf pine stands are very well-suited to frequent thinning as reported by Lauer and Kush (2011). These frequent thinnings can remove the potential suppression mortality before it happens by being pro-active instead of reactive. Nature managed longleaf pine by taking out a tree or two on a yearly basis to create openings that were then filled with regeneration. Why do we not manage longleaf the way nature did?

While much research focus shifted to loblolly and slash pine, this study survived despite threats from conversion, lack of funding, and natural catastrophic events such as tornados and hurricanes. Over the past nearly 50 years, the RLGS has provided much information for landowners and land managers. The RLGS is and has been an incredible resource and an underutilized wealth of information and knowledge about longleaf pine ecology and management. While not perfect, and no study is, the RLGS has data that cannot be found in any other study due to the combination of different age, site, and density classes. Hopefully, the RLGS will continue another 50 years, and it still will be just a small portion in the lifespan of one of these trees.

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LITERATURE CITED

America's Longleaf Initiative. 2009. http://www.americaslongleaf.net/initiative/goals. [Date accessed: May 17, 2011].

Ashe, W.W. 1894. The longleaf pine and its struggle for existence. Journal of the Elisha Mitchell Science Society. 11: 1-16.

Chapman, H.H. 1909. A method of studying growth and yield of longleaf pine applied in Tyler County, Texas. Proceedings of the Society of American Forestry. 4: 207-220.

Chapman, H.H. 1912. Forest fires and forestry in the southern states. American Forestry. 18: 510-517.

Chapman, H.H. 1923. The causes and rate of decadence in stands of virgin longleaf pine. Lumber Trade Journal. 84 (6): 11: 16-17.

Chapman, H.H. 1926. Factors determining natural reproduction of longleaf pine on cut-over lands in LaSalle Parish, Louisiana. Bull. 16. New Haven, CT: Yale University School of Forestry. 44 p.

Chapman, H.H. 1932. Is the longleaf type a climax? Ecology. 13: 328–334

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.

Harper, R.M. 1928. Economic botany of Alabama, part 2. Catalogue of the trees, shrubs, and vines of Alabama, with their economic properties and local distribution. Geological Survey of Alabama. Monogr. 9. University, AL: Birmingham Printing Company. 357 p.

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.

Lauer, D.K.; Kush, J.S. 2011. A variable density stand level growth and yield model for even-aged natural longleaf pine. Special Rep. No. 10. Auburn, AL: Auburn University, Alabama Agricultural Experiment Station. http://www.aaes.auburn.edu/comm/pubs/specialreports/sr10-longleaf-pine.pdf. [Date accessed: December 18, 2013].

Mohr, C.T. 1897. The timber pines of the southern United States. Bull. 13, revised. Washington, DC: U.S. Department of Agriculture Division of Forestry. 176 p.

Reed, F.W. 1905. A working plan for forest lands in central Alabama. Bull. 68. Washington, DC: U.S. Department of Agriculture Forest Service. 71 p.

Sargent, C.S. 1884. Report on the forests of North America (exclusive of Mexico). 10th Census Rep. Vol. 9. Washington, DC: U.S. Department of Interior Census Office. 612 p.

Schwarz, G.F. 1907. The longleaf pine in virgin forest. A silvical study. New York: John Wiley & Sons. 135 p.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth and management. 1st ed. Washington, DC: Charles Lathrop Pack Forestry Foundation. 429 p. In cooperation with: The U.S. Department of Agriculture Forest Service.

EFFECTS OF PRECOMMERCIAL THINNING AND MIDSTORY CONTROL ON AVIAN AND SMALL MAMMAL COMMUNITIES DURING LONGLEAF PINE SAVANNA RESTORATION

Vanessa R. Lane, Robert P. Simmons, Kristina J. Brunjes, John C. Kilgo, Timothy B. Harrington, Richard F. Daniels, W. Mark Ford, and Karl V. Miller[1]

Abstract--Restoring longleaf pine (*Pinus palustris* Mill.) savanna is a goal of many southern land managers, and longleaf plantations may provide a mechanism for savanna restoration. However, the effects of silvicultural treatments used in the management of longleaf pine plantations on wildlife communities are relatively unknown. Beginning in 1994, we examined effects of longleaf pine restoration with plantation silviculture on avian and small mammal communities using four treatments in four 8- to 11-year-old plantations within the Savannah River Site in South Carolina. Treatments included prescribed burning every 3 to 5 years, plus: (1) no additional treatment (burn-only control); (2) precommercial thinning; (3) non-pine woody control with herbicides; and (4) combined thinning and woody control. We surveyed birds (1996-2003) using 50-m point counts and small mammals with removal trapping. Thinning and woody control alone had short-lived effects on avian communities, and the combination treatment increased avian parameters over the burn-only control in all years. Small mammal abundance showed similar trends as avian abundance for all three treatments when compared with the burn-only control, but only for 2 years post-treatment. Both avian and small mammal communities were temporarily enhanced by controlling woody vegetation with chemicals in addition to prescribed fire and thinning. Therefore, precommercial thinning in longleaf plantations, particularly when combined with woody control and prescribed fire, may benefit early-successional avian and small mammal communities by developing stand conditions more typical of natural longleaf stands maintained by periodic fire.

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.) was once the dominant forest type across much of the southeastern United States. Longleaf pine's historic range encompassed most of the Atlantic and Gulf Coastal Plains from southeastern Virginia to eastern Texas and included part of the Piedmont and Ridge and Valley physiographic provinces of Alabama and Georgia (Simberloff 1993). Today less than 3 percent of the estimated 37 million ha of longleaf that existed prior to European colonization remain, and much of the remainder is in a degraded condition (Frost 1993). Losses of moist tropical rainforest worldwide amount to 40 percent of that ecosystem in comparison to the loss of 97 percent of the historic longleaf ecosystem, making the longleaf pine ecosystem critically endangered (Noss 1989, Ware and others 1993).

Historically, the longleaf ecosystem occupied a wide variety of site types, and the structure and composition of the vegetative communities varied greatly across this site gradient (Peet and Allard 1993). Commonalities among these communities include an overstory dominated by longleaf pine, lack of midstory hardwoods, and rich and diverse herbaceous ground cover. The longleaf ecosystem supports some of the most diverse vegetative and faunal communities in the temperate zone, including many endemic species (Peet and Allard 1993, Simberloff 1993).

The longleaf pine ecosystem depends on disturbance, particularly frequent low-intensity fires (Ware and others 1993). Early-successional plant and animal communities in longleaf pine ecosystems rely upon periodic fire to persist, particularly endemic species such as red-cockaded woodpecker (*Picoides borealis*) and Bachman's sparrow (*Aimophila aestivalis*) (Conner and others 2001, Kilgo and Blake 2005, Plentovich and others 1998). In general, early-successional plant and animal communities associated with pine forests decline with the establishment of midstory woody species, which eventually leads to crown closure and understory shading (Atkeson and Johnson 1979, Lane and others 2011). Without fire, the southern pine ecosystems succeed to other forest types, often the southern mixed hardwood

[1]Lecturer, University of Minnesota-Crookston, Fisheries and Wildlife Management, Crookston, MN 56716; Editor, TimberMart South, University of Georgia, Athens, GA 30602; Big Game Coordinator, Kentucky Department of Fish and Wildlife Resources, Division of Wildlife, Frankfort, KY 40601; Research Wildlife Biologist, USDA Forest Service, Southern Research Station, New Ellenton, SC 29809; Research Scientist, USDA Forest Service, Pacific Northwest Research Station, Olympia, WA 98512; Professor, University of Georgia, Warnell School of Forestry and Natural Resources, Athens, GA 30602; Unit Leader, USGS Virginia Cooperative Fish and Wildlife Research Unit, Virginia Polytechnic Institute and State University, Department of Fish and Wildlife Conservation, Blacksburg, VA 24060; and Professor, University of Georgia, Warnell School of Forestry and Natural Resources, Athens, GA 30602.

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forest (Frost 1993, Landers and others 1995, Ware and others 1993). . Unfortunately, early-successional forest habitats are in decline throughout the United States due to the removal of natural disturbances (notably fire) which slow or prevent early-successional habitats from succeeding to climax communities (Trani and others 2001). Pine plantation management frequently seeks to create early-successional habitat via treatments such as precommercial or commercial thinning and understory woody plant control, especially in situations where hunting leases generate additional income or restoration of native pine communities is a management goal (Hedman and others 2000, Stroh and others 2002).

During the last 2 decades, interest has increased to restore longleaf pine savanna on appropriate sites throughout its historical range (McMahon and others 1998). Because there is often an insufficient seed source to regenerate these areas naturally, plantation silviculture has been suggested as a means of restoring this species and associated ecosystems (Harrington and Edwards 1999, Landers and others 1995). Although the floral and faunal characteristics of natural longleaf forests have been well documented (Peet and Allard 1993, Ware and others 1993), the effects of longleaf plantation silviculture on plant and wildlife communities are less well understood (Repenning and Labisky 1985). Loblolly pine (*Pinus taeda* L.) plantations can support early-successional plant and animal communities through a combination of site preparation methods, herbaceous and/or woody control, and thinnings (Krementz and Christie 2000, Lane and others 2011), but the characteristics and persistence of these communities in longleaf plantations are not fully understood. Therefore, we examined the effects of thinning and woody competition control on avian and small mammal communities in longleaf pine plantations through 10 years post-treatment in the Upper Coastal Plain of South Carolina.

MATERIALS AND METHODS

We established a long-term study to assess longleaf pine ecosystem restoration techniques using plantation silviculture at the Savannah River Site (SRS), a National Environmental Research Park of the U.S. Department of Energy near Aiken, SC (Harrington and Edwards 1999). The study was conducted in the Sandhills physiographic province of South Carolina (Miller and Robinson 1995), similar to work initiated by Brunjes and others (2003). During the winter of 1993-1994, we selected four longleaf pine plantations established between 1982 and 1986. We selected sites that contained fully stocked stands of longleaf pine (> 1,200 stems/ha) and hardwoods (> 600 stems/ha). Sites ranged from 17.4 to 20.6 ha. Each plantation had been established by machine planting 1-year-old bare-root seedlings at 1.8- by 3-m spacing in clearcut-harvested areas in which woody debris had been windrowed or piled, then burned. The sites supported mature stands of naturally regenerated longleaf and loblolly pines prior to longleaf plantation establishment. The study sites represent a range of moisture classifications from xeric to moderately mesic (Van Lear and Jones 1987). Soils are loamy sands, which range from well-drained to excessively well-drained (Rogers 1990).

The USDA Forest Service, Savannah River, applied a prescribed fire of moderate to high intensity to each site in February 1994, which top-killed all shrubs and most hardwoods < 5 cm d.b.h. Similar prescribed fires were applied to all sites in February 1998 and January-February 2003.

Each site was divided into four treatment areas of similar size at the initiation of the study, burned every 3 to 5 years with prescribed fire, and randomly assigned one of the following treatments to each: (1) nontreated: no treatments applied other than prescribed fire; (2) pine thinning: in May 1994, we thinned the pines to leave a uniform spacing of trees at approximately half the original stem density, resulting in 635 and 1,440 pine trees/ha for thinned and unthinned plots, respectively. We cut the trees with a brush saw and left them to decay, resulting in minimal litter and soil disturbance; (3) non-pine woody control: in April 1995, we applied undiluted Velpar® L (hexazinone, E.I. du Pont de Nemours and Company, Wilmington, DE) at a rate of 1.7 kg active ingredient/ha with a spotgun to grid points on approximately 1-m spacing. In March 1996, we targeted surviving non-pine stems with a basal spray of Garlon® 4 (triclopyr ester, Dow AgroSciences LLC, Indianapolis, IN) at 7 percent concentration in oil. In late June 1996, we applied a directed foliar spray of Arsenal® AC (imazapyr, American Cyanamid Company, Wayne, NJ), Accord® (glyphosate, Monsanto Company, St. Louis, MO), and X-77® surfactant

(Loveland Industries, Inc, Greeley, CO) mixed in water at 0.5, 5, and 0.5 percent concentrations, respectively, to surviving target vegetation within 8 m of each sample point (described below). We applied all herbicides with a backpack sprayer and left vegetation standing; and (4) combined treatment: we combined pine thinning with woody control.

We used a randomized complete block design with four blocks, each with a 2 x 2 factorial arrangement of treatments. These treatment plots were the experimental units. Within each of the 16 treatment plots, we permanently marked 10 small mammal sampling points on a 40-m grid and 1 avian sample point near the center of each plot for repeated measurements. We hypothesized that both silvicultural treatments would increase the abundance of early-successional birds and small mammals and decrease the abundance of mature forest wildlife. We further hypothesized that the duration of any effects noted would be relatively short when treatments were applied separately but that combining the treatments would extend the duration of the effects.

Avian Sampling

We surveyed the breeding bird community using 50-m fixed-radius point counts within the first 4 hours following sunrise. We performed five counts at the permanent avian sample points in each treatment area in April-June 1996 and 1997 and in May 2001-2003. During each 5-minute count, we recorded all birds seen or heard within 50 m of the point. We calculated the mean number of individuals of each species encountered on each treatment area by year. We also calculated Shannon H´ diversity and richness of these bird communities (Ricklefs 1997).

Small Mammal Sampling

We surveyed small mammal populations by removal trapping at two sites during 1996-1997 and surveyed two additional sites (four total) during 2001-2004. We placed one Victor® (Woodstream Corporation, Lititz, PA) rat-trap 4 m north or south of each of the 10 sample points per treatment area and placed one Victor® mousetrap opposite the rat-trap, 4 m from the sample point. We baited traps daily with peanut butter and oatmeal. We trapped the original two sites in April 1996, December 1996, and April 1997, and all four sites in May 2001, May 2002, December 2002, May 2003, and December 2003. We surveyed all sites simultaneously. Each trapping period consisted of four consecutive nights, and captured animals were identified using morphological characteristics (Cothran and others 1991). *Peromyscus leucopus* Raf. and *P. gossypinus* Le Conte were difficult to differentiate morphologically (Burt and Grossenheider 1976, Cothran and others 1991). Although Cothran and others (1991) reported that only two *P. leucopus* have been found from the SRS, we combined *Peromyscus leucopus* and *P. gossypinus* into a *Peromyscus* spp. category.

Vegetation Sampling

We sampled vegetation characteristics to explain potential differences in avian and small mammal communities among treatments. In winter 1994-1995, 1995-1996, 1997-1998 and 2002-2003, we measured diameter at breast height (d.b.h.) of each hardwood and pine tree rooted within 6 m of each small mammal sampling point and measured total height, height to the base of the live crown (HBLC), and crown width (CW) of 20 percent of randomly selected surrounding stems.

We recorded each understory species rooted within 3.6 m of each small mammal sampling point in August 1994-1996. We estimated percent ground cover of each species and woody debris at each sample point using the line-intercept method (Mueller-Dombois and Ellenberg 1974). Understory plant cover data were grouped into categories of forbs, grasses, vines, shrubs, or tree seedling according to Radford and others (1968). In 1998, 2001, and after the fire in 2003, we employed sampling protocols developed for the North Carolina Vegetation Survey (Peet and others 1996) to provide more comprehensive estimates of herbaceous species density and understory cover. At each odd-numbered sample point (120 total), we located nested square subplots of 0.01, 0.1, 1, 10, and 100 m^2 with their diagonal overlaid onto the original vegetation transect. We generated a list of understory species rooted within each subplot. We visually assessed species percent within the 10-m^2 subplot using the following cover classes and assigned class midpoint values: trace (class midpoint 0.1 percent), 0-1, 1-2, 2-5, 5-10, 10-25, 25-50, 50-75, 75-95, 95-100 percent. All values were averaged by vegetation category to provide one estimate per category for each experimental unit to correlate with avian and small mammal metrics.

Analysis

We analyzed treatment effects as a 2 x 2 factorial design with repeated measurements for each response variable (mean avian abundance, avian species richness, avian Shannon H´ diversity, and mean small mammal capture rates). Square root transformations of bird abundance data were used to satisfy the normality assumptions of analysis of variance (ANOVA). Once transformed, the residuals from all avian analyses were approximately normal. When residuals for small mammals were severely non-normal, we transformed the data to improve the distribution of the residuals. If standard transformations did not normalize the residuals, we used the rank transformation approach of Conover and Iman (1981) for small mammal data. We ranked treatment area means within each trapping period, assigning average rank to ties. ANOVA performed on ranks was a non-parametric test that retained the advantages of the full experimental design (Conover and Iman 1981). We used SAS® System version 8.02 (SAS Institute, Cary, NC) for all analyses, and used GLM, MIXED, and CORR procedures to perform ANOVA and correlations procedures, respectively. We considered significance at α = 0.10 for all analyses.

RESULTS

Avian Community Responses

We analyzed 80 point counts from each year that we surveyed: 1996, 1997, 2001, 2002, and 2003. Overall, we detected 746 birds of 41 species, including 28 residents (including short distance migrants) and 13 neotropical migrants (table 1). We detected 133 individuals of 25 species in 1996, 99 individuals of 18 species in 1997, 132 individuals of 23 species in 2001, 258 individuals of 25 species in 2002, and 124 individuals of 24 species in 2003. Sampling year explained variability in all of our habitat and migration strategy groupings ($F_{4,36} \geq 2.23$, $P \leq 0.09$).

Total avian abundance was affected by the interaction of thinning and midstory-control ($F_{1,9} = 9.71$, $P = 0.01$). Total avian abundance was greatest in the combined treatment in 1996 and 2001-2003, while total avian abundance was similar among all other treatments (fig. 1). Total avian abundance was weakly correlated with sampling year (Pearson's r = 0.26, $P = 0.02$), number of years since the last burn (r = 0.26, $P = 0.02$), mean pine d.b.h. (r = 0.26, $P = 0.02$), and mean pine crown width (r = 0.25, $P = 0.02$). Total avian abundance was weakly negatively correlated with mean density of pine trees/ha (r = -0.28, $P = 0.01$) and the mean density of hardwood trees/ha (r = -0.21, $P = 0.07$).

Avian species richness and diversity were not affected by midstory-control alone ($F_{1,9} \leq 2.39$, $P \geq 0.16$) but were affected by thinning alone ($F_{1,9} \geq 6.94$, $P \leq 0.03$) and by the interaction of thinning and midstory-control ($F_{1,9} \geq 3.72$, $P \leq 0.09$). Avian species richness and diversity were greatest in the combined treatment in all years except 2003, when values on all treated plots were less than the control (fig. 1). Thinned plots also contained greater avian species richness and diversity than midstory-control alone and the control in most years except 2003 (fig. 1).

Small Mammal Community Responses

We captured 211 mammals of eight species during 8,320 trap nights (table 2). *Peromyscus* species other than *P. polionotus* accounted for 64 percent of all captures. Oldfield mice (*P. polionotus*) comprised 18 percent of captures, and eastern woodrats (*Neotoma floridana*) comprised 8 percent. We also captured cotton rats (*Sigmodon hispidus*, 6 percent), golden mice (*Ochrotomys nuttalli*, 3 percent), pine voles (*Microtus pinetorum*, 1 percent), and eastern harvest mice (*Reithrodontomys humulis*,

Table 1--Common and scientific names, migratory strategy, and habitat association of bird species encountered during the breeding season in longleaf pine plantations at the Savannah River Site, South Carolina, in 1996, 1997, and 2001-2003. Species are classified as year-round residents (R), short distance migrants (SD), or neotropical migrants (T)

Common name	Scientific name	Migratory strategy
American crow	*Corvus brachyrhynchos*	SD
American goldfinch	*Carduelis tristis*	SD
Black-and-white warbler	*Mniotilta varia*	T
Blue jay	*Cyanocitta cristata*	SD
Blue-gray gnatcatcher	*Polioptila caerulea*	SD
Brown thrasher	*Toxostoma rufum*	SD
Brown-headed nuthatch	*Sitta pusilla*	R
Carolina chickadee	*Poecile carolinensis*	R
Carolina wren	*Thryothorus ludovicianus*	R
Downy woodpecker	*Picoides pubescens*	R
Eastern bluebird	*Sialis sialis*	SD
Eastern towhee	*Pipilo erythrophthalmus*	SD
Eastern wood-pewee	*Contopus virens*	T
Field sparrow	*Spizella pusilla*	SD
Golden-crowned kinglet	*Regulus satrapa*	SD
Gray catbird	*Dumetella carolinensis*	T
Great crested flycatcher	*Myiarchus crinitus*	T
Hairy woodpecker	*Picoides villosus*	R
Indigo bunting	*Passerina cyanea*	T
Mourning dove	*Zenaida macroura*	SD
Northern bobwhite	*Colinus virginianus*	R
Northern cardinal	*Cardinalis cardinalis*	R
Northern flicker	*Colaptes auratus*	SD
Northern parula	*Parula Americana*	T
Ovenbird	*Seirus aurocapillus*	T
Pileated woodpecker	*Dryocopus pileatus*	R
Pine warbler	*Setophaga pinus*	SD
Prairie warbler	*Setophaga discolor*	T
Red-bellied woodpecker	*Melanerpes carolinus*	R
Red-eyed vireo	*Vireo olivaceus*	T
Red-shouldered hawk	*Buteo lineatus*	SD
Red-tailed hawk	*Buteo jamaicensis*	SD
Song sparrow	*Melospiza melodia*	SD
Summer tanager	*Piranga rubra*	T
Tufted Titmouse	*Baeolophus bicolor*	R
White-eyed vireo	*Vireo griseus*	SD
Wild turkey	*Meleagris gallopavo*	R
Wood thrush	*Hylocichla mustelina*	T
Yellow-breasted chat	*Iceria virens*	T
Yellow-rumped warbler	*Setophaga coronata*	SD
Yellow-throated vireo	*Vireo flavifrons*	T

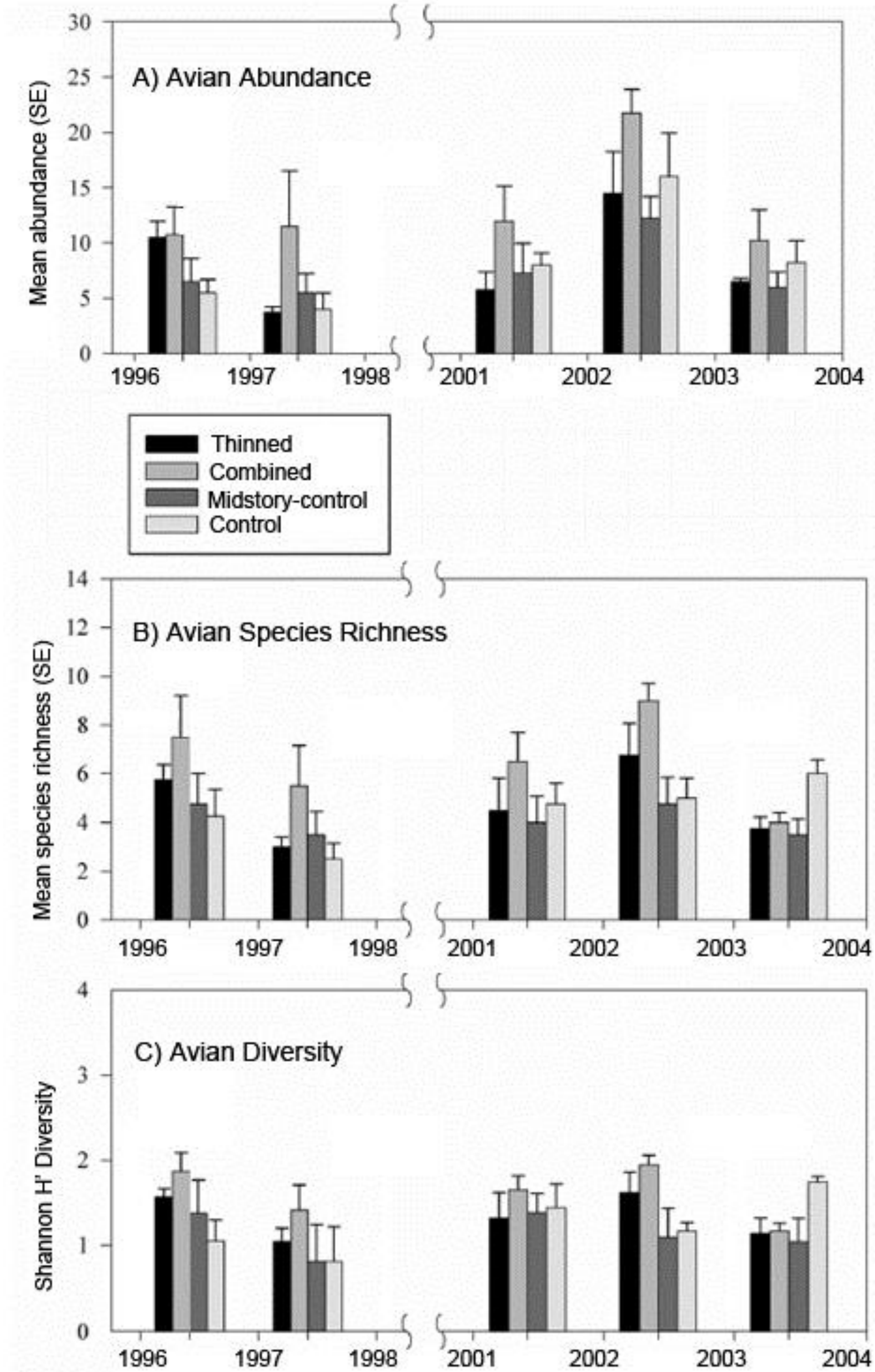

Figure 1--Total avian abundance, species richness, and Shannon's H' diversity in factor level combinations of thinning and midstory-control in longleaf pine plantations at the Savannah River Site, SC, in Spring 1996, 1997, 2001-2003. Abundance values are the mean number of birds counted per treatment area per year. Stands were thinned in May 1994. Moderate- to high-intensity prescribed fire occurred in winter 1994, 1998, and 2003. Midstory-control occurred in spring 1995 and 1996.

Table 2--Count of small mammals captured in longleaf pine plantations on the Savannah River Site, South Carolina, in 1996-2003 among four treatments

Year	Treatment	Other *Peromyscus* spp.	*Peromyscus polionotus*	*Neotoma floridanum*	*Sigmodon hispidus*	Other	Total	Trap success
								%
Spring	Thinned	4	9	2	1		16	10.0
1996	Combined	4	12			1	17	10.6
n[b]=2	MC[a]	7	1	1		1	10	6.3
	Control	2		3	1		6	3.8
Winter	Thinned			2	3		5	3.1
1996	Combined	4	8	1			13	8.1
n=2	MC	2		1	1		4	2.5
	Control			2			2	1.3
Spring	Thinned	4		1			5	3.1
1997	Combined	2	8			1	11	6.9
n=2	MC	5		2			7	4.4
	Control	2					2	1.6
Spring	Thinned	7					7	2.2
2001	Combined	2				2	4	1.3
n=4	MC	6				1	7	2.2
	Control	5					5	1.6
Spring	Thinned	3					3	0.9
2002	Combined	6					6	1.9
n=4	MC	3					3	0.9
	Control						0	0.0
Winter	Thinned	7					7	2.2
2002	Combined	3				3	6	1.9
n=4	MC	6					6	1.9
	Control	5				1	6	1.9
Spring	Thinned	6					6	1.9
2003	Combined	11				1	12	3.8
n=4	MC	11		1			12	3.8
	Control	7					7	2.2
Winter	Thinned	4					4	1.3
2003	Combined	1			5		6	1.9
n=4	MC	2			1		3	0.9
	Control	3					3	0.9
	Total	134	38	16	12	11	211	

[a]MC = Midstory-control
[b]Number of sites sampled

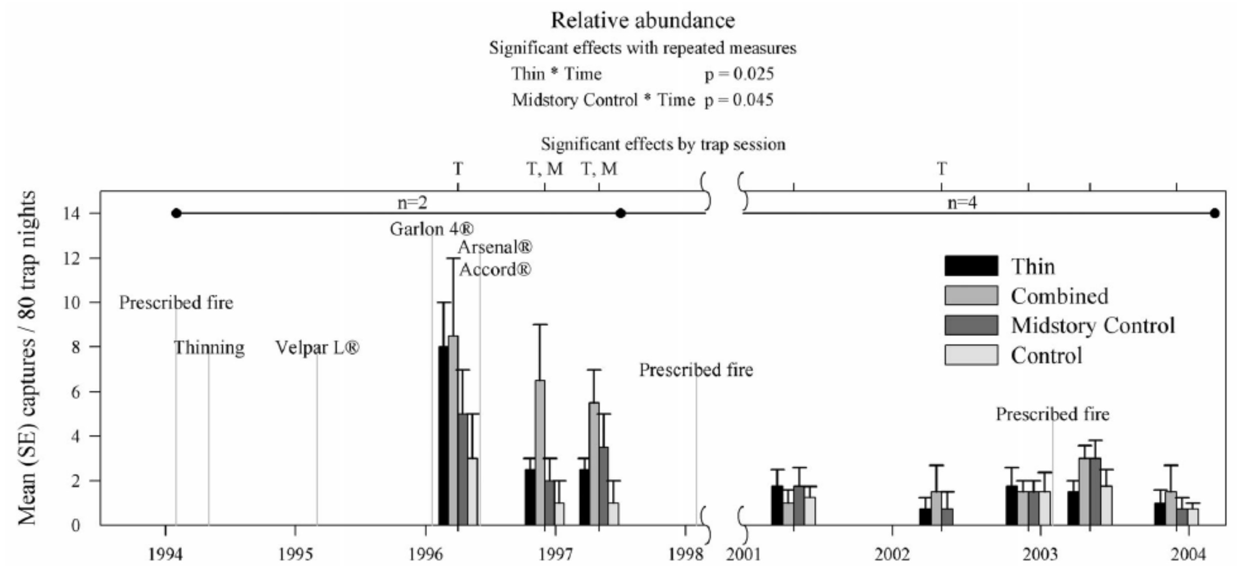

Figure 2--Small mammal relative abundance by factor level combination of precommercial thinning and midstory-control in longleaf pine plantations at the Savannah River Site, SC, by trapping session (spring and winter) 1996, 1997, 2001-2003. Abundance values are mean small mammals of all species captured per 80 trap nights. Significant treatment effects of repeated measures analysis are reported below the title. Significant effects for each trapping session are included on the upper x-axis [thinning (T), midstory-control (M), and interaction (I)]. Significance level is α = 0.10.

1 percent). Thinning and midstory-control alone affected small mammal relative abundance, but the interaction of thinning and midstory-control was nonsignificant, indicating a simple additive effect of treatments in the combined treatment (fig. 2). Combined treatment plots contained the most small mammal captures for spring 1996, winter 1996, and spring 1997, but captures were similar among treatments in later years. Small mammal relative abundance was greatest in 1996-1997 (average 8.0 captures per 80 trap nights in combined treatment); by 2001, small mammal relative abundance was low (average 1.8 captures per 80 trap nights in combined treatment) and remained low for the rest of the study. Small mammal captures were negatively correlated to all pine overstory characteristics that we measured (pine height, d.b.h., crown width, height to base of live crown, and basal area; r = -0.20 to -0.50, P = 0.04 to <0.001). Small mammal relative abundance was also negatively correlated with trapping period (r = -0.48, P < 0.001), hardwood height (r = -0.20, P = 0.03), hardwood d.b.h. (r = -0.19, P = 0.05), hardwood crown width (r = -0.20, P = 0.04), hardwood height to base of live crown (r = -0.24, P = 0.01), and hardwood trees/ha (r = -0.19, P = 0.06).

DISCUSSION

Avian Community Responses

Our combination treatment of thinning and midstory-control was most effective at creating early-successional vegetative conditions that supported the greatest bird species richness and abundance. Application of thinning and midstory-control alone frequently created conditions similar to conditions observed in the nontreated control, thus they were likely ineffective at altering understory vegetation characteristics beyond the effects of periodic prescribed fire. Thinning also increased abundance of midstory hardwoods, which increased shading in the understory (Harrington 2011). A combination of thinning and intensive midstory-control has also been used in Texas loblolly and shortleaf (*P. echinata* Mill.) pine forests to stabilize endemic red-cockaded woodpecker[*Picoides borealis* (Vieillot)] populations with moderate success (Conner and Rudolph 1994). In contrast, a study in young loblolly pine plantations in the Coastal Plain of North Carolina observed positive effects of wide pine spacing on bird communities but alternating effects of understory woody control; in early years, bird abundance was less in plots receiving understory woody control, but this trend reversed in later years due to decreased understory herbaceous vegetation structure and diversity as pines reached canopy closure (Lane and others 2011). However, Kilgo and Bryan (2005) observed increased abundance of birds following removal of understory woody vegetation in longleaf pine forests. Similarly, a study in Georgia Piedmont and Coastal Plain pine stands did not observe any bird species whose abundance was positively related with increasing understory woody vegetation, but 10 species were negatively associated with this condition (Klaus and Keyes 2007). Thus, avian diversity and abundance in southern pine stands appear to be associated with open canopy and midstory conditions.

We did not measure fauna or flora characteristics prior to implemented treatments; thus we are unable to discern direct before and after effects of our treatments. However, other studies suggest that early-successional plant and wildlife species show a rapid but short-lived response to disturbances such as thinning and midstory-control (Lane and others 2011, O'Connell and Miller 1994). Although sampling year significantly explained some variation in our results, we did not observe a noticeable, consistent change in avian abundance, species richness, diversity, or change in bird assemblages throughout our study. This suggests that periodic prescribed fire used in our study was likely effective at holding a relatively stable early- to mid-successional vegetation community in our stands. Thus, treatments used to restore early-successional pine communities may create conditions that will remain fairly stable in following years with the use of periodic prescribed fire unless additional treatments are used to further change plant communities and vegetation structure.

Our results indicate that a combined treatment of thinning and midstory-control benefit avian communities in longleaf pine plantations by developing stand conditions more typical of natural stands when maintained by periodic fire. However, the optimal fire return interval to maintain the savanna conditions and abundant herbaceous vegetation in longleaf stands is likely more frequent than the 4- to 5-year interval we implemented. Shortening fire intervals to 2 to 3 years may facilitate further reductions in understory woody cover that will likely promote the establishment of early-successional forbs and grasses and make singular thinning or midstory-control treatments more effective (Glitzenstein and others 2003). Otherwise, additional midstory-control treatments may be necessary to mitigate establishment of understory woody vegetation, particularly stems that grow large enough to survive low-intensity prescribed fire during prolonged fire rotations.

Small Mammal Community Responses

All treatments resulted in short-term increases in small mammal relative abundance over the control, but treatment effects were minimal by 2001. Few studies have examined the effects of herbicides on small mammals in southern pines, and most of these have focused on site preparation rather than mid- or late-rotation stands (Hood and others 2002, Miller and Miller 2004). We did not observe correlations between small mammal relative abundance and understory herbaceous and woody cover as previously cited (Atkeson and Johnson 1979). However, small mammal relative abundance was negatively correlated with the density and size of hardwood stems, suggesting midstory-control when combined with thinning created habitat conditions favorable to small mammals, particularly *Peromyscus* spp., if only for short time.

Short-term increases of 1 to 3 years and then precipitous declines to low but apparently stable populations have been observed in small mammal communities following anthropogenic and natural disturbances in southern pine forests (Moore 1993, O'Connell and Miller 1994). Decreasing the return interval of prescribed fires (from 4 years to 2 to 3 years) may provide better long-term hardwood control and thus support greater numbers of small mammals, especially pioneer species that utilize early-successional habitats (Bechard 2008). A 4-year prescribed-fire return interval is considered the upper limit of the range generally recommended to maintain longleaf dominance and understory abundance and diversity (Frost 1993, Glitzenstein and others 2003). Provided there is sufficient fuel to result in a moderately intense fire, shorter fire-free periods may prevent hardwoods from becoming large enough to be fire resistant.

Our findings support Brennan and others' (1998) assertion that combining herbicide use with fire may be more beneficial than either treatment alone and could have long-lasting (10 to 15 year) effects on understory flora and fauna communities. Therefore, precommercial thinning in longleaf plantations, particularly when combined with woody control and prescribed fire, may benefit early-successional avian and small mammal communities by developing stand conditions more typical of natural longleaf stands maintained by periodic fire.

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LITERATURE CITED

Atkeson, T.D.; Johnson, A.S. 1979. Succession of small mammals on pine plantations in the Georgia Piedmont. American Midland Naturalist. 101: 385-392.

Bechard, A.M. 2008. Influence of row spacing and debris distribution on vegetation and small mammals in Louisiana pine plantations. Baton Rouge, LA: Louisiana State University. 81 p. M.S. thesis.

Brennan, L.A.; Engstrom, R.T.; Palmer, W.E. [and others]. 1998. Whither wildlife without fire? Transactions of the North American Wildlife and Natural Resources Conference. 63: 402-414.

Brunjes, K.J.; Miller, K.V.; Ford, W.M. [and others]. 2003. Effects of thinning and herbicide application on vertebrate communities in longleaf pine plantations. Proceedings of the annual conference of Southeastern Association of Fish and Wildlife Agencies. 57: 252-267.

Burt, W.H.; Grossenheider, R.P. 1976. A field guide to the mammals. 3rd ed. Boston: Houghton Mifflin. 289 p.

Conner, R.N.; Rudolph, D.C. 1994. Red-cockaded woodpecker population trends and management on Texas national forests. Journal of Field Ornithology. 66: 140-151.

Conner, R.N.; Rudolph, D.C.; Walters, J.R. 2001. The red-cockaded woodpecker: surviving in a fire-maintained ecosystem. Austin: University of Texas Press. 400 p.

Conover, W.J.; Iman, R.L. 1981. Rank transformations as a bridge between parametric and nonparametric statistics. American Statistician. 35: 124-129.

Cothran, E.G.; Smith, M.H.; Wolff, J.O.; Gentry, J.B. 1991. Mammals of the Savannah River Site. Aiken, SC: Savannah River Site, National Environmental Research Park Program. 191 p.

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. Proceedings of the Tall Timbers Fire Ecology Conference. 18: 17-43.

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: 22-37.

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.

Harrington, T.B.; Edwards, H.B. 1999. Understory vegetation, resource availability, and litterfall responses to pine thinning and woody vegetation control in longleaf pine plantations. Canadian Journal of Forest Research. 29: 1055-1064.

Hedman, C.W.; Grace, S.L.; King, S.E. 2000. Vegetation composition and structure of southern coastal plain pine forests: an ecological comparison. Forest Ecology and Management. 134: 233-247.

Hood, S.A.; Miller, D.A.; Leopold, B.D.; Burger, L.W. 2002. Small mammal and herptile response to mid-rotation pine management in Mississippi. Proceedings of the annual conference of the Southeastern Association of Fish and Wildlife Agencies. 56: 171-186.

Kilgo, J.C.; Blake, J.I. 2005. Ecology and management of a forested landscape: fifty years on the Savannah River Site. Washington, DC: Island Press. 479 p.

Kilgo, J.C.; Bryan, A.L., Jr. 2005. Nongame birds. In: Kilgo, J.C.; Blake, J.I. Ecology and management of a forested landscape: fifty years on the Savannah River Site. Washington, DC: Island Press: 223-253.

Klaus, N.; Keyes, T. 2007. Effect of two native invasive tree species on upland pine breeding bird communities in Georgia. Wilson Journal of Ornithology. 119: 737-741.

Krementz, D.G.; Christie, J.S. 2000. Clearcut stand size and scrub-successional bird assemblages. Auk. 117: 913-924.

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.

Lane, V.R.; Miller, K.V.; Castleberry, S.B. [and others]. 2011. Bird community responses to a gradient of site preparation intensities in pine plantations in the Coastal Plain of North Carolina. Forest Ecology and Management. 262: 1668-1678.

McMahon, C.K.; Tomczak, D.J.; Jeffers, R.M. 1998. Longleaf pine ecosystem restoration: the role of the USDA Forest Service. In: Kush, J.S., comp. Proceedings of the longleaf pine ecosystem restoration symposium, 9th annual international conference, Society of Ecological Restoration. Longleaf Alliance Report. 3: 20-31.

Miller, J.H.; Robinson, K.S. 1995. A regional perspective of the physiographic provinces of the southeastern U.S. In: Edwards, M.B., comp. Proceedings of the 8th biennial southern silvicultural research conference. Gen. Tech. Rep. 1. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 581-591.

Miller, K.V.; Miller, J.H. 2004. Forestry herbicide influences on biodiversity and wildlife habitat in southern forests. Wildlife Society Bulletin. 32: 1049-1060.

Moore, W.F. 1993. Effects of chemical site preparation on vegetative, small mammal, and songbird communities in the Georgia sandhills. Athens, GA: University of Georgia. 208 p. M.S. thesis.

Mueller-Dombois, D.; Ellenberg, H. 1974. Aims and methods of vegetation ecology. New York: Wiley & Sons. 547 p.

Noss, R.F. 1989. Longleaf pine and wiregrass: keystone components of an endangered ecosystem. Natural Areas Journal. 9: 211-213.

O'Connell, W.E.; Miller, K.V. 1994. Site preparation influences on vegetative composition and avian and small mammal communities in the South Carolina Upper Coastal Plain. Proceedings of the annual conference of the Southeastern Association of Fish and Wildlife Agencies. 48: 321-330.

Peet, R.K.; Allard, D.J. 1993. Longleaf pine vegetation of the Southern Atlantic and Eastern Gulf Coast regions: a preliminary classification. Proceedings of the 18th Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station. 18: 45-81.

Peet, R.K.; Wentworth, T.R.; White, P.S. 1996. A flexible, multipurpose method for recording vegetation composition and structure. Castanea. 63: 262-274.

Plentovich, S.; Tucker, J.W.; Holler, N.R.; Hill, G.E. 1998. Enhancing Bachman's sparrow habitat via management of red-cockaded woodpeckers. Journal of Wildlife Management. 62: 347-354.

Radford, A.E.; Ahles, H.E.; Bell, C.R. 1968. Manual of the vascular flora of the Carolinas. Chapel Hill: University of North Carolina Press. 1183 p.

Repenning, R.W.; Labisky, R.F. 1985. Effects of even-aged timber management on bird communities of the longleaf pine forest in northern Florida. Journal of Wildlife Management. 49: 1088-1098.

Ricklefs, R.E. 1997. The economy of nature: a textbook in basic ecology. 4th ed. New York: W.H. Freeman. 550 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. 139 p.

Simberloff, D. 1993. Species-area and fragmentation effects on old-growth forests: prospects for longleaf pine communities. Proceedings of the 18th Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station. 18: 1-14.

Stroh, M.H.; Marsinko, A.P.; Guynn, D.C. 2002. Selected characteristics of forest industry hunt-lease programs in the South. In: Smathers, W.M., Jr.; Bergstrom, J.C., eds. Issues in wildlife economics and the ecology–economics interface. Southern Natural Resource Economics Committee workshop proceedings: 27–34. http://srdc.msstate.edu/publications/archive/230.pdf. [Date accessed: May 14, 2013].

Trani, M.K.; Brooks, R.T.; Schmidt, T.L. [and others]. 2001. Patterns and trends of early successional forests in the eastern United States. Wildlife Society Bulletin. 29: 413-424.

Van Lear, D.H.; Jones, S.M. 1987. An example of site classification in the southeastern Coastal Plain based on vegetation and land type. Southern Journal of Applied Forestry. 11: 23-28.

Ware, S.; Frost, C.C.; Doerr, P.D. 1993. Southern mixed hardwood forest: the former longleaf forest. In: Martin, W.H.; Boyce, S.G.; Echternacht, A.C., eds. Biodiversity of the southeastern United States. Lowland terrestrial communities. New York: Wiley & Sons: 447-493.

DEVELOPMENT OF AN ECOLOGICAL CLASSIFICATION SYSTEM FOR THE COOPER CREEK WATERSHED OF THE CHATTAHOOCHEE NATIONAL FOREST: A FIRST APPROXIMATION

W. Henry McNab, Ronald B. Stephens, Erika M. Mavity, Joanne E. Baggs, James M. Wentworth, Richard D. Rightmyer, Alex J. Jaume, Brian D. Jackson and Michael P. Joyce[1]

Abstract--The 2004 management plan for the Chattahoochee National Forest states that many future resource objectives and goals have an ecological basis. Assessment of resource needs in the Cooper Creek watershed area of the southern Appalachian Mountains of north Georgia were identified with awareness of ecological constraints and suitability. An interdisciplinary team of resource specialists developed a land-classification system for the watershed that identifies and maps 28 recurring land and water units with unique ecological characteristics. The classification will provide a basis to plan and implement management activities that are appropriate, cost effective, and consistent with views and concerns of a larger community of stakeholders.

INTRODUCTION

The Chattahoochee National Forest (CNF) extends across about 750,000 acres in Appalachian Mountains and Ridge and Valley physiographic provinces in northern Georgia. Beginning in 1911, tracts later designated as the CNF were purchased under the Weeks Act for control of wildfires and management of lands forming watersheds of headwater streams of navigable rivers. These lands, once in a forested condition maintained for thousands of years by natural- and Native American-influenced disturbances, were highly altered at the time of USDA Forest Service acquisition, resulting from subsistence agriculture practices of early European settlers in the 1800s and extensive commercial logging. Loss of the American chestnut [*Castanea dentate* (Marshall) Borkh.] in the 1920s, extensive planting of eastern white pines (*Pinus strobus* L.) through the Civilian Conservation Corps program in the 1930s, and most recently the gradual demise of eastern hemlock [*Tsuga canadensis* (L.) Carrière], have resulted in additional changes of species composition. Also, nearly 100 years of suppressing both natural- and human-caused fires has resulted in additional changes of vegetation species composition. The present CNF consists of a mosaic of forest stands with varying histories of disturbance, which are slowly changing with age toward a species composition compatible with the physical environment under a reduced disturbance, low-intensity management.

Vegetation management in the CNF provides products, services, and benefits desired by society at local and regional scales. The current forest plan of 2004 was crafted with input from many stakeholders to meet a range of objectives and goals, with a strong emphasis on ecosystem restoration and basing management goals on ecologically sound information with considerations of social needs and economic limitations. Planning for management activities in the CNF is done at a landscape scale typically formed by large single or multiple watersheds called project areas, which are appropriate for assessment of the effects of vegetation management activities on multiple resources, particularly actions with an ecological basis. Project areas are used to identify resource management opportunities to meet Forest Plan objectives using the traditional approach of describing current conditions of vegetation based on stand-level data. If the current vegetative species composition, structure, and age distribution is unlikely to meet the desired

[1]Research Forester, USDA Forest Service, Southern Research Station, Asheville, NC 28806; Forest Silviculturist (retired), Forest GIS Specialist, and Forest Botanist, respectively, USDA Forest Service, Chattahoochee-Oconee National Forest, Gainesville, GA 30501; Wildlife Biologist, USDA Forest Service, Chattahoochee-Oconee National Forest, Blairsville, GA 30512; and Forest Soil Scientist, GIS Analyst Trainee, Forest Silviculturist, and Forest Fisheries Biologist, respectively, USDA Forest Service, Chattahoochee-Oconee National Forest, Gainesville, GA 30501.

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future conditions, resource managers then propose actions that accelerate development of the stand toward that condition. Resource management objectives of the current Forest Plan bring together goals that were largely not attainable with traditional methods where emphasis was given to increasing yields of a few timber species with high commercial value. Now, desired future stand conditions often include restoration of a prescribed fire regime that in turn requires consideration of the biological potential of the area, which may be based on experience of the forester or by reference to a previously developed site classification.

Classification of land units based on ecological principles involves identifying physical properties of the environment that combine to define the productive potential of sites associated with temperature, moisture, and fertility gradients (Barnes and others 1982). The ecological potential of terrestrial sites is expressed by the vegetative community that would be present resulting from natural disturbances such as from climate, fauna, fire, insects, disease, and non-European humans. In 1992, the Forest Service adopted a policy of taking an ecological approach to management of national forests (Salwasser 1992) and developed a hierarchical framework of ecological units appropriate at a range of scales from national assessments, to regional inventories, and local land management projects (table 1) (Cleland and others 1997).

Relatively little information is available for application of an ecological-based classification to support project-level management actions in the CNF. Griffith and others (2001) mapped ecoregions in Georgia using a national hierarchy adopted by the U.S. Environmental Protection Agency primarily for water quality issues, which later provided a framework for describing the natural vegetative communities (Edwards and others 2013). Ecological mapping at the mid-scale subregion-level (suitable for state-level planning) was done by Cleland and others (2007) based on the Forest Service hierarchy.

The Forest Service national framework was the basis for an ecological classification developed for the Oconee National Forest in the Appalachian Piedmont of north-central Georgia (McNab and others 2012). These broad scale classifications are appropriate for state-level assessments but do not provide detail needed for planning at the watershed or project levels.

Although ecological studies have not been done in the Cooper Creek project area, relevant information is available from investigations of vegetation and environments in nearby areas. Ike and Huppuch (1968) found throughout the mountains of north Georgia that composition of species of arborescent vegetation was associated with features of soil and landform that defined a moisture gradient. Moffat (1993) working in watersheds east of the Cooper Creek project area (primarily in White County with smaller areas of Union and Towns Counties) reported 10 vegetative communities were related to a temperature and moisture gradient defined by elevation, slope position, and landform. Graves and Monk (1985) found that composition of understory vegetation and some tree species were related to soils derived from differing geologic substrate. Chafin and Jones (1989) found differences in composition of vegetation in high-elevation boulder fields compared to coves. These and other studies in nearby areas of the Southern Appalachians have demonstrated that vegetation is a biological integrator of environmental conditions and is responsive to varying but relatively stable physical properties of sites. However, vegetation can be an imperfect indicator of specific environment conditions because species composition and dominance may vary in response to the type and intensity of disturbance and the time since disturbance (Clinton and Vose 2000).

This report describes the development of an ecologically based method for identifying and grouping areas of land in a small area of the

Table 1--USDA Forest Service national hierarchy of ecological units[a]

Planning and analysis scale	Ecological units	Purpose and general use	General size range
Ecoregion	Domain Division Province	Broad applicability for modeling and sampling; national planning	Millions to tens of thousands of square miles
Subregion	Section Subregion	Strategic, multi-agency analysis and assessment	Thousands to tens of square miles
Landscape	Landtype association	Watershed analysis	Thousands of acres
Land unit	Landtype Landtype phase	Project and management area planning	Hundreds to tens of acres
	Site	Field sampling	Ten to less than one acre

[a]From Cleland and others 1997.

southern Appalachian Mountains of northern Georgia where certain management goals are best suited ecologically. The objectives of our paper are to: (1) describe the rationale and process used to develop an ecological-based classification, and (2) provide a tabulation of the ecological units identified. The scope of this study was limited to national forest lands of the Cooper Creek watershed assessment area in the Blue Ridge Ranger District of the Chattahoochee National Forest, in the Southern Appalachian Mountains, near Blairsville, GA. We consider this study as a pilot project for small-scale ecological unit delineation in the Blue Ridge Mountains using a broad base of disciplines, but without field data collection. The primary goals of the project were to: (1) determine if such an approach is feasible, and (2) develop a foundation for implementing ecosystem management. The methods we used deviated from those recommended by the agency for development of an ecological classification because of limited resources. Lessons learned from the modified process will serve as a foundation for future refinement and development of our classification.

METHODS

Study Area

The Cooper Creek watershed assessment is an area of about 34,000 acres extending across three large watersheds (Cooper, Young Cane, and Coosa Creeks) near Blairsville, GA (fig. 1), where management goals were identified based on being ecologically sustainable, appropriate or providing desirable benefits to both local and regional user groups (Unpublished office report on file at the Supervisors Office, Chattahoochee National Forest, Gainesville, GA.). Included in the assessment area is the Cooper's Creek Wildlife Management Area, administered in cooperation with the Georgia Department of Natural Resources, Wildlife Resources Division, which occupies 30,000 acres in Fannin and Union Counties. Physiography of this area is mountainous with broad intermountain valleys; elevations range from 1,978 to 4,330 feet. Bedrock geology is mostly Precambrian metamorphic formations consisting mostly of gneiss and schist. Climate is hot continental with temperature averaging 34.9 °F (range 22.7 °F to 47.1 °F) in January and 72.5 °F in July (range 60.9 °F to 84.0 °F). The frost-free period ranges between 140 to 180 days. Average annual precipitation ranges between 59 and 83 inches with highest amounts along the Blue Ridge watershed divide crest. Rainfall steadily decreases to the north into a 'rain shadow' of atypically low rainfall near Blairsville, GA. Precipitation is generally evenly distributed annually, although mild drought is common in the fall. The dominant soil orders are Ultisols and Inceptisols; moisture regime is udic and temperature regime is mesic. Soil depths range from shallow to deep; texture classes are typically loamy or clayey. Forest vegetation is

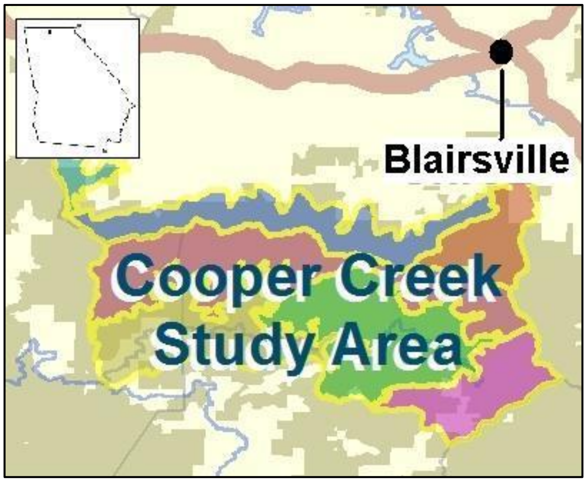

Figure 1--The study area in the Cooper Creek and adjoining watersheds of the Chattahoochee National Forest southwest of Blairsville, GA. Colored polygons within the study area show ecological units delineated at the landtype level in the Forest Service national hierarchical framework. Areas of national forest beyond the study area are shaded tan. The black dot on the inset map shows the general location of the study area in the southern Appalachian Mountains of north Georgia.

predominantly deciduous hardwoods throughout, varying from xerophytic species [oaks (*Quercus spp*) and hickories (*Carya spp*)] on ridges and slopes and mixed mesophytic species [yellow-poplar (*Liriodendron tulipifera* L.), sweet birch (*Betula lenta* L.), red maple (*Acer rubrum* L.)] on lower slopes and in valleys. Above about 3,000 feet elevation, species composition on cool north-facing slopes gradually changes to include yellow buckeye (*Aesculus octandra* Aiton) and basswood (*Tilia americana* L.) in coves and sweet birch on slopes and ridges. Conifers include shortleaf (*P. echinata* Mill.) and Virginia pines (*P. virginiana* Mill.) on ridges; eastern white pine occurs throughout. Eastern hemlock was a common former component of riparian areas along streams and in stands on lower slopes until recently, when almost total mortality resulted from effects of the hemlock woolly adelgid (*Adelges tsugae*). Its presence today is limited to isolated areas where insecticide and biological control methods have been used to conserve small populations of hemlock trees. Until about 1920, American chestnut likely occurred throughout and was a major component of most oak stands, particularly on middle and upper slopes. Almost all stands were heavily logged during the late 1890s and burned during the 1830 to 1930 mountain farmstead era (Brender and Merrick 1950). Natural- and human-caused fire has been controlled in the study area since about 1920, with the exception of a few areas where fire has been re-introduced through the prescribe fire program on selected landscape-scale burn units ranging in size from hundreds to low thousands of acres since the mid-1990s.

Classification Framework and Ecological Units

We used the Forest Service hierarchical framework of ecological units (Cleland and others 1997) as the basis for our classification (table 1). The national framework is based primarily on climatic factors appropriate at each level, with increased emphasis and importance of progressively localized physiographic, geologic, and edaphic factors that modify the effects of temperature and precipitation (Bailey 1983). National- and regional-scale ecological units had been identified and described at the ecoregion and subregions levels, which were identified and mapped using successive stratification of large somewhat heterogeneous units into smaller, relatively homogeneous units (Cleland and others 1997). Stratification into ecologically uniform areas becomes increasingly difficult at lower levels in the hierarchy because ecotones between units represent gradients of compensating environmental factors that are not clearly seen or measured. An important consideration in development of the classification was delineation of landscape units easily recognizable and of sufficient size for use by resource managers.

In the Cooper Creek study area, we identified and delineated ecological units by successive stratification of previously mapped landtype associations (LTAs) into smaller landtypes (LTs), which were subdivided into smaller and more homogenous landtype phases (LTPs) (table 1). Time and funding resources available for this study did not allow collection of field data for developing quantitative relationships between vegetative communities and environmental factors, such as elevation and aspect. Instead, land-stratification criteria were based on personal knowledge of biological relationships provided by a multidisciplinary team that included silviculturists, a botanist, soil scientist, fisheries biologist and wildlife biologist using the display on-screen and tabular results of iterative geographic information system (GIS) analysis of basic and inter-related data layers. Available sources of field vegetation data at the stand level included conventional forest cover, age and wood production-based condition data [Forest Service databases included Field-Sampled Vegetation (FSVeg) and its precursor Continuous Inventory of Stand Conditions (CISC)] and permanent inventory plots installed by the Forest Inventory and Analysis (FIA) branch of the Forest Service] for national assessment of timber resources. Evaluation of these two sources of data revealed the FSVeg/CISC data were not suitable for delineation of ecological units or identification of potential vegetation because it had been collected mainly for silvicultural purposes. The FIA data set consisted of about seven field plots in the study area, and therefore was too small to be used for analysis. Until other data become available, we use vegetative communities described by Edwards and others (2013) as the description of natural communities associated with the ecological units. Another decision was not to nest the smallest land units (LTPs) within LTs or LTAs, which would have increased the number of classification units in the classification to an unwieldy size. Therefore, the same LTPs may occur in all LTAs and LTs.

Utilization of a GIS was essential for implementing conceptual models of ecological relationships developed by the interdisciplinary team using digital elevation data sets. The topographic position index (Guisan and others 1999) was used with a digital elevation model to group areas within the landscape into categories of landform, ranging from convex ridges, nearly linear slopes, to concave valleys. This index, when combined with aspect and slope gradient, provided a means of subdividing the landscape into units of similar ecological potential (related to moisture gradient and solar radiation) represented by a distinctive vegetative community of characteristic species composition. Development and refinement was an iterative process where concepts developed by the team were implemented and displayed via GIS, evaluated and revised. Verification of ecological relationships was done through several field visits to representative sites for evaluation of predicted and actual conditions, followed by refinement of the model and additional field verification.

RESULTS

Landscape Scale Ecological Units

LTAs had been tentatively mapped in the Southern Blue Ridge Mountains Subsection (M221Dc) as part of the Southern Appalachian Assessment (SAMAB 1996). LTAs in this subsection were identified to account primarily for environmental variation associated with physiography and differential climate related to landform, primarily cooler climates at higher elevation and precipitation related to orographic effects from mountain ranges. The Cooper Creek study area occupied parts of two LTAs: M221Dc17 and M221Dc18. We made mostly minor adjustments to the boundaries of these two previously mapped LTAs as a result of on-screen review of GIS analysis.

Land Unit Scale Ecological Units

A total of seven LTs were identified in the two LTAs that accounted for environmental variation at a smaller scale related to elevation, landform, and predominant aspect. For example, Duncan Ridge North LT and Duncan Ridge South LT together account for areas of mid-elevation in the study area but are distinctly different in predominant aspect, terrain sheltering and landform. Individually, these LTs stratify the project area into landscapes that differ ecologically. The other LTs were mapped to account for similar variation at other locations.

We identified 28 LTPs in the study area (table 2). LTPs are smaller parts of LTs with increased environmental uniformity resulting from elevation (primary versus secondary ridge, secondary versus minor ridge), landform including terrain sheltering (exposed versus protected slope), and aspect (cool versus warm slope). Using a GIS, the conceptual models of combinations of elevation, landform, aspect, and relative slope position were mapped as LTP ecological units throughout the Cooper Creek study area (fig. 2).

DISCUSSION

Management Interpretations

Identification and classification of ecological units from our study using expert knowledge methods were more detailed compared to results reported by Moffat (1993), who used multivariate analysis of extensive field data from the Chattahoochee Game Management Area, which forms headwaters of the Chattahoochee River, east of the Cooper Creek study area. Moffat (1993) found four groups of land units: one riparian unit along large streams and three units associated with elevation zones. Excluding the sparsely sampled riparian and high-elevation sites, Moffat (1993) reported 4 subunits within the low- and middle-elevation zones that were related to moisture gradients, resulting in a total of 10 ecological units in his study area. Our results differed from Moffat (1993) primarily in the combined low- and middle-elevation zones where we identified three moisture gradients associated with each of three ridge types: primary, secondary, and minor. Additional ecological variation present in our study area, but not recognized by Moffat (1993), included eight types of slopes and three types of coves (table 2).

The 28 ecological units in our classification system will provide basic information for natural resource planning and management at the landscape scale in the Cooper Creek project area. Examples of anticipated uses are listed below:

1. Planning-LTPs offer an ecological based foundation for organizing, assessing and integrating information from stakeholder groups with common goals of a sustainable supply of forest resources.
2. Restoration-LTPs are a source of information on those parts of the Cooper Creek landscape where prescribed burning can be appropriately and economically used for limiting the spread of eastern white pine to areas where it was not historically present.
3. Recreation-LTP mapping allows direct analysis of the capability for managers to substitute camp sites in non-riparian Ecological Classification System (ECS) units for currently over-used sites in riparian areas.
4. Silviculture and wildlife habitat-LTP descriptions contain the information on physical site components that allow informed prediction of species composition of regeneration following planned disturbance resulting from silvicultural activities such as timber harvest. Maintaining oaks and diversity of habitat structure is important for many species of wildlife, particularly neotropical migratory birds.
5. Water quantity and quality-LTPs provide information on water yields with different vegetative covers and the effects on water quality and watershed health resulting from hemlock mortality in riparian zones caused by the hemlock woolly adelgid.

Lessons Learned

An important component of this study was to develop a context for estimating resources needed to complete ECS throughout the CNF, and to identify efficiencies, barriers, and opportunities for breakthroughs in future studies. Some of the more important lessons we learned include:

1. Mapping concepts of higher-level ECS units must also be considered when delineating small scale units. This project was very near the Hot Continental and Humid Subtropic Division boundary which is also coincident with: the Broadleaf-Coniferous Forest-Meadow Province of

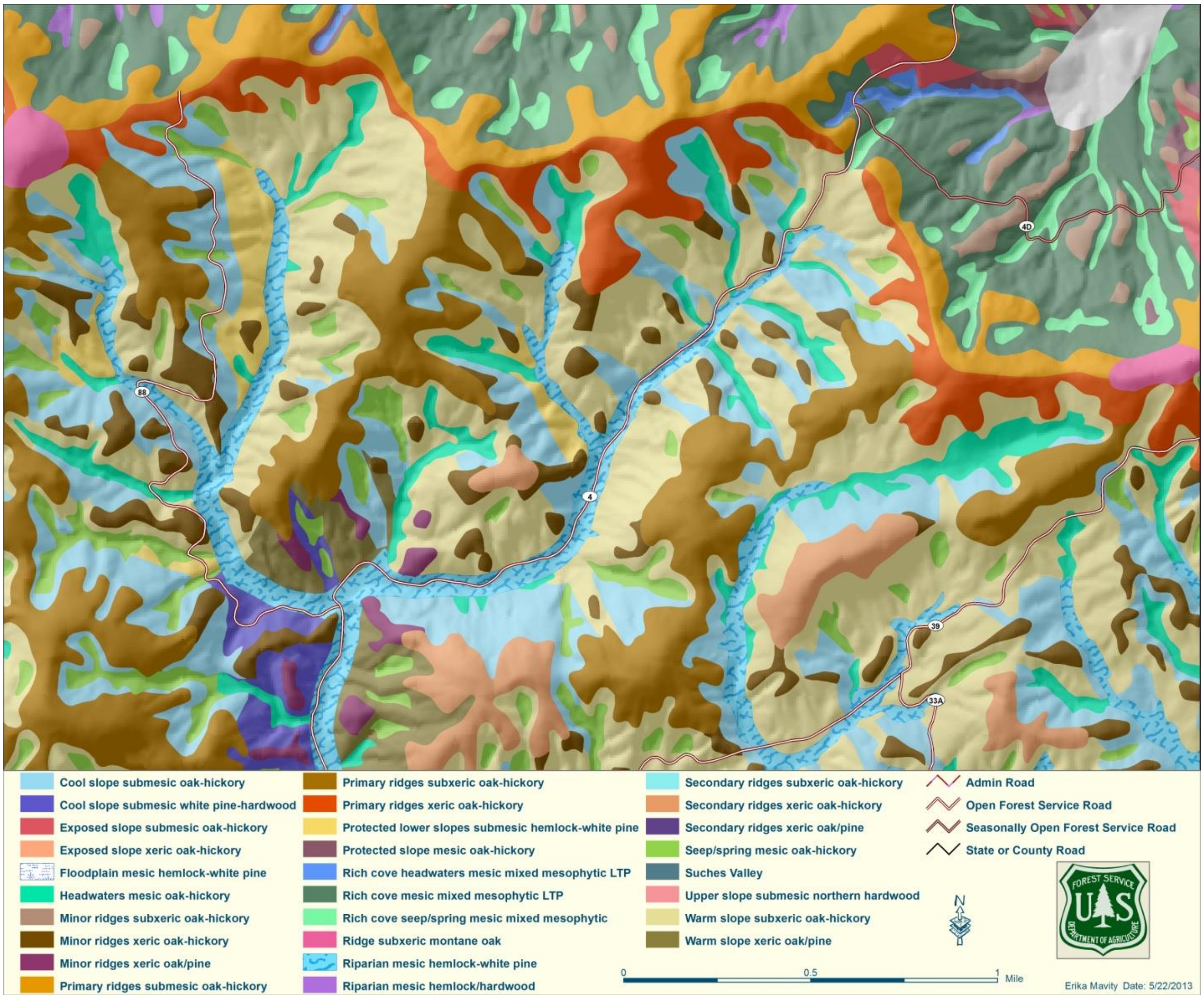

Figure 2--Twenty eight landtype phases accounting for environmental differences associated with variations of elevation, landform, aspect, slope position, and other environmental variables in the Cooper Creek project area of the Chattahoochee National Forest, in north Georgia. This graphic illustrates how the GIS applied the conceptual model to classify the landscape in a selected part of the project area.

the Appalachians and the Southeastern Mixed Forest Province of the Piedmont. These high-level units have a wide ecotone along their boundary extending both into the mountains and out into the Piedmont. The concept of ECS is that delineation of lower-level units refines the boundary of higher-level ones, but it is necessary to recognize that small-scale units such as LTPs may better belong to a much higher level of the hierarchy. In our case, mountainous landform and its ability to modify climate caused the Division and Province boundary to be placed south of the mountains. But the ecotone extends into the mountains. This can cause either relocation by refinement of the higher-level unit boundaries or the delineation of a rather large transitional unit within the mountains. Correctly understanding ecological behavior in the transition zone is challenging.

2. Learning the inefficiencies of classifying a small part of a larger area - be it a watershed, a mountain ridge, or a county - when their boundaries are not nested within existing ECS unit boundaries. We mapped portions of two LTPs. Inefficiencies arose because the

Table 2--Preliminary non-hierarchical ecological units occurring at the landtype phase level of the ecological classification system in the Cooper Creek study area of the Chattahoochee National Forest, Union County, Georgia

Vegetative community[a]	LandType Phase ecological unit name[b]	N. units	Mean size (acres)	Median elevation	Relative slope position[c]
			acres	*feet*	
Rich cove forests	Rich cove mesic mixed mesophytic	48	53	2,701	25
	Rich cove headwaters mesic mixed mesophytic	30	7	2,808	58
	Rich cove seep/spring mesic mixed mesophytic	147	3	2,696	37
Acidic cove forests	Cool slope submesic white pine-hardwood	88	15	2,316	24
	Protected lower slopes submesic hemlock-white pine	49	10	2,814	31
	Riparian mesic hemlock-white pine	37	70	2,646	65
	Floodplain mesic hemlock-white pine	2	395	2,525	63
	Suches Valley	1	153	2,966	34
Low to mid elevation oak forests	Primary ridges subxeric oak-hickory	12	118	2,999	8
	Primary ridges xeric oak-hickory	18	47	3,007	9
	Secondary ridges subxeric oak-hickory	10	41	2,841	7
	Secondary ridges xeric oak-hickory	41	34	2,989	6
	Minor ridges subxeric oak-hickory	246	3	2,691	7
	Minor ridges xeric oak-hickory	359	3	2,782	10
	Warm slope subxeric oak-hickory	206	23	2,877	23
	Exposed slope xeric oak-hickory	16	33	2,218	18
	Secondary ridges xeric oak/pine	9	12	2,503	6
	Minor ridges xeric oak/pine	149	4	2,429	8
	Warm slope xeric oak/pine	71	24	2,323	22
Low to mid elevation oak	Cool slope submesic oak-hickory	238	12	2,831	23
	Exposed slope submesic oak-hickory	125	12	2,658	23
	Primary ridges submesic oak-hickory	25	93	2,835	8
	Protected slope mesic oak-hickory	142	14	2,614	22
	Riparian mesic hemlock/hardwood	29	8	2,364	71
	Headwaters mesic oak-hickory	256	6	2,748	59
	Seep/spring mesic oak-hickory	420	3	2,787	38
Montane oak	Ridge subxeric montane oak	54	23	3,624	15
Northern hardwood	Upper slope submesic northern hardwood	9	8	3,700	32

[a]Based on information in Edwards and others (2013).
[b]These units may occur in any of the landtype associations or landtypes within the study area.
[c]Relative slope position of 0 indicates a ridge and 100 is a Valley.

total range of variability was not known. In our case, the range of variation should have been considered for at least the Georgia portion of the Southern Blue Ridge Mountain Subsection as the proper context of the variation in any of the variables used in classification. LTP concepts may change as the entire LTA landscape is considered. Such an approach would have facilitated using adjective descriptors (low, medium, high) both accurately and in a way that would be more useful into the future.

3. Delineation of smaller units is more difficult and thus more time-consuming than larger scale units because large-scale units rely on coarse and often

easily observable differences, such as mountain versus piedmont landforms. Boundaries of small units are often associated with subtle environmental gradients that are likely to be neither easily observable nor easily understood and described. Also, small scale units are usually discontinuous and are bordered on all sides by a different unit. This hypothesized difference should be observable when the ECS is applied in the field. If that difference is not observable in the field, it jeopardizes credibility of the classification by other users.

4. Correlating existing vegetation with potential vegetation is difficult because of the unknown influences of past land use and effects of disturbance. For example, ecological units that are hypothesized to be similar may show a wide variation of the existing vegetation growing on that unit based on past land use. For example, a middle-slope cool-aspect submesic oak-hickory unit might have significant component of cove hardwood cover or even southern yellow pine. The many successional vegetation communities appropriate for one ecological unit also introduce a complexity to interpreting existing vegetation by the user.
5. The scale of units particular users will want or find most useful can be expected to vary by purpose. For example, a prescribed-fire planner may need general information about blocks of hundreds to thousands of acres, but a silviculturalist may need detailed information about individual stands. Consideration of needs is related to choices made about map display, map scales, and even description detail.
6. Appropriate use of the delineated units required a narrative context. It was not difficult to predict questions that would arise from the use of the units. So we developed an introduction chapter that was designed to be the first and best source to answer these typical questions. It was also intended to be a beginning point for future ecological classification studies on the CNF.
7. It is likely that a GIS analysis of topographic data will result in questionable artifacts of information that form a hypothesized small ecological-classification unit. Many artifacts probably originated from scale-related mapping errors of the original data layers. When the data layers are merged using GIS, the resulting polygons represent illogical ecological units with no apparent biological relationship. We learned that considerable judgment is needed to decide which of the two or more adjacent ecological units represents the proper assignment for each artifact.

In conclusion, the Cooper Creek ecological classification system was a multiphase project to develop a useful tool for improving our ability to plan and implement project-level resource management activities to achieve desired future conditions with a minimum investment of human and economic resources. Because development of an ecological classification is an iterative process of testing and refinement, strengths and weaknesses of this first approximation will be identified and addressed through application, evaluation, and revision. This ecological classification may be viewed as a planning and management tool with many handles for working in collaboration with the Forest Service to achieve common goals of management and sustainable utilization of water and land resources in the Cooper Creek watershed and elsewhere on public lands of the Chattahoochee National Forest.

An unresolved issue was our inability to devise a hierarchical relationship for the ecological units of the Cooper Creek watershed, similar to that developed by Moffat (1993) for a neighboring watershed. Without a hierarchical framework, the 28 ecological units form a single group that appears overly detailed for easy understanding by unfamiliar users. Lack of field data suitable for a quantitative analysis was one reason why we did not develop a hierarchy of units, which would have subdivided the 28 units into groups based on environmental factors of varying ecological importance, such as elevation zones and included moisture regimes. Absence of a hierarchy, however, does not affect the validity and usefulness of the current configuration of the classification for project planning. Development of a hierarchical framework is an opportunity for future study of ecological relationships in the watershed.

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This project would not have been possible without the vision and support of Mr. Ray M. Ellis, retired staff officer for natural resources, Chattahoochee-Oconee National Forests, who provided an understanding and experience with the landscape elements of an ecological classification system (ECS) and the perspective needed by land managers to integrate data into decisions. District Ranger Andrew Baker coordinated the field review of the ECS. Forest Service personnel Denise Bailey, Danny Skojac, and Kurt Steele reviewed an early draft of the manuscript. Questions about the ECS should be directed to the project Team Leader, Dick Rightmyer; Henry McNab, was responsible for the manuscript.

LITERATURE CITED

Bailey, R.G. 1983. Delineation of ecosystem regions. Environmental Management. 7: 365-373.

Barnes, B.V.; Pregitzer, K.S.; Spies, T.A.; Spooner, V.H. 1982. Ecological forest site classification. Journal of Forestry. 80: 493-498.

Brender, E.V.; Merrick, E. 1950. Early settlement and land use in the present Toccoa Experimental Forest. The Scientific Monthly. 71(5): 318-325.

Chafin, L.G.; Jones, S.B, Jr. 1989. Community structure of two southern Appalachian boulderfields. Castanea. 54(4): 230-237.

Cleland, D.T.; Avers, P.E. McNab, H. [and others]. 1997. National hierarchy of ecological units. Chapter 9. In: Boyce, M.S.; Haney, A., eds. Ecosystem management applications for sustainable forest and wildlife resources. New Haven, CT: Yale University Press: 181-200.

Cleland, D.T.; Freeouf, J.A.; Keys, J.E., Jr. [and others]. 2007. Ecological subregions: sections and subsections of the conterminous United States. [1:3,500,000] [CD-ROM]. Sloan, A.M., cartog. Gen. Tech. Rep. WO-76. Washington, DC: U.S. Department of Agriculture Forest Service.

Clinton, B.D.; Vose, J.M. 2000. Plant succession and community restoration following felling and burning in the Southern Appalachian Mountains. In: Moser, W.K.; Moser, C.E., eds. Fire and forest ecology: innovative silviculture and vegetation management. Proceedings of the 21st Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station. 21: 22-29.

Edwards, L.; Ambrose, J.; Kirkman, L.K. 2013. The natural communities of Georgia. Athens, GA: The University of Georgia Press. 675 p.

Graves, J.H.; Monk, C.D. 1985. A comparison of soils and vegetation over marble and schist along tributaries to Panther Creek, Stephens County, Georgia. Castanea. 50: 146-163.

Griffith, G.E.; Omernik, J.M.; Comstock, J.A. [and others]. 2001. Ecoregions of Alabama and Georgia. [1:1,700,00]. Reston, VA: U.S. Geological Survey. [Color poster with map, descriptive text, summary tables, and photographs].

Guisan, A.; Weiss, S.B.; Weiss, A.D. 1999. GLM versus CCA spatial modeling of plant species distribution. Plant Ecology. 143: 107-122.

Ike, A.F.; Huppuch, C.D. 1968. Predicting tree height growth from soil and topographic site factors in the Georgia Blue Ridge Mountains. Georgia Forest Res. Pap. 54. Macon, GA: Georgia Forestry Commission. 11 p.

McNab, W.H.; Stephens, R.B.; Rightmyer, R.D. [and others]. 2012. A preliminary test of an ecological classification system for the Oconee National Forest using forest inventory and analysis data. In: McWilliams, W.; Roesch, F.A., eds. Monitoring across borders. 2010. Joint meeting of the Forest Inventory and Analysis (FIA) symposium and the Southern Mensurationists. e-Gen. Tech. Rep. SRS-157. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 77-86. http://www.srs.fs.usda.gov /pubs/40775. [Date accessed: May 10, 2013].

Moffat, S.O. 1993. Landscape ecosystem classification for the Chattooga Ranger District, Chattahoochee National Forest, Georgia. Clemson, SC: Clemson University. 83 p. M.S. thesis.

Salwasser, H. 1992. From new perspectives to ecosystem management: response to Frissell *et al*. and Lawrence and Murphy. Conservation Biology. 6: 469-472.

SAMAB [Southern Appalachian Man and the Biosphere]. 1996. The Southern Appalachian assessment summary report. Report 1. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region. 118 p.

PARTICIPATORY GENETIC IMPROVEMENT: LONGLEAF PINE

C. Dana Nelson, Gwendolyn Boyd, Randall J. Rousseau, Barbara S. Crane, Craig S. Echt, and Kurt H. Johnsen[1]

University-industry-state cooperative tree improvement has been highly successful in the southern United States. Over nearly 60 years, three cooperative programs have led the way in developing and deploying genetically improved planting stocks for loblolly (*Pinus taeda* L.) and slash (*P. elliottii* Engelm.) pines. However, much lower levels of success have been achieved for species of lesser economic importance such as longleaf (*P. palustris* Mill.) and shortleaf (*P. echinata* Mill.) pines and the many southern hardwoods. The result is that many important forest tree species are in need of sustained genetic enhancement for both short-term silvicultural and long-term conservation purposes. To address this need, we are studying the concept of participatory plant breeding (Atlin and others 2001, Ceccarelli and Grando 2007) for application in forest trees. In particular, we are working on a program for longleaf pine where three types of forest landowner participants would be involved covering the main functions in tree breeding: mother tree selection, progeny testing, and seed production. The program would be organized through a web portal with a back-end database containing the tree, test planting, and orchard data. The program's goal is to provide landowners with an opportunity to actively participate in a region-wide longleaf pine genetic improvement and gene conservation program. In addition, all landowners would benefit from the low-cost availability of well-bred longleaf pine planting stock for optimal performance in a changing climate.

Several decisions are required in establishing a participatory tree breeding program including base population and population structure, target participants and their environmental conditions, the breeding and testing scheme, the improved materials deployment scheme, and how the work is organized among the participants. For longleaf pine, we suggest a base population of about 200 trees per ecoregion and maintaining this size over generations (fig. 1). An ecoregion partially gets at the environmental conditions question in that the associated population is tested and selected for performance within these areas. The areas are defined by similar climate and photoperiod conditions. Within each ecoregion, more specific environmental conditions can be defined (sand hills versus piedmont versus montane), and we suggest that these be used for deployment. For example, within an ecoregion, progeny tests may be established on different physiographic regions or major soil types. Selections from these tests may be used to set up clonal seed orchards specific for the within ecoregion environmental type, or if converting the progeny tests to seedling seed orchards, their seeds can be directed towards similar environmental types. Ecoregions can be defined in many different ways. A couple of approaches seem most useful for longleaf pine, including Craul and others (2005) (site zones) and Griffith and others (2008) (EPA levels III, IV), especially when combined with winter hardiness information (Schmidtling 2001). Potter and Hargrove (2012) have developed a quantitative method for determining ecoregions and projecting their future locations based on climate models that may prove more useful. In addition, practicalities concerning participants' locations and interests and the need to sample and conserve the whole species will affect the number of ecoregions and their borders. One possible case is depicted in figure 1, where six ecoregions are defined, resulting in an overall base population size of about 1,200 trees.

[1]Research Geneticist, USDA Forest Service, Southern Research Station, Southern Institute of Forest Genetics, Saucier, MS 39574; Associate Professor, Alcorn State University, Department of Agriculture, Forestry Program, Lorman, MS 39096; Professor, Mississippi State University, Department of Forestry, Mississippi State, MS 39762; Regional Geneticist, USDA Forest Service, Southern Region, Atlanta, GA 30309; Research Geneticist, USDA Forest Service, Southern Research Station, Saucier, MS 39574; and Research Physiologist, USDA Forest Service, Southern Research Station, Research Triangle Park, NC 27709.

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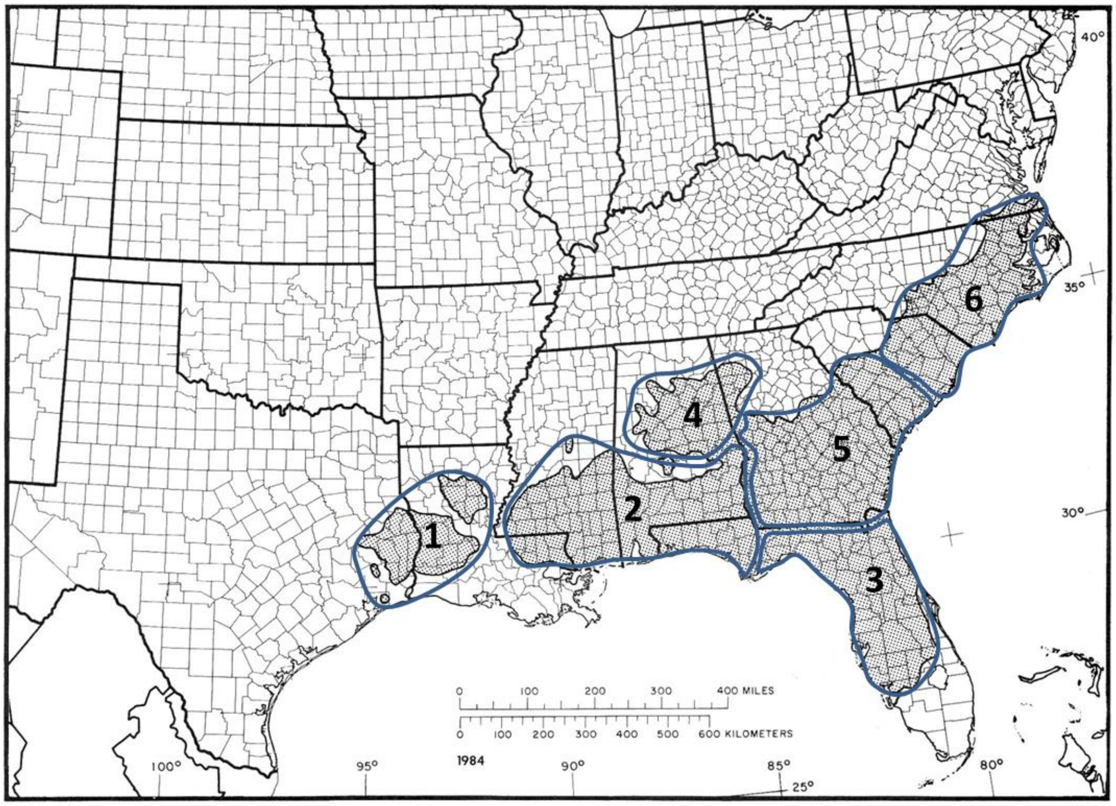

Figure 1—*The native range of longleaf pine.*

Figure 1--The native range of longleaf pine (gray shade) with proposed longleaf pine breeding zones (ecozones, numbered 1 to 6).

The breeding and testing scheme needs to meet a few criteria, including maintaining large enough base populations to insure gene diversity and the potential for artificial and natural selection (Eriksson and others 1993), and be small and simple enough to allow participants to manage and maintain the program indefinitely. We meet the first criteria by starting with 200 unrelated trees per ecoregion and six ecoregions (Echt and others 2011, Gapare and others 2008, Lawrence and others 1995). These trees are provided by the participants and should meet some minimum standard of condition in their native environment. This could be a tree that has tested well in a previous tree improvement program or one that exhibits a good phenotype and cone crop potential. Of course, availability of fresh cones or viable seeds is needed as well, since the scheme relies on progeny performance in the next generation for forward selection. Seeds will be germinated and seedlings transplanted into progeny tests that provide family and individual-tree performance information serving as a basis for selection to establish both seedling and clonal seed orchards. All progeny will be from wind pollinations, further simplifying the participants' workload with recurrent selection. Figure 2 depicts the open-pollinated, recurrent breeding and selection scheme (adapted from Simmonds 1979) through three generations with seed orchard development and seed deployment options at each generation.

Participatory tree improvement offers the forestry community an opportunity for collectively developing and maintaining genetic materials for tree planting and gene conservation. It can range from a highly centralized program that may resemble university-industry-state cooperatives to a decentralized program where essentially all components of the program are managed and conducted by the participants. An intermediate approach seems most likely to succeed in species such as longleaf pine that have some economic and ecologic importance but not to the

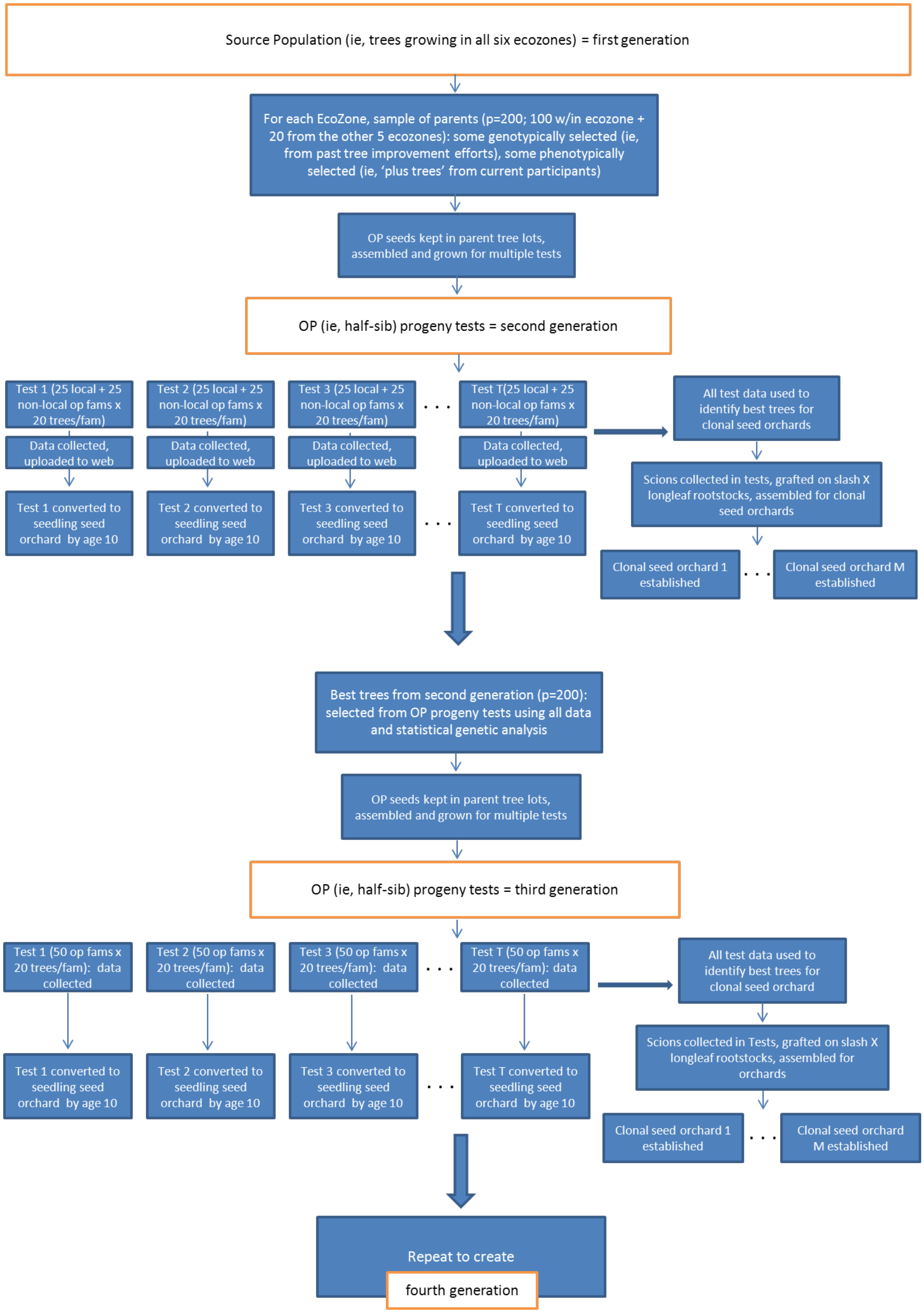

Figure 2--Proposed participatory breeding plan for longleaf pine.

level of the major industrialized species such as loblolly pine or the very high-valued hardwoods such as black walnut (*Juglans nigra* L.). In such a program, we can anticipate a geneticist coordinating the program with limited technical and clerical support and a network of participating landowners. These landowners could range from private individuals or companies to non-governmental organizations to public agencies, where they fall into three basic functions: plus-tree identification, performance testing, and cone/seed production. Plus-tree identification participants collect open-pollinated cones from their favorite tree(s) and document the tree(s) in the project's online database. The coordinating geneticist will ensure that the plus trees meet a basic phenotypic standard (i.e. desirable traits or physical attributes) and originate on a variety of site types (i.e. uplands, sand hills, flatwoods) and their respective plant associates (e.g. wiregrass, bluestems, saw palmetto). Performance testing participants (a landowner or a group of landowners) will identify potential sites, choose the test planting sites, establish tests, and grow the trees. This group will also collect the needed data and collect cones for establishing next-generation performance tests. Cone/seed production participants (a landowner or group of landowners) will produce seed from the rogued performance tests (i.e. seedling seed orchards), or they may establish a grafted (i.e. clonal) seed orchard to produce the highest genetic quality seed. Clear, reliable, and timely communications facilitated by the internet and mobile/cloud computing offer new opportunities for distributed forest research and monitoring and tree improvement should benefit from this development.

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LITERATURE CITED

Atlin, G.N.; Cooper, M.; Bjørnstad, Å. 2001. A comparison of formal and participatory breeding approaches using selection theory. Euphytica. 122: 463-475.

Ceccarelli, S.; Grando, S. 2007. Decentralized-participatory plant breeding: an example of demand driven research. Euphytica. 155: 349-360.

Craul, P.J.; Kush, J.S.; Boyer, W.D. 2005. Longleaf pine site zones. Gen. Tech. Rep. SRS–89. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 23 p.

Echt, C.S.; Crane, B.S.; Nelson, C.D. 2011. Establishing restoration seed reserves in national forest system seed orchards. In: Nelson, C.D.; Rousseau, R.J.; Yuceer, C., eds. Proceedings of the 31st southern forest tree improvement Conference: 46-49.

Eriksson, G.; Namkoong, G.; Roberds, J.H. 1993. Dynamic gene conservation for uncertain futures. Forest Ecology and Management. 62:15-37.

Gapare, W.J.; Yanchuk, A.D.; Aitken, S.N. 2008. Optimal sampling strategies for capture of genetic diversity differ between core and peripheral populations of *Picea sitchensis*. Conservation Genetics. 9:411-418.

Griffith, G.E.; Omernick, J.M.; McGinley, M. 2008. Ecoregions of the United States - level IV (EPA). Environmental Information Coalition, National Council for Science and the Environment. http://www.eoearth.org/article/Ecoregions_of_the_United_States-Level_IV_(EPA). [Date accessed: August 24, 2011].

Lawrence, M.J.; Marshall, D.F.; Davies, P. 1995. Genetics of genetic conservation. I. Sample size when collecting germplasm. Euphytica. 84: 89-99.

Potter, KM; Hargrove WW. 2012. Determining suitable locations for seed transfer under climate change: a global quantitative method. New Forests. 43.5-6: 581-599.

Schmidtling, RC. 2001. Southern pine seed sources. Gen. Tech. Rep. SRS-44. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 25 p.

Simmonds, N.V. 1979. Principles of crop improvement. New York: Longman, Inc. 408 p.

IS THE FOOTPRINT OF LONGLEAF PINE IN THE SOUTHEASTERN UNITED STATES STILL SHRINKING?

Christopher M. Oswalt, Christopher W. Woodall, and Horace W. Brooks[1]

Longleaf pine (*Pinus palustris* Mill.) was once one of the most ecologically important tree species in the southern United States. Longleaf pine and the accompanying longleaf forest ecosystems covered vast swaths of the South. Longleaf forests covered an estimated 92 million acres at their peak distribution and represented one of the most extensive forest ecosystems in America. Only a fraction of longleaf pine ecosystems remain today. Remaining longleaf pines are scarce compared to the historical extent and are spread among eight southern states in largely fragmented stands. Additionally, scientists, conservationists, and land managers hypothesize that much of the remaining acreage is in poor condition. Therefore, it is imperative that longleaf pine receive continuous focused monitoring.

Data from the Forest Inventory and Analysis (FIA) program of the USDA Forest Service were used to explore both temporal and spatial trends in longleaf pine population dynamics of the southern U.S. in order to better understand the potential future of the species. In 2010, 4.3 million acres of longleaf pine-dominated forests existed across the South. According to broad-scale inventory data, longleaf pine forests are still fewer than those found as recently as in the 1970s. Upon visual analysis, considerable contraction of the geographic distribution of longleaf pine has occurred from 1970 to 2010 (Oswalt and others 2012).

We used FIA data collected in the 1970s to compare the geographic extent of the species to the distribution in 2010 (table 1) and quantify any range contraction or expansion along all range boundaries. We combined an outer range analysis pioneered by Woodall and others (2009) with a longitudinal/latitudinal band analysis (Zhu and others 2012). Outer ranges were identified by the 90th and 10th percentiles for latitude and longitude. Each outer range was then dissected into 1^{o} latitudinal or longitudinal bands. Comparisons between the mean latitude or longitude of 1970 and 2010 longleaf pine stems ≥ 1 inch diameter at breast height (d.b.h.) were made using Welch's two-sample t-tests and significant differences noted using an α = 0.05.

Table 1--Inventory year for each state in the southern United States where longleaf pine observations were used for a comparison of the 1970 and 2010 decades

State	1970s	2010
Alabama	1972	2011
Florida	1970	2010
Georgia	1972	2011
Louisiana	1974	2010
Mississippi	1977	2010
North Carolina	1974	2011
South Carolina	1978	2011
Texas	1975	2010
Virginia	1977	2010

Significant contraction occurred on all boundaries. In the northern outer range, contraction occurred primarily in the eastern latitudes. The largest range contraction in the northern outer range was approximately 78 kilometers (48 miles). In the southern outer range, there were two longitudinal bands (-94 and -84) where longleaf pine was observed in the 1970s but not in 2010. Southerly expansion of approximately 33 kilometers (20 miles) occurred in the far eastern portion of the range in Florida. In the western outer range, significant contractions eastward [48 kilometers (30 miles)] occurred in the lower latitudes of the range. The eastern outer range was relatively stable with both minimal contractions and expansions observed between the two time periods (1970

[1]Research Forester, USDA Forest Service, Southern Research Station, Knoxville, TN 37917; Research Forester, USDA Forest Service, Northern Research Station, St. Paul, MN 55108; and Forestry Technician, USDA Forest Service, Southern Research Station, Knoxville, TN 37917.

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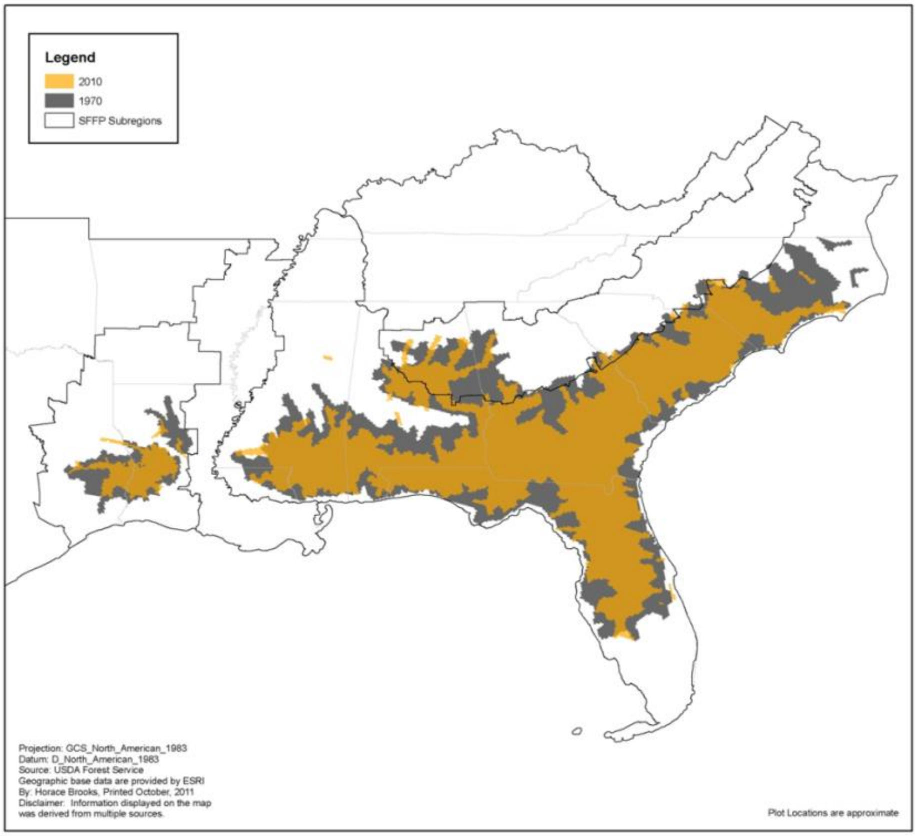

Figure 1--Comparison of the longleaf pine (*Pinus palustris* Mill.) "footprint" of observed stems ≥ 1 inch d.b.h. for the 1970 and 2010 periods using Forest Inventory and Analysis data.

versus 2010). The contractions are apparent when the "footprint" of the 1970 observations was coupled with the 2010 observations (fig. 1). Within the contracting footprint of longleaf pine, 32 percent of counties with longleaf pine forests in the 1970s have experienced significant (70 percent or greater) losses of longleaf pine-dominated forest area. While losses occurred throughout the longleaf range, heaviest losses occurred along the Gulf Coast and in western Louisiana. Results indicate that considerable longleaf pine loss can be attributed to the conversion to the loblolly pine (*Pinus taeda* L.) forest type. Longleaf pine forests represent an important resource in the context of the southern U.S. forest. While many conservation efforts have been and are currently active in efforts to re-establish longleaf forests across the South, these valuable forests have continued to decline over recent decades.

LITERATURE CITED

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.

Woodall, C.W.; Oswalt, C.M.; Westfall, J.A. [and others]. 2009. An indicator of tree migration in forests of the eastern United States. Forest Ecology and Management. 257: 1434-1444.

Zhu, K.; Woodall, C.W.; Clark, J.S. 2012. Failure to migrate: lack of tree range expansion in response to climate change. Global Change Biology. 18: 1042-1052.

FOREST CANOPY REDUCTION AND BREEDING BIRD RESPONSES: TREATMENT- AND TEMPORAL-DEPENDENT PATTERNS

Brandie K. Stringer, Yong Wang, and Callie J. Schweitzer[1]

Abstract--We examined the effects of oak regeneration forest management treatments on territorial density of breeding forest birds. The study area was located on the southern end of the mid-Cumberland Plateau in northern Jackson County, AL. Fifteen 4-ha stands were treated in 2001 with one of five target overstory retention (percentage) treatments: 0 (clearcut); 25; 50; 75; and 100 (control). In 2010-2011, the residual trees in the initial 25, 50, 75, and 100 (control) percent retention stands were harvested, and three new controls were added, which resulted in three forest stand cohorts: (1) mature (control, not harvested for 50 to 70 years); (2) 10-year-old regenerated clearcut; and (3) final harvest of the shelterwood prescriptions (25, 50, 75, and 100 percent retention stands in 2001). Breeding songbirds were surveyed 9 to 10 times per year during the peak of breeding season (April to July) of 2002, 2003, 2010, 2011, and 2012. Territory mapping was used based on detections in each year in each stand. Two-way analysis of variance (ANOVA) and Tukey test were used to compare average territory density among treatments and years. Results of temporal responses of two breeding songbird conservation concern species, Kentucky Warbler (*Geothlypis formosa*) and Worm-eating Warbler (*Helmitheros vermivorum*), to treatments showed that responses were treatment-dependent. Territory density of Kentucky Warbler (an interior-edge species) showed positive response to 50 percent retention stands. Worm-eating Warbler (an interior species) territory density responded positively to control stands.

INTRODUCTION

The importance of songbird conservation has received much attention in the ecological community, as many species are declining (Askins 2000, Sauer and others 2011). Particularly vulnerable are Neotropical migrants that breed during summer months in North America and overwinter in South and Central America, the Caribbean, and Mexico (Cornell Laboratory of Ornithology 2007). A subset of these migrants is in danger of becoming listed as threatened, having been included on the conservation concern list (Rich and others 2004) of Partners in Flight (PIF), an international bird conservation organization. Included in the PIF list are Kentucky Warbler (*Geothlypis formosa*) and Worm-eating Warbler (*Helmitheros vermivorum*) (Rich and others 2004). Research suggests that habitat alteration or loss (Askins 2000, Sauer and others 2011) and brood parasitism by cowbirds (*Molothrus* spp.), in which female cowbirds lay eggs in nests of other host species (Robinson and others 1995), are major contributors in the decline of these species and other Neotropical migratory songbirds.

Forest songbirds that breed in the southeastern United States have been particularly vulnerable as a result of massive agricultural clearing in the late 19th and early 20th centuries, followed by pine plantation reforestation, coupled with fire suppression during most of the 20th century (Rauscher 2004). These practices decreased periodic disturbance and contributed to the loss of forest understory (North American Bird Conservation Initiative U.S. Committee 2011) necessary to maintain populations of many bird species (Rich and others 2004). Despite numerous studies focusing on breeding grounds (Carpenter and others 2011, Lesak and others 2004), it is still not clear how anthropogenic disturbance affects the conservation of many species (Rich and others 2004). What has been suggested is that conservation efforts by land managers should include the production and maintenance of early successional habitat (Rich and others 2004).

Land managers of southern upland hardwood forests are faced with multiple challenges. Oak (*Quercus* spp.) is an important component of these forest systems (Hicks and others 2004). Oaks are mostly shade intolerant (Loftis 1990, Schweitzer and Dey 2011, Stringer 2006) and as juveniles expend much energy in root development and less on height growth (Hicks and others 2004). This causes oaks to be out-competed by shade-tolerant species such as sugar maple (*Acer saccharum* Marsh.) and light-responsive species such as yellow-poplar (*Liriodendron tulipifera* L.) (Schweitzer and Dey 2011). Surface fires may enable the domination of oaks over their competitors, but fire suppression during the 20th century and an increasing abundance of browsing deer (*Odocoileus virginianus*) have been major culprits in decreased oak reproduction in

[1]Master's Candidate and Professor, respectively, Alabama A&M University, Department of Biological and Environmental Sciences, Normal, AL 35762; and Research Forester, USDA Forest Service, Southern Research Station, Huntsville, AL 35801.

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southern upland hardwood forests (Hicks and others 2004). The resultant mature, even-aged forests with closed canopies do not provide adequate light conditions that are required for oak reproduction (Larsen and Johnson 1998). Because of the difficulties of recruiting oak into competitive and dominant positions within mixed species forests, managing these forests to sustain demand for oak and other wildlife-dependent (McShea and Healy 2002) and economically-valuable hardwoods has become the focus of some forest researchers (e.g. Rathfon 2011, Schweitzer 2004, Schweitzer and Dey 2011). To address the lack of sufficient competitive oak reproduction, managers are altering regeneration techniques to facilitate development of sustainable levels of oak stocking. One such technique is the oak shelterwood method, which involves removing undesirable midstory species, allowing more light penetration to the forest understory and encouraging oak seedling height growth (Loftis 1990, Stringer 2006).

Forest managers also need to monitor forest ecosystem health and consider the effects of management actions on forest birds, since forest health is partially maintained by birds (Connor and others 1999, Greenberg and others 2001, Traveset and others 2001), and songbirds rely on forests for suitable habitat (Lesak and others 2004, Rich and others 2004, Wang and others 2006). PIF and other conservation organizations recognize the need for examination of the influence of silvicultural practices on forest songbirds (Rich and others 2004). Studying forest bird response to anthropogenic disturbance can help managers formulate strategies that will help maintain healthy populations of forest birds. Studies observing the long-term effects of shelterwood treatments on songbirds have been limited (e.g. Augenfeld and others 2008) and have not included songbird response to multiple ages (e.g. clearcut + oak-shelterwood + control). Since Alabama contains the third largest commercial forest industry and the second largest private forest landholdings in the nation (Alabama Forestry Commission 2009), it is important that land managers be equipped with the knowledge and tools to sustain healthy forests for multiple uses.

Clearcut Harvesting

Clearcut harvesting often abruptly changes species composition (Chambers and others 1999, Costello and others 2000, Lesak and others 2004) and provides habitat for early successional species (Costello and others 2000, Lesak and others 2004). Previous studies examining forest bird response to management practices often compared clearcut stands with untreated stands (Conner and Adkisson 1975, Thompson and others 1992, Thompson and Fritzell 1990). Relatively short-term studies revealed that clearcuts often negatively impacted the forest-interior-nesting Worm-eating Warbler (Conner and Adkisson 1975, Gram and others 2003, Thompson and others 1992). The interior-edge-nesting Kentucky Warbler appeared less sensitive to clearcut stands and re-inhabited these stands in a relatively short amount of time (Thompson and others 1992). Intermediate- and long-term responses of forest birds to clearcut harvesting are limited (e.g. McDermott and Wood 2009).

Shelterwood Harvesting

Studies showing the effects of shelterwood harvesting on forest bird species were short-term and limited, and few compared shelterwood stands with clearcut stands (Annand and Thompson 1997, King and DeGraaf 2000, Lesak and others 2004). Even less studied are the responses of conservation priority species (Rich and others 2004) to both shelterwood and clearcut treatments in the Cumberland Plateau region (e.g. Lesak and others 2004), especially in multiple phases. By the second year after shelterwood harvesting, Lesak (2004) noticed favorable response to all shelterwood treatments by Kentucky Warbler and similar responses by Worm-eating Warbler to treatments of 25 percent overstory retention. However, there still remain gaps in intermediate- and long-term temporal response patterns of these and other forest bird species on the Cumberland Plateau.

Certain forest management prescriptions can create early successional habitat that may be beneficial for some early successional avian species (Askins 2000, Lesak and others 2004) and interior-edge species, such as Kentucky Warbler. However, other species, such as Worm-eating Warbler, may suffer immediate decline due to loss of mature forest habitat (Askins 2000). With sufficient re-growth of the forest vegetation structure, disturbance may eventually provide sustainable resources for mature forest species (Lesak and others 2004). In this study, we examined how different stages of regenerating forests contributed to habitat creation conducive to particular birds. In alignment with recommendations made by PIF for conservation priority forest birds, we monitored the territory density of selected species with high conservation concern such as Kentucky Warbler and Worm-eating Warbler (Rich and others 2004) on the forest stands with canopy reduction treatments.

METHODS

Study Site

The study area was located on the southern end of the mid-Cumberland Plateau in northern Jackson County, AL. Average temperature in this region is approximately 13 °C and average annual precipitation is 149 cm (Smalley 1982). Two sites were used, one located at Miller Mountain (MM) (34” 58’ 30” N, 86” 12’ 30” W) and one at Jack Gap (JG) (34” 56’ 30” N, 86” 04’ 00” W) (Lesak 2004). Miller Mountain has a southern to southwestern aspect and JG has a northern aspect. Elevation for both sites varies between 260 to 520 m, with slopes ranging from 15 to 30 percent. Upland hardwood is the primary forested land cover type, composed mainly of oak and hickory (*Carya* spp.) with yellow-poplar, sugar maple, red maple (*Acer rubrum* L.), and American beech (*Fagus grandifolia* Ehrh.) (Schweitzer 2004).

Experimental Design

In 2000, two blocks of 10 stands were established at JG and one block of 5 stands at MM, for a total of three blocks and 15 stands. Each stand was approximately square in shape and 4 ha in size (Schweitzer 2004). Each block was approximately 20 ha, for a total study area (in 2010) of 60 ha. All stands were arranged adjacently within each block (Schweitzer 2004). In 2011, 3 new stands (2 at JG and 1 at MM) were added to the study for a total of 18 stands comprising a 72-ha study site. Five 0.01-ha circular plots were established in each stand for vegetation characterization. All trees 3.8 cm in diameter at breast height (d.b.h.) were monumented with a permanent tag, and species and d.b.h. were recorded.

Silvicultural treatments--The USDA Forest Service Southern Research Station-initiated study consisted of a randomized complete block replicate design with five overstory retention treatment units replicated three times. The five treatments consisted of stands with the following target overstory retention percentages: 0 (clearcut); SW25; SW50; SW75; and SW100 (control, not harvested for 40 years or greater), and were blocked by location. Trees in the SW25 to 50 percent retention stands were marked and retained according to species (preference was given to oak, ash, and persimmon), vigor, class, and crown position (Schweitzer 2004). Initial tree harvesting was accomplished by chainsaw felling and grapple skidding. Stands of SW75 percent retention were treated with an herbicide injection (Arsenal®, containing active ingredient imazapyr) in 2001 to remove the midstory. The intermediate harvest intensity stands (SW25-75 percent retention) were initiated as shelterwood stands to investigate the relationship between varying levels of overstory retention and oak regeneration. Stands of 0 percent retention were treated with a single clearcut prescription, which was completed in 2002 (Schweitzer 2004).

All 15 original stands were allowed to grow for approximately 10 years prior to final harvest in 2011, when the overstory canopies in these stands were removed. At that time, residual trees in the initial overstory retention (percentage) treatment stands of SW25, SW50, SW75, and SW100 (control) were harvested with the use of chainsaw felling and grapple skidding. Stands of SW100 percent retention (controls for bird surveys) were also harvested. Three new stands, Control2010 (not harvested for 50 to 70 years), were added for bird survey controls. Once 2011 treatments were completed and new control stands were established, there were approximately three forest stand cohorts: mature (Control2010), 10-year-old regenerated clearcut, and new shelterwood harvests (SW25, SW50, SW75, and SW100 percent retention treatment in 2001).

Basal area data--Pre-treatment basal area data were collected from five measurement plots that were systematically located within each treatment unit (Schweitzer 2004). At these locations, all overstory trees ≥ 14.2-cm d.b.h. were tallied to estimate initial basal area in 2001, prior to treatment. In 2002, post-treatment residual trees were measured and used to determine residual basal area and overstory retention. Following treatments, these locations were also used for measuring canopy cover with a handheld spherical densitometer.

Avian territory mapping--Three transects were established within each stand. Each transect was spaced evenly across its width and parallel with the slope; there were ≤ 50 m between transects, and between transects and stand boundaries. Each transect had marked reference points every 25 m which were used to facilitate bird territory mapping (Lesak 2004).

During the peak of breeding season in 2002 and 2003 (late April to beginning July), the territory spot-mapping technique (Bibby and others 2000, International Bird Census Committee 1970, Ralph and others 1993, Williams 1936) was used to survey the original 15 forest stands 10 times, an appropriate amount of sampling effort to obtain reliable breeding-territory data (Ralph and others 1993). Between approximately 05:30 and 10:30 every survey morning, one block of five units was visited. Each block was visited once before the next rotation was started. Order of visits within blocks was assigned uniquely for each rotation to ensure that all stands were visited equal amounts at all possible morning times, and approximately 1 hour was spent in each unit. Stand entrance and exit locations were also rotated. During surveys, territorial defense displays (songs, calls, distraction displays) and other behaviors indicative of an active territory were recorded on topographic maps and were later transposed onto transparency films (Lesak 2004). These steps were replicated in 2010. Each stand was visited 9 to 10 times between approximately May 1 and June 30 in 2010, 2011, and 2012. In 2011 and 2012, topographic data maps were scanned for territory delineation in ArcMap (ArcGIS 10.0). All territory spot-mapping was conducted by one or two observers per season, and stands were rotated between observers to reduce inter-observer bias.

Due to unfavorable weather conditions, logging activity was postponed, and final harvest of shelterwood and control stands at JG could not be completed prior to the bird breeding season of 2011. As a result, only 7 of the original 15 stands (all 5 original stands at MM and 3 prior clearcut stands at JG) plus the 3 new control stands were available for bird surveys. This resulted in inadequate replications. Therefore, these data were omitted from this analysis; control stand data were retained and used.

Statistical Analyses

Bird data from 2002, 2003, 2010, and 2012 were analyzed using two-way factorial repeated-measures analysis of variance (ANOVA, SPSS v 20.0) with year as the within-subjects repeated factor and effects of block and treatment as between-subject factors. Tree data were used to compute basal area (BA) and stems per ha (SPH) of each stand, and statistics were applied as with the bird data. Normality and homogeneity of variance assumptions of the data were tested with Shapiro-Wilk and Levene test, respectively, at the significance level of $p > 0.05$. We tested the effect of treatment, year, and their interactions on the territory density. If there was no interaction between year and treatment, we directly examined the year and treatment main effect, followed by Tukey multiple comparison tests. If there was an interaction between year and treatment, we examined treatment effect by each year separately, followed by Tukey multiple comparisons for each year.

RESULTS AND DISCUSSION

Tree Basal Area and Stems per Hectare

Following initial treatment application, a gradient of three different BAs resulted (Schweitzer and Dey 2011). For the SW100 and SW75, BA remained relatively unchanged from pre-treatment values. The herbicide treatment targeted the midstory trees, and few overstory trees were treated. The number of SPH of trees ≥ 3.8 cm d.b.h. was reduced from 791 to 290 in the herbicide treatment; conversely, the SW100 treatment gained stems, from 719 SPH pre-treatment to 779 SPH in 2010. The clearcut resulted in the lowest residual BA, 1.4 m^2 ha^{-1}, but went from 217 SPH immediately post-harvest to 1,054 SPH in 2010. The clearcut BA was significantly different from all other treatments except the SW25 ($F_{4,2}$ = 37.31, P = 0.0001). The SW50 left 10.1 m^2 ha^{-1}, which was significantly different from all other treatments except the SW25. The SW25 had a residual BA of 8.5 m^2 ha^{-1}, and this was only significantly different than the control and 75SW. By 2010, the SW50 and SW25 had 871 and 1,102 SPH and BAs of 13.5 m^2 $ha^{-1}$and 11.9 m^2 $ha^{-1,}$ respectively. Following the second phase of the shelterwood treatments, all merchantable stems were harvested from the SW100, SW75, SW50, and SW25. The SW75 had the lowest SPH (53) and BA (3.3 m^2 ha^{-1}), reflective of the missing midstory (deadened 10 years prior and thus no sprouting). The SW50 and SW25 had SPH of 1,392 and 1,412, respectively, although the residual BA of the SW50 was 10.6 compared to 6.3 m^2 ha^{-1} for the 25SW m^2 ha^{-1}. Both the SW50 and SW25 were dominated by trees 4 cm d.b.h.; there were no SPH > 28 cm in the SW25,

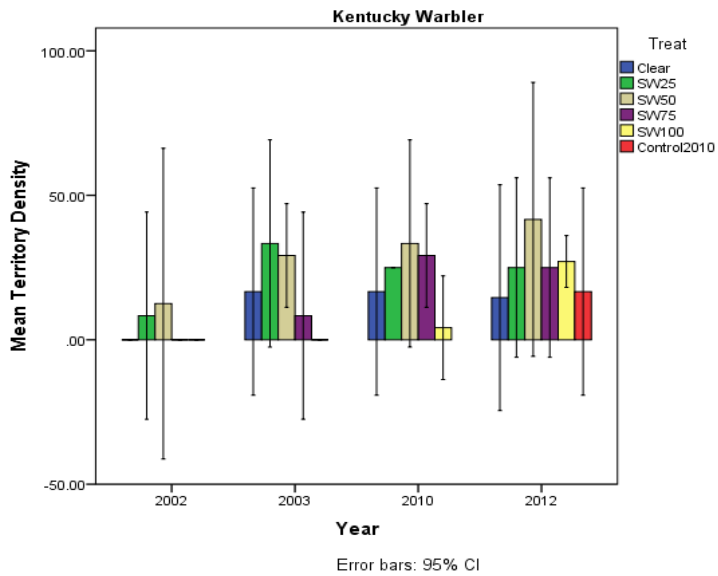

Figure 1--Mean territory density [(territories/4-ha)*100] of Kentucky Warbler (*Geothlypis formosa*) displaying treatment and year effects in 2002, 2003, 2010, and 2012 at Miller Mountain and Jack Gap, Jackson County, AL. Treatments were as follows: clear = clearcut in 2002; SW25 = shelterwood with 25 percent target overstory retention level in 2001; SW50 = shelterwood with 50 percent target overstory retention level in 2001; SW75 = shelterwood with 75 percent target overstory retention level in 2001; SW100 = shelterwood with 100 percent target overstory retention level in 2001, used as bird survey control for 2002, 2003, and 2010; Control2010 = control installed for 2012 bird surveys. Residuals in all SW stands were harvested after 2010 bird surveys.

and 8 SPH of trees > 28cm in the SW50. These treatments were predominately populated with new stump sprouts. Control2010 had a BA of 26.9 m^2 ha^{-1} with 845 SPH, and the now 11-year old clearcut had 9.8 m^2 ha^{-1} with 2,481 SPH.

Kentucky Warbler

Treatment (P = 0.001) and year (P = 0.0001) had significant effects on the territory density but not the treatment and year interactions (table 1). Territory density increased in all treatments except control between 2002 and 2010 after treatments were implemented and was the highest in SW50 stands in 2010 before the residual trees were removed from the treatment stands (fig. 1). After the removal of the residual trees in 2010, there was a sharp increase in territory density in the initial control (SW100) stands in 2012. The territory density was the highest at initial SW25 stands and was the lowest in clearcut stands and new control (Control2010) stands in 2012.

Worm-eating Warbler

The treatment (P = 0.0003) and the interaction between year and treatment (P = 0.051) significantly affected the territorial density of Worm-eating Warbler (table 2). The clearcut negatively affected the density of this species after the initial treatment, but the density gradually increased between 2002 and 2012, though it was still lower than the density in control stands. Other shelterwood treatments (SW25, SW50, and SW70) all had lower densities of Worm-eating Warblers compared to the control but higher than the clearcut after initial treatment and in 2010 before the removal of the residual trees. After the removal of residual trees, the territory density in initial control (SW100), SW75, and SW25 all declined, with the initial control (SW100) and SW75 having the faster rate of decline. Territory density was highest in control (SW100) stands for all years except 2012, when it was highest in Control2010 stands (fig. 2). In 2012, approximately 10 years after initial harvest, clearcut stands had the second highest territory density.

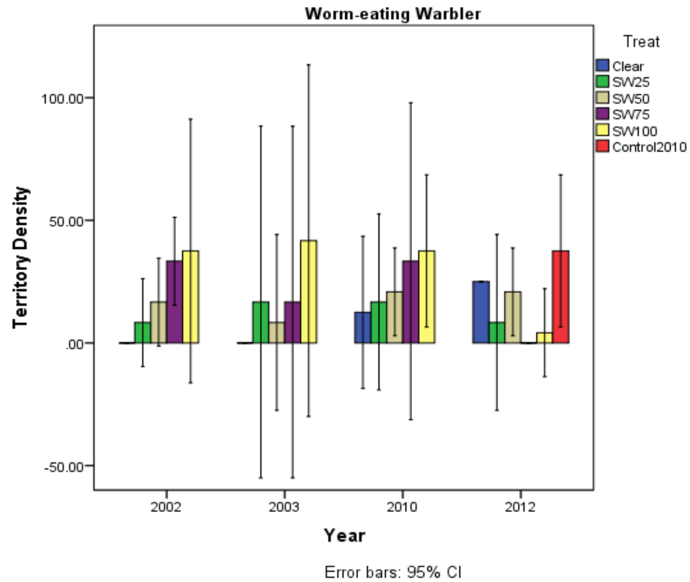

Figure 2--Mean territory density [(territories/4-ha)*100] of Worm-eating Warbler (*Helmitheros vermivorum*) displaying treatment*year interaction in 2002, 2003, 2010, and 2012 at Miller Mountain and Jack Gap, Jackson County, AL. Treatments were as follows: clear = clearcut in 2002; SW25 = shelterwood with 25 percent target overstory retention level in 2001; SW50 = shelterwood with 50 percent target overstory retention level in 2001; SW75 = shelterwood with 75 percent target overstory retention level in 2001; SW100 = shelterwood with 100 percent target overstory retention level in 2001, used as bird survey control for 2002, 2003, and 2010; Control2010 = control installed for 2012 bird surveys. Residuals in all SW stands were harvested after 2010 bird surveys.

Table 1--Tests of between-subjects effects for Kentucky Warbler (*Geothlypis formosa*) territory density at Miller Mountain and Jack Gap, Jackson County, AL[a]

Source	Type III sum of squares	df	Mean square	F	Significance
Treat	0.380	5	0.076	5.179	0.001
Year	0.416	3	0.139	9.439	0.0001
Treat x Year interactions	0.200	13	0.015	1.046	0.428
Error	0.646	44	0.015		

[a]Tests of between-subjects effects for Kentucky warbler and dependent variable: territory density.

Table 2--Tests of between-subjects effects for Worm-eating warbler (*Helmitheros vermivorum*)

Source	Type III sum of squares	df	Mean square	F	Significance
Treat	0.673	5	0.135	5.866	0.0003
Year	0.151	3	0.050	2.194	0.103
Treat x Year interactions	0.582	13	0.045	1.949	0.051
Error	0.987	43	0.023		

[a]Tests of between-subjects effects for Worm-eating Warbler and dependent variable: territory density.

Creating and altering habitat conditions through active forest management is an on-going practice in the Cumberland Plateau region. Forest structure and composition has changed with disturbance or lack thereof, causing suitable bird habitat to also fluctuate. We are more acutely aware of the need to have a mosaic of habitats, spatially and temporally distributed, in order to sustain the highest diversity of birds (Augenfeld and others 2008, Lesak 2004, Lesak and others 2004). Studying bird response to active forest management is often done within a short time frame, with the loss of knowledge about how successional dynamics influence habitat creation and subsequent bird activities. Although certainly not long-term, this on-going study is providing insight into those dynamics as related to two bird species of conservation concern.

In productive forest systems such as those found on more mesic escarpment sites on the Cumberland Plateau, the vegetation response to disturbance is vigorous (Schweitzer and Dey 2011). Within 8 years following the initial harvests, the SW25, SW50, and clearcuts had a densely occupied under- and midstory. The response of the targeted bird species in this study was reflective of this growth. From these data, it appears that SW50 is most favorable for the territory density of the Kentucky Warbler. For the Worm-eating Warbler, controls created the most favorable habitat. This report does not take into account overall breeding success (i.e. nesting success/failure) of the targeted bird species; future analyses of these data will help provide more information on how the bird species are responding to the treatments in this study. In addition, the examination of the understory composition data will allow us to identify possible relationships that exist between the targeted bird species and their respective flora.

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LITERATURE CITED

Alabama Forestry Commission. 2009. Forest statistics: Alabama forest facts. (http://www.forestry.state.al.us/forest_facts.aspx). [Date accessed: September 30, 2010].

Annand, E.M; Thompson, F.R., III. 1997. Forest bird response to regeneration practices in central hardwood forests. Journal of Wildlife Management. 61: 159-171.

Askins, R.A. 2000. Restoring North America's birds, lessons from landscape ecology. New Haven, CT: Yale University Press. 288 p.

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.

Bibby, C.J.; Burgess, N.D.; Hill, D.A.; Mustoe, S. 2000. Bird census techniques. 2d ed. London: Academic Press. 302 p.

Carpenter, J.P.; Wang, Y.; Schweitzer, C.; Hamel, P. 2011. Avian community and microhabitat associations of cerulean warblers in Alabama. Wilson Journal of Ornithology. 123: 206-217.

Chambers, C.L.; McComb, W.C.; Tappeiner, J.C. 1999. Breeding bird responses to three silvicultural treatments in the Oregon coast range. Ecological Applications. 9: 171-185.

Conner, R.N.; Adkisson, C.S. 1975. Effects of clearcutting on the diversity of breeding birds. Forestry. 73: 781-785.

Connor, E.F.; Yoder, J.M.; May, J.A. 1999. Density-related predation by the Carolina chickadee, *Poecille carolinensis*, on the leaf-mining moth, *Cameraria hamadryadella* at three spatial scales. Oikos. 87: 105-112.

Cornell Laboratory of Ornithology. 2007. All about birds. http://www.birds.cornell.edu/AllAboutBirds/studying/migration/patterns. [Date accessed: August 12, 2011].

Costello, C.A.; Yamasaki, M.; Pekins, P.J. [and others]. 2000. Songbird response to group selection harvests and clearcuts in a New Hampshire northern hardwood forest. Forest Ecology and Management. 127: 41-54.

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.

Greenberg, C.H.; Smith, L.M.; Levey, D.J. 2001. Fruit fate, seed germination, and growth of an invasive vine: an experimental test of 'sit and wait' strategy. Biological Invasions. 3: 363-372.

Hicks, R.R., Jr.; Conner, W.H.; Kellison, R.C.; Van Lear, D. 2004. Silviculture and management strategies applicable to southern hardwoods. In: Rauscher, H.; 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: 51-62.

International Bird Census Committee. 1970. An international standard for a mapping method in bird census work recommended by the International Bird Census Committee. Audubon Field Notes. 24: 722-726.

King, D.I.; DeGraaf, R.M. 2000. Bird species diversity and nesting success in mature, clearcut and shelterwood forest in northern New Hampshire, USA. Forest Ecology and Management. 129: 227-235.

Larsen, D.R.; Johnson, P.S. 1998. Linking the ecology of natural oak regeneration to silviculture. Forest Ecology and Management. 106: 1-7.

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.

Lesak, A.A.; Wang, Y.; Schweitzer, C.J. 2004. Songbird community variation among five levels of overstory retention in northern Alabama. 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: 11-17.

Loftis, D.L. 1990. A shelterwood method for regenerating red oak in the southern Appalachians. Forest Science. 36(4): 917-929.

McDermott, M.E.; Wood, P.B. 2009. Short- and long-term implications of clearcut and two-age silviculture for conservation of breeding forest birds in the central Appalachians, USA. Biological Conservation.142: 212-220.

McShea, W.J.; Healy, W.M. 2002. Oak forest ecosystems: ecology and management for wildlife. Baltimore: Johns Hopkins University Press. 432 p.

North American Bird Conservation Initiative U.S. Committee. 2011. The state of the birds 2011 report on public lands and waters. Washington, DC: U.S. Department of the Interior. 48 p.

Ralph, C.J.; Geupel, G.R.; Pyle, P. [and others]. 1993. Handbook of field methods for monitoring landbirds. Gen. Tech. Rep. PSW-144. Albany, CA: U.S. Department of Agriculture Forest Service, Pacific Southwest Research Station. 41 p.

Rathfon, R.A. 2011. Midstory removal methods tested for timely release of newly established oak seedlings under oak shelterwoods. 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: 521-534.

Rauscher, H.M. 2004. A history of southern forest science, management, and sustainability issues. In: Rauscher, H.; 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: 3-4.

Rich, T.D.; Beardmore, C.J.; Berlanga, H. [and others]. 2004. Partners in Flight North American landbird conservation plan. Ithaca, NY: Cornell Lab of Ornithology. 25 p. http://www.partnersinflight.org/cont_plan/. [Date accessed: March 1, 2012].

Robinson, S.K.; Rothstein, S.I.; Brittingham, M.C. [and others]. 1995. Ecology and behavior of cowbirds and their impact on host populations. In: Martin, T.E.; Finch, D.M., eds. Ecology and management of neotropical migratory birds. New York: Oxford University Press: 428-460.

Sauer, J.R.; Hines, J.E.; Fallon, J.E. [and others]. 2011. The North American breeding bird survey, results and analysis 1966-2009, version 3.23.2011. http://www.mbr-pwrc.usgs.gov/bbs/. [Date accessed: February 15, 2012].

Schweitzer, C.J. 2004. First-year response of an upland hardwood forest to five levels of overstory tree retention. 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: 287-291.

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: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 58 p.

Stringer, J. 2006. Oak shelterwood: a technique to improve oak regeneration. Professional hardwood notes. Publ. SP 676. Knoxville, TN: University of Tennessee Extension. 8 p.

Thompson, F.R., III; Dijak, W.D.; Kulowiec, T.G.; Hamilton, D.A. 1992. Breeding bird populations in Missouri Ozark forests with and without clearcutting. Journal of Wildlife Management. 56: 23-30.

Thompson, F.R., III; Fritzell, E.K. 1990. Bird densities and diversity in clearcut and mature oak-hickory forest. Res. Pap. NC-293. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 7 p.

Traveset, A.; Riera, N.; Mas, R.E. 2001. Passage through bird guts causes interspecific differences in seed germination characteristics. Functional Ecology. 15: 669-675.

Wang, Y.; Lesak, A.A.; Felix, Z.; Schweitzer, C.J. 2006. Initial response of an avian community to silvicultural treatments in the southern Cumberland Plateau, Alabama, USA. Integrative Zoology. 3: 126-129.

Williams, A.B. 1936. The composition and dynamics of a beech-maple climax community. Ecological Monographs. 6: 317-408.

THE REPLACEMENTS OF AMERICAN CHESTNUT: A RANGE-WIDE ASSESSMENT BASED ON DATA FROM FOREST INVENTORY AND PUBLISHED STUDIES

G. Geoff Wang and Huifeng Hu[1]

American chestnut [*Castanea dentata* (Marshall) Borkh.] was a dominant or co-dominant species in the upland forests of much of eastern North America (Braun 1950). It was one of the most widely distributed species in eastern deciduous forest, with its main range from southern Maine all the way to central Mississippi (fig. 1). Stands with American chestnut as a significant canopy tree covered 84 million acres, typically in mixture with oaks (*Quercus* spp.) and other deciduous species, but may have occasionally formed a pure stand (Wang and others 2013). American chestnut was also the largest tree in the eastern deciduous forests. It might have grown up to 130 feet tall and 10 feet in diameter and lived up to 600 years of age. No single species today has achieved its abundance in the eastern deciduous forest.

American chestnut was an economically important tree species. It was a good timber species, with a tall and straight truck, and provided 25 percent of all harvested hardwood timber at the turn of the 20th century. The wood of American chestnut was of very high quality, with straight grain and high resistance to decay. American chestnut produced tasty nuts, consumed by humans and animals, with a dependable crop every year. Another byproduct was tannin, which was important to leather industry. Recent studies also support that American chestnut was the fastest growing tree species in the eastern deciduous forest. So it is not surprising that American chestnut was regarded as one of the most promising trees for forest management by the Society of American Foresters.

However, the great potential of American chestnut to modern forestry was never realized because of an introduced fungus, known as chestnut blight (*Cryphonectria parasitica*). The fungus was likely introduced with imported nursery stock from Asia and was first detected in 1904 (Wang and others 2013). The fungus entered the cambium and girdled trees, effectively shutting down the water supply from the roots; it killed almost every tree it infected. After its first detection in 1904, chestnut blight spread quickly. By the 1950s, the infestation covered the entire range of American chestnut, killing nearly all American chestnut trees. However, because of its prolific vegetative regeneration and the fact that blight does not affect small stems, American chestnut did not become extinct. In stands previously supporting American chestnut, small chestnut sprouts persist and manage to survive repeated top kills. In open areas, American chestnut can even produce fruit before it is attacked and killed by blight.

Which species have replaced American chestnut in the canopy 60 to 100 years after the blight? Although there were many local studies (fig. 1), a range-wide assessment has not yet been conducted. In this study, we performed such an assessment based on the most recent data obtained through the Forest Inventory and Analysis (FIA) program of the USDA Forest Service as well as data from published studies. Specifically, we assessed the current status of American chestnut based on FIA data, determined what species have replaced American chestnut based on FIA data and published studies, and compared the results.

We acquired data from the most recent (2001 to 2009) measurements of Phase II FIA plots. Study plots were selected if: (1) one or more subplots have live or dead American chestnut trees, saplings, or seedlings; (2) all subplots are

[1]Professor, Clemson University, School of Agricultural, Forest and Environmental Sciences, Clemson, SC 29634; and Associate Professor, Chinese Academy of Sciences, Institute of Botany, Beijing, China.

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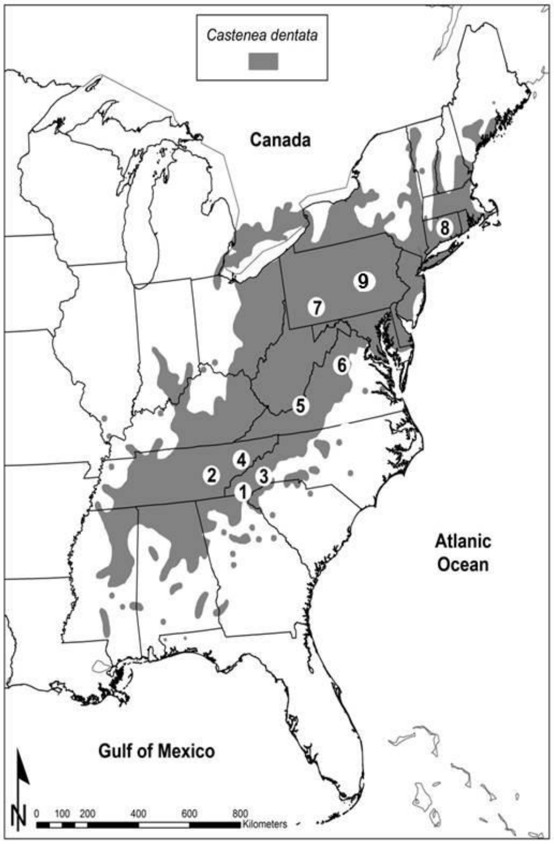

Figure 1--Data from the following published studies are used in the our analysis: (1) Nelson 1955, Elliot and Swank 2008; (2) Myers and others 2004; (3) Keever 1953; (4) Woods and Shanks 1959; (5) Stephenson 1974, McCormick and Platt 1980, Stephenson 1986; (6) Karban 1978; (7) Mackey and Sivec 1973; (8) Korstian and Stickel. 1927; (9) Aughanbaugh 1935.

forested; and (3) subplots are located within the contiguous part of the native range of American chestnut distribution. As a result, we selected 512 plots from 16 states. Importance value of each species was calculated separately for trees (≥ 12.7 cm d.b.h.) and saplings (2.54 to 12.7 cm d.b.h.) on each plot. Diameter distributions of living American chestnut stems (fig. 2) reveal that the majority of American chestnut live stems are seedlings and saplings, with only 5 percent of the plots having any stems ≥ 12.7 cm d.b.h, (4 inches) which is defined as a tree in our study. American chestnut was primarily replaced by oaks [predominantly chestnut oak (*Q. prinus* L.) and northern red oak (*Q. rubra* L.)], followed by red maple (*Acer rubrum* L.), hickories (*Carya* spp.), and other mesophytic species [e.g., yellow poplar (*Liriodendron tulipifera* L.)] (table 1).

Pines (*Pinus* spp.) rarely replaced American chestnut although some replacements by eastern white pine (*P. strobus* L.) were observed. Based on the current population structure, the dominance of oaks is not likely sustainable while the dominance of red maple and other mesophytic species will increase in the future (table 1). This result confirms that oaks are not regenerating well relative to their dominance in the canopy as widely reported in eastern deciduous forests (e.g., Nowaki and Abrams 2008).

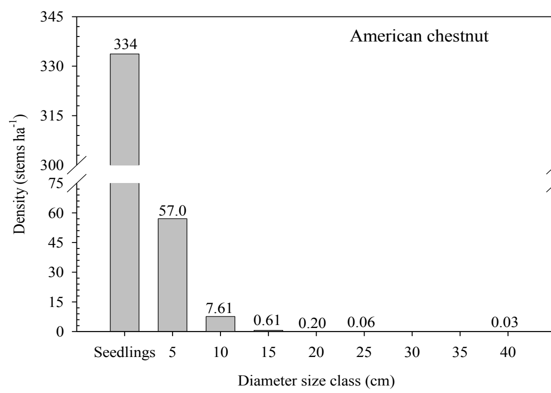

Figure 2--Diameter size-class distribution of living American chestnut stems based on the most recent FIA phase II plot data from 16 states.

We compiled data derived from 13 published studies on composition change after blight in nine different locations (fig. 1). Among them, five studies had both pre- and post-measurements. Importance value was either obtained directly or calculated from reported data. Results on the replacements of American chestnut based on the analysis of these published studies are given in table 1. These results collaborate well with results from the analysis of FIA data (table 1).

In conclusion, American chestnut is rarely found in the overstory but remains abundant in the understory. Oaks are the predominant replacement species for American chestnut, followed by maples. Maples and other shade-tolerant species are currently replacing oaks in the canopy. Results from the analysis of published studies support findings based on FIA data, giving confidence to our range-wide assessment.

Table 1--Importance values of major tree species currently occurring in the stands with American chestnut as a significant component before blight. Species are separated into trees (d.b.h. $\geq$ 12.7 cm) and saplings (2.54 $\leq$ d.b.h. < 12.7 cm)

	-------Tree-------		----Sapling----	
Species	FIA[a]	LIT[b]	FIA	LIT
Quercus spp.	48.7	46.9	14.4	16.9
Acer spp.	15.5	13.9	28.6	26.9
Pinus spp.	7.3	1.4	4.7	0.7
Liriodendron tulipifera	4.8	2.7	3.0	0.1
Oxydendrum arboreum	3.9	1.3	5.9	1.2
Carya spp.	3.7	6.8	2.8	1.8
Betula spp.	3.5	3.9	4.3	2.7
Nyssa sylvatica	2.0	1.7	9.7	4.5
Tsuga canadensis	1.5	2.0	2.1	0.5
Fagus grandifolia	1.1	0.2	2.3	0.8
Magnolia spp.	1.0	0.4	1.3	0.8
Castanea dentata	0.2	1.8	4.6	3.6
Sassafras albidum	0.6	0.6	2.8	7.0
Amelanchier spp.	0.4	0.5	2.5	2.4
Prunus spp.	0.6	2.5	2.0	1.6
Cornus florida	0.1	2.2	1.7	1.5

[a]FIA = analyses based on forest inventory and analyses program.
[b]LIT = analyses based on published studies.

LITERATURE CITED

Aughanbaugh, J.E. 1935. Replacement of the chestnut in Pennsylvania. Bull. 54. Harrisburg, PA: Commonwealth of Pennsylvania, Department of Forests and Waters. 38 p.

Braun, E.L. 1950. Deciduous forests of eastern North America. Philadelphia, PA: Blakiston. 596 p.Elliott, K.J.; Swank, W.T. 2008. Long-term changes in forest composition and diversity following early logging (1919-1923) and the decline of American chestnut (*Castanea dentata*). Plant Ecology. 197: 155-172

Karban, R. 1978. Changes in an oak-chestnut forest since the chestnut blight. Castanea. 43: 221-228.

Keever, C. 1953. Present composition of some stands of former oak-chestnut forest in the southern Blue Ridge Mountains. Ecology. 34: 44-54.

Korstian, C.F.; Stickel, P.W. 1927. The natural replacement of blight-killed chestnut in the hardwood forests of the northeast. Journal of Agricultural Research. 34: 631-648.

Mackey, H.E.; Sivec, N. 1973. The present composition of a former oak-chestnut forest in the Allegheny Mountains of western Pennsylvania. Ecology. 54: 915–919.

McCormick, J.F.; Platt, R.B. 1980. Recovery of an Appalachian forest following the chestnut blight or Catherine Keever-you were right! American Midland Naturalist. 104: 264-273.

Myers, B.R.; Walck, J.L.; Blum, K.E. 2004. Vegetation change in a former chestnut stand on the Cumberland Plateau of Tennessee during an 80-year period (1921-2000). Castanea. 69: 81-91.

Nelson, T.C. 1955. Chestnut replacement in the southern highlands. Ecology. 36: 352-353

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the eastern United States. BioScience. 58: 123-138.

Stephenson S.L. 1974. Ecological composition of some former oak-chestnut communities in western Virginia. Castanea. 39: 278-286.

Stephenson, S.L. 1986. Changes in a former chestnut-dominated forest after a half century of succession. American Midland Naturalist. 116: 173–179

Wang, G.G.; Knapp, B.O.; Clark, S.L.; Mudder, B.T. 2013. The silvics of *Castanea dentata* (Marsh.) Borkh., American chestnut, Fagaceae (beech family). Gen. Tech. Rep. SRS-GTR-173. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 18 p.

Woods, F.W.; Shanks, R.E. 1959. Natural replacement of chestnut by other species in the Great Smoky Mountains National Park. Ecology. 40: 349–361.

Methods

Moderator:

Jim Guldin
USDA Forest Service
Southern Research Station

USE OF THE FAKOPP TREESONIC ACOUSTIC DEVICE TO ESTIMATE WOOD QUALITY CHARACTERISTICS IN LOBLOLLY PINE TREES PLANTED AT DIFFERENT DENSITIES

Ralph L. Amateis and Harold E. Burkhart[1]

Abstract–A Fakopp TreeSonic acoustic device was used to measure time of flight (TOF) impulses through sample trees prior to felling from 27-year-old loblolly pine (*Pinus taeda* L.) plantations established at different planting densities. After felling, the sample trees were sawn into lumber and the boards subjected to edgewise bending under 2-point loading. Bending properties evaluated included MOR (modulus of rupture) and MOE (modulus of elasticity). Regression methods were used to relate these bending properties to the TOF measurements collected from the standing trees. Results suggest TOF measurements alone are unlikely to be adequate when predicting MOR and MOE of loblolly pine lumber from standing trees growing at different planting densities.

INTRODUCTION

Southern pine plantation forests are among the most productive in the world. Much of this productivity is attributable to the application of intensive management practices that have dramatically increased growth rates over the past 25 years or so (Fox and others 2007, Stanturf and others 2003). These practices have resulted in growth rates of as much as 8 green tons per acre per year on some sites with a potential of as much as 10 green tons per acre per year (Stanturf and others 2003). While overall growth rates can be considered a general estimate of productivity, particular product objectives must be taken into account for specific assessments of productivity. For example, Amateis and Burkhart (2012) found a significant correlation between initial planting density and the amount of wood produced for particular product objectives. Where the production of solid wood products is the goal, the quantity of sawtimber can be maximized by planting fewer trees at wider spacings. When pulpwood is the management objective, planting densities of about 680 trees per acre have been found optimal.

While the application of intensive management practices has been shown to increase productivity, the ultimate value of wood harvested from loblolly pine (*Pinus taeda* L.) plantations is determined by both quantity and quality. In a recent study that examined the effect of planting density on wood production, Amateis and Burkhart (2013) found a positive relationship between planting density and visual grade of lumber off the green chain. Higher planting densities produced a greater proportion of board feet per acre graded 2 or better. The better grades associated with higher planting densities offset, to some degree, the smaller quantities of lumber produced at higher densities. In short, the concern is that many of the intensive management practices that have boosted pine plantation productivity in recent years may be having offsetting, or negative impact on some measures of wood quality (Biblis and others 1993, Clark and others 2008, Pearson and Gilmore 1980). Therefore, to properly assess value per acre, managers need estimates of both quantity and quality (Amateis and Burkhart 2013; Clark and others 2004, 2010).

Estimates of wood quantity are obtained from field measurements or from models that provide predictions of harvested wood from stand and tree variables that are highly correlated with tree volume and stand yield. Either way, destructive sampling is not needed to estimate wood quantities. Estimates of wood quality, however, are not so easily obtainable because they rely on information about wood properties that are not readily observable or directly measureable. Therefore, since the development of acoustic devices, interest has grown in their use as a non-destructive and efficient tool to collect information about wood quality characteristics. The fundamentals of acoustic wave propagation through trees and logs have been known for some time. From them, devices implementing two measurement methods have been developed: time of flight (TOF) for standing trees and resonance for logs (Wang 2011). While the devices and methods differ, both rely on the movement of acoustic waves through wood as a

[1]Senior Research Associate and University Distinguished Professor, respectively, Virginia Polytechnic Institute and State University, Department of Forest Resources and Environmental Conservation, Blacksburg, VA 24061.

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convenient way to obtain information on wood properties. However, since wood characteristics can vary greatly within a tree or log, both radially and longitudinally, the application of TOF and resonance methods for assessing them has not been straight forward or standardized. For example, Mahon and others (2009) studied the use of TOF methods for identifying trees with high stiffness. They determined a number of potential paths that an acoustic wave might travel both longitudinally and radially and tested some alternatives and the effect of probe placement on TOF measures. They found that placing probes on opposite faces of the tree minimizes the variance of TOF measures. In a study using both TOF and resonance methods, Grabianowski and others (2006) applied a three-probe TOF system where the three probes were lined up on the same side of the tree. Results showed significant correlation between TOF and resonance measurements, but the TOF measurements were on average higher suggesting that the TOF method is measuring outerwood TOF rather than a weighted average TOF, which includes bark, given by the resonance method. In order to bring measurements from both methods in line, Wang and others (2005) and Mora and others (2009) developed models for adjusting observed tree TOF values to equivalent log velocities.

In a comprehensive genetic study using Monterey pine (*Pinus radiata* D. Don) from two seedlots, Matheson and others (2002) applied several different acoustic wave devices to standing trees and logs to correlate acoustic wave velocity with average MOE of sawn lumber obtained by machine stress grading. They found that the speed of sound along felled logs was sufficiently correlated with average MOE of the boards to allow log segregation into MOE classes. A weaker, but still significant correlation was found between standing trees and board MOE for a control seedlot, but, unexpectedly, no significant correlation was exhibited for the orchard seedlot.

The purpose of this study was to relate TOF measurements made on standing trees from a loblolly pine spacing trial to the wood properties of boards recovered from those trees.

DATA

In the spring of 1983, a set of loblolly pine spacing trials was established in the Piedmont of Virginia and the Coastal Plain areas of Virginia and North Carolina. Three replicates containing 16 treatment plots in each replicate were established at each of four sites. Only the Lower Coastal Plain site in Virginia was used for this study. Of the 16 treatment plots available from each replicate, only the square planting densities of 6- by 6-feet (1,210 trees per acre), 8- by 8-feet (681 trees per acre), and 12- by 12-feet (302 trees per acre) were utilized. Thus, three plots of each of the three spacing treatments were used for the study reported here. Additional information about the study design, a history of the research results over the life of the study, and final growth and yield results at age 25 can be found in Amateis and Burkhart (2012).

At age 27 data were collected on all live standing trees from the three square spacing treatment plots at the three replicates on the Lower Coastal Plain site. Standing tree measurements included diameter at breast height (d.b.h.), total height, height to live crown, and product categorization as either sawtimber or pulpwood. Trees with a d.b.h. of at least 8.6 inches with a 16-foot butt log free of damage and disease, and straight such that a line connecting the center of the stem at any two points above the stump would not lie outside the tree bole were categorized as sawtimber.

From each treatment plot up to six trees were selected at random from the pool of qualifying sawtimber quality trees. For plots that did not have six qualifying trees in a particular replicate, additional qualifying trees from the same treatment plot in the other two replicates were used. An attempt was made to obtain 18 sample trees representing each treatment plot for a total of 54 sample trees. Due to the lack of suitable sample trees from the 1,210 trees per acre plots, however, the total number of sample trees was 48.

A Fakopp TreeSonic acoustic device was used to measure time of flight (TOF) acoustic velocities on the 48 sample trees prior to felling. The device consists of two probes, one with a start sensor and the other with a stop sensor, a portable scopemeter, and a hammer. The two probes were angled toward each other and inserted through the bark and cambium into the sapwood. The lower probe was positioned about 0.7 m above the ground and the upper probe 1 m directly above the lower probe on the same face of the stem. Three measures of TOF were obtained for each tree and averaged to obtain one estimate of TOF acoustic velocity for each tree.

Sample trees were felled and the butt log was cut from each tree, skidded off the plots, and subsequently transported to the sawmill. A portable Wood-Mizer sawmill was used to recover lumber from the sample logs. All logs were milled employing the same operator and sawn into structural 2-inch and non-structural 1-inch boards according to mill-run specifications. All 2-inch boards were graded green as #1, #2 or #3 by a certified grader according to national grading rules for structural light framing lumber (National Grading Rule Committee 2004). Following green grading, all #1 and #2 boards were racked on drying sticks and air-dried for several months reaching a stable moisture content of about 20 percent (table 1).
Out of the 2-inch boards green graded #1 and #2, 1 inside board (containing pith) and 1 randomly selected outside board per log were chosen for testing of mechanical properties resulting in 87 test boards. The center 80 inches of each board were extracted. If the sample board had a width of 6 inches or 8 inches, one edge was randomly selected and set down on the carriage. The board was then ripped to a width of 4 inches. Thus, each sample board was of dimension 2 x 4 x 80 inches.

The boards were kiln-dried to approximately 10 percent moisture content. Following drying, each board was re-graded by the same certified grader. Of the 87 #2 and better green graded boards, the drying process resulted in 29 boards being down-graded to #3. All test boards were then shipped to the Timber Products Inspection (TPI) lab. Upon receipt at TPI, the boards were stored at ambient conditions of 75 °F and 50 percent relative humidity awaiting testing.

In the lab, each board was subjected to edge-wise bending in accordance with ASTM D4761 testing protocols. A static bending load was applied to each board across a span of 68 inches (span to depth ratio of 17) at a testing speed that applied stress at a rate of 4,000 psi per minute. Deflection data were collected until the load reached 1,000 pound force, whereupon each board was loaded until failure. Data collected included strength (MOR), stiffness (MOE) and cause of failure (table 1).

METHODS AND RESULTS

Pearson correlation coefficients relating TOF to the average MOE and MOR across boards by tree were computed. Neither MOE nor MOR was significantly correlated with TOF (probability values of 0.1420 and 0.8253, respectively). A number of different linear regression equations were explored to link TOF with mechanical properties. No regressions for predicting MOE or MOR that included TOF as a regressor were significant.

Another effort was made to correlate the TOF with MOE of the outer and inner boards separately in the anticipation that board position might be a useful indicator variable. These regressions were also not significant.

Table 1–Mean values of time-of-flight (TOF) acoustic velocities, modulus of elasticity (MOE) and modulus of rupture (MOR) (standard deviation in parentheses) for sample boards sawn from 48 sample trees at age 27 years. Standard deviations are in parentheses

Planting density	Board position[a]	Number sample trees	Number sample boards	TOF	MOE	MOR
trees ac^{-1}				*m/s*	*10^6 psi*	*psi*
1,210	Inside		10		1.345 (0.374)	6,749 (1,966)
	Outside		11		1.286 (0.276)	7,155 (1,796)
	All	12	21	3468 (339)	1.314 (0.319)	6,961 (1,843)
681	Inside		16		0.994 (0.316)	4,945 (1,774)
	Outside		15		1.116 (0.293)	5,626 (2,062)
	All	18	31	3580 (318)	1.053 (0.306)	5,275 (1,918)
302	Inside		16		0.832 (0.240)	4,713 (1,375)
	Outside		19		0.954 (0.261)	6,192 (2,067)
	All	18	35	3305 (327)	0.898 (0.255)	5,516 (1,912)

[a]Inside boards contained pith.

DISCUSSION

Efforts to correlate TOF measurements collected on sample trees using the Fakopp TreeSonic acoustic device were unsuccessful with these data. There are a number of possible reasons for this. One of the most obvious is that our sample size consisting of one inside and one outside board per tree was small while the variability of MOE was very large. As Feeney and others (1998) have noted, the great variability in the structure and properties of wood in the radial, tangential, and longitudinal directions make the measurement of the propagation of ultrasound through wood a challenging task. A larger sample size might have yielded more positive results.

We also had no information about the position of each board within the tree in relation to the position of the Fakopp probes when the TOF measurements were taken. Obviously, additional TOF measurements with alternative probe placements might have led to better correlations with MOE. Another source of variation in our data was the impact of knots on mechanical properties. The correlation of MOE with wood density weakens when testing full-sized sample boards as done in this study (Biblis and others 2004). Grabianowski and others (2004) have also documented the impact that local climatic effects, such as wind direction and speed, can have on acoustic velocity measurements. No measurements of or accounting for any of these effects was made.

At present, sawlog markets are not rewarding landowners for growing loblolly pine trees with particular mechanical properties. However, future markets may account for trees that can produce lumber meeting overall stiffness grades or MOE values that meet particular design specifications. In order for that to happen, it is likely that a more robust method of non-destructively estimating mechanical properties will be needed.

ACKNOWLEDGMENTS

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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: 11-11.

Amateis, R.L.; Burkhart, H.E. 2013. Relating quantity, quality, and value of lumber to planting density for loblolly pine plantations. Southern Journal of Applied Forestry. 37: 97-101.

Biblis, E.J.; Brinker, R.; Carino, H.F.; McKee, C.W. 1993. Effect of stand age on structural properties and grade compliance of lumber from loblolly pine plantation timber. Forest Products Journal. 43: 23-28.

Biblis, E.J.; Meldahl, R.; Pitt, D.; Carino, H.F. 2004. Predicting flexural properties of dimension lumber from 40-year-old loblolly pine plantation stands. Forest Products Journal. 54: 109-113.

Clark, A.; Daniels, R.F.; Jordan, L. [and others]. 2010. Impact of initial spacing on yield per acre and wood quality of unthinned loblolly pine at age 21. 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: 333-337.

Clark, A., III; Jordan, L.; Schimleck, L.R.; Daniels, R.F. 2008. Effect of initial planting spacing on wood properties of unthinned loblolly pine at age 21. Forest Products Journal. 58: 78-83.

Clark, A.; Saucier J.R.; Baldwin V.C. [and others]. 2004. Effect of initial spacing and thinning on lumber grade, yield and strength of loblolly pine. Forest Products Journal. 44: 14-20.

Feeney, F.E.; Chivers, R.C.; Evertsen, J.A.; Keating, J. 1998. The influence of inhomogeneity on the propogation of ultrasound in wood. Ultrasonics. 36: 449-453.

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

Grabianowski, M.B.; Manley, B.; Walker, J. 2004. Impact of stocking and exposure on outerwood acoustic properties of *Pinus radiata* in Eyrewell Forest. New Zealand Journal of Forestry. 49: 13-17.

Grabianowski, M.; Manley, B; Walker, J.C.F. 2006. Acoustic measurements on standing trees, logs and green lumber. Wood Science and Technology. 40: 205-216.

Mahon, J.M., Jr.; Jordan, L.; Schimleck, L.R. [and others]. 2009. A comparison of sampling methods for a standing tree acoustic device. Southern Journal of Applied Forestry. 33: 62-68.

Matheson, A.C.; Dickson, R.L.; Spencer, D.J. [and others]. 2002. Acoustic segregation of *Pinus radiata* logs according to stiffness. Annals of Forest Science. 59: 471-477.

Mora, C.R.; Schimleck, L.R.; Isik, F. [and others]. 2009. Relationships between acoustic variables and different measures of stiffness in standing *Pinus taeda* trees. Canadian Journal of Forest Research. 39: 1421-1429.

National Grading Rule Committee. 2004. National grading rule for dimension lumber. Germantown, MD: U.S. Department of Commerce. 32 p.

Pearson, R.G.; Gilmore, R.C. 1980. Effect of fast growth rate on the mechanical properties of loblolly pine. Forest Products Journal. 30: 47-54.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: 26-31.

Wang, X. 2011. Fundamentals of acoustic measurements on trees and logs and their implication to field application. In: Divos, F., ed. Proceedings of the 17th international symposium on nondestructive testing and evaluation of wood. Sopron, Hungary: University of West Hungary: 25-33.

Wang, X.; Ross, R.J.; Carter, P. 2005. Acoustic evaluation of standing trees-recent research development. In: Broker, F-W., ed. Proceedings of the 14th international symposium on nondestructive testing of wood. Aachen, Germany: Shaker Verlag: 455-465.

LOBLOLLY PINE GENETICS VERIFICATION TEST FOR PRIVATE NON-INDUSTRIAL LANDOWNERS

Jon E. Barry, Victor L. Ford, and John L. Trauger[1]

Abstract--Forest industry has invested in loblolly pine (*Pinus taeda* L.) genetics to improve growth, branching, and form. Until recently, superior families were destined for industry lands with little of this superior genetic material available for other landowners. Seedlings of superior families are now available to non-industrial private forest (NIPF) landowners at a greater cost than seedlings from first generation selections. The University of Arkansas Division of Agriculture conducts variety and verification trials for agronomic crops and serves as a source of unbiased information on crop genetics. This study applies the same philosophy to loblolly pine genetics by testing growth, survival, form, and branching in southwest Arkansas. The families included Weyerhaeuser first and second-generation select families, four select families and two MCP families from ArborGen, and unrogued and rogued first generation seed orchard mixes from the Arkansas Forestry Commission (AFC). These plantings were established in January 2009 in southwest Arkansas on a fine sandy loam soils. Ten families were planted in 100-tree plots divided among four blocks. Weeds were controlled the first 2 years after planting. Tree height measurements and survival counts have been made annually. Survival after three growing seasons was lowest (61percent) in the AFC unrogued mix and highest (91percent) in an MCP family, averaging 84 percent. This is impressive considering 2011 was a record drought year. Height after three growing seasons was least (2.8 feet) in the Weyerhaeuser first generation family and greatest (4.2 feet) in the Arbor-Gen AG-34 family, averaging 3.4 feet. Branching and form cannot be evaluated until after crown closure.

INTRODUCTION

The variety of loblolly pine (*Pinus taeda* L.) seedlings available to landowners in Arkansas has dramatically increased during the last 10 years. While the vendors of these seedlings have tested the families, no head-to-head verification trials conducted by unbiased entities are known to us. Companies that produce the seedlings claim that tests have shown that they do well in Arkansas, but performance under field conditions in southwest Arkansas is unknown. In keeping with our mission to provide objective science-based information to landowners in Arkansas about the performance of these families, we established a variety trial to compare the performance of 10 commonly available loblolly pine families. These families range from an unrogued woods-run family to an elite MCP family. Unfortunately, one family was available in only limited quantities and had to be excluded from the statistical analysis. We believe this to be the first third-party loblolly pine variety trial to be conducted in Arkansas. The variety trial was established at the University of Arkansas Division of Agriculture's Southwest Research and Extension Center (SWREC) near Hope, AR. This test will be used as a variety trial for loblolly pine and as a demonstration to educate landowners about the seedling families that are available in southern Arkansas.

MATERIALS AND METHODS

The site chosen for the study had been occupied by a pine plantation for approximately 25 years. That stand was harvested in late 2007 and early 2008. Site preparation during 2008 consisted of removing the existing loblolly pine plantation, then windrowing and burning the slash.

Layout

Four thousand bare root 1-0 loblolly pine seedlings representing 10 families (table 1) from three sources were planted in 40 plots of 100 trees each (fig. 1) on the Spencer Tract at SWREC in early March 2009. The seedlings were planted on a 10- by 10-foot spacing by a work crew provided by the Arkansas Department of Corrections. Herbicides were used to control competing vegetation as needed during the first 2 years of the study (table 2). In March and April 2011, hardwood stems and volunteer pine stems were mechanically removed from the plots. Soils underlying the site are predominantly Savannah fine sandy loam on 3 to 8 percent slopes. A small portion of the soils along the north edge of the stand are Sacul fine sandy loam on 3 to 8 percent slopes.

The 40 plots of trees were divided into four blocks of 10 plots each. The intention was that

[1]Extension Forester, Director and Research Associate, respectively, Arkansas Forest Resources Center, Southwest Research and Extension Center, Hope, AR 71801.

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Table 1--Loblolly pine families represented in the variety trial

Producer	Family name	Number of blocks	Source classification
ArborGen	MCP-22M	7	Atlantic coast MCP elite
ArborGen	MCP-29	1	Atlantic coast MCP
ArborGen	34	4	Atlantic coast advanced (2^d gen.)
ArborGen	520	4	Arkansas elite
ArborGen	522	4	Arkansas elite
ArborGen	579	4	Arkansas advanced (2^d gen.)
AFC	Imp. mix	4	Unknown
AFC	Nat.mix	4	Unknown
Weyerhaeuser	1^{st} gen.	4	Unknown
Weyerhaeuser	2^d gen.	4	Unknown

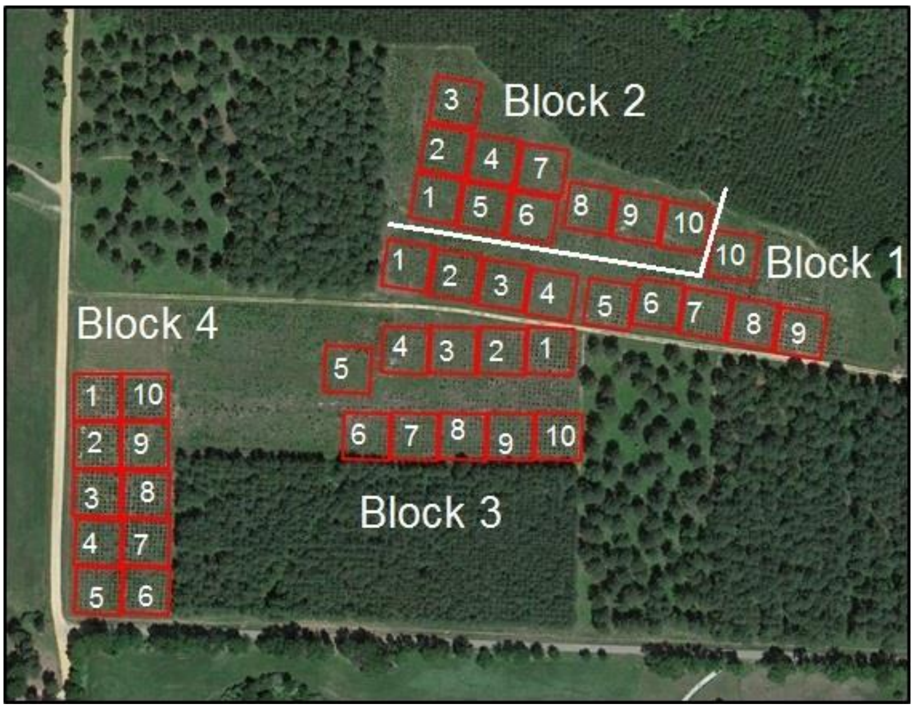

Figure 1--Block and plot layout for the loblolly pine variety trial on the Spencer Tract of the Southwest Research and Extension Center, Hope, AR.

Table 2--Herbicide treatments applied to control competing vegetation

Date	Treatment
March 2009	Velpar, 2 quarts + Oust, 2 ounces per acre
August 2009	Arsenal, 16 ounces + Escort, 1 ounce per acre
March 2010	Velpar, 2 quarts + Escort, 1 ounce per acre
Oct/Nov 2011	Basal bark application of Remedy® to hardwoods and volunteer pines

each of the 10 families of seedlings was represented by one plot in each block. The layout had to be modified as will be explained in the statistical design section. The interior 36 trees of each plot were measurement trees with a 2-tree buffer surrounding the measurement trees for an initial total of 1,440 measurement trees divided among the 40 plots.

This report covers measurements taken by a crew from SWREC on 27 November 2012, after four growing seasons. For each tree the crew recorded tree presence (i.e. survival), tree height measured in feet and tenths of feet, and presence of stem defects such as crooks, sweeps, forks, broken tops, and animal damage. Stem defects were recorded only for those trees alive at the time of sampling. A stem defect rate was calculated for each plot by dividing the number of stems with defects by the number of live stems in the plot.

Statistical Design

The statistical design was originally conceived as a balanced complete random block design consisting of 10 families with four blocks of each family. However, a shortage of the ArborGen MCP-29 family seedlings and the consequent substitution from the ArborGen MCP-22M family resulted in a design that could not be effectively analyzed. ArborGen MCP-29 was represented by only one plot, while ArborGen MCP-22M was represented by two plots in each of three blocks and one plot in the fourth block for a total of seven plots. This problem was resolved by discarding ArborGen MCP-29 from the study and removing three of the ArborGen MCP-22M plots from the study. One ArborGen MCP-22M plot was selected by coin toss from each of the blocks containing two plots of that family and was eliminated from the study. The Arkansas Forestry Commission (AFC) Natural Seedling Mix plot was removed from Block 1 due to poor survival related to poor planting. This resulted in a final statistical design that was an unbalanced complete random block for the survival test. The statistical design for height growth and defect rates were balanced. The data were analyzed using PROC GLM with a Duncan's Multiple Range Test within SAS 9.2 for Windows (SAS Institute 2008). Survival was analyzed as the percent surviving in each plot (n = 4, except the AFC Natural Mix family where n = 3). Defect was analyzed as the percent of trees with a defect in each plot (n = 4). Tree height was analyzed as the average height of all trees within each family with error assigned to blocks (n varies by family).

Climatic Conditions

Climatic conditions in southwest Arkansas during the term of the study have been characterized by extremes of rainfall and higher than normal temperatures during 3 of the 4 years. This has resulted in growing conditions that were less than ideal for loblolly pine seedlings.

Normal annual rainfall accumulation at SWREC is 56.3 inches per year (iAIMS Climatic Data 2013). During 2009, SWREC received 85.4 inches of rainfall. This was followed by 35.0 inches of rainfall in 2010, by 40.0 inches in 2011, and by 44.0 inches in 2012 (Unpublished data. 2012. S. Pote, Administrative Assistant, Southwest Research and Extension Center, 362 Hwy 174N, Hope, AR 71801).

Normal mean high temperatures for June, July, August and September are 88.9, 92.1, 91.9, and 84.4 ^{o}F, respectively. During the course of this study, June mean high temperatures were 90.2, 92.6, 94.3, and 91.5 ^{o}F for 2009, 2010, 2011, and 2012, respectively. July mean high temperatures were 91.1, 92.8, 98.8, and 94.0 ^{o}F for the 4 years. August mean high temperatures were 88.4, 98.2, 101.5, and 93.4 ^{o}F for the 4 years. September mean high temperatures were 80.7, 89.4, 89.0, and 86.7 ^{o}F for the 4 years.

RESULTS

Height Growth

Mean overall height growth for the study was 10.7 feet. Mean height growth for the different seedling families ranged from 8.9 feet to 12.1 feet (table 3). Three ArborGen varieties (34, MCP-22M, and 520) produced the greatest mean height growth. The least height growth was produced by the Weyerhaeuser First Generation seedlings and the AFC Natural Seedling Mix.

Survival

Mean overall survival rate for the study was 82.0 percent. Mean survival rate for the different families ranged from 57.4 percent to 89.6 percent (table 4). Due to very low survival (25 percent), we excluded Block 1 Plot 4 from the survival calculations. Block 1 Plot 4 was the first plot planted by the ADC crew, and we believe the low survival in that plot was partly a result of

Table 3--Mean height growth for the nine loblolly pine families. Means followed by the same letter are not significantly different at $p < 0.05$

Family	Height	Sample size
	--feet--	
ArborGen 34	12.1a	125
ArborGen MCP-22M	12.0a	125
ArborGen 520	11.8a	118
ArborGen 522	11.2b	113
ArborGen 579	10.4c	118
AFC Imp. mix	10.4c	116
Weyerhaeuser 2^{d} gen.	10.0c	116
AFC Nat. mix	9.3d	67
Weyerhaeuser 1st gen.	8.9d	127

Table 4--Mean survival rate for the nine loblolly pine families. Means followed by the same letter are not significantly different ($p < 0.05$)

Family	Survival	Sample size
	percent	
Weyerhaeuser 1st gen.	89.6a	4
ArborGen 34	88.9a	4
ArborGen MCP-22M	86.8a	4
ArborGen 520	82.6a	4
AFC Imp. mix	82.6a	4
ArborGen 579	82.6a	4
Weyerhaeuser 2^{d} gen.	81.3a	4
ArborGen 522	79.9a	4
AFC Nat. mix.	57.4b	3

planting crew inexperience and did not accurately reflect the capability of the family. As a result of dropping this plot, the AFC Natural Mix loblolly pine family was represented by only three plots in the survival calculations. Note that the planting crew was halted and retrained after Block 1 Plot 4 was completed.

The statistical analysis did not reveal many significant differences in survival among the nine families. Survival rate for the Arkansas Forestry Commission Natural Mix family was 57.4 percent. This was significantly lower than the survival rate for all of the other families which ranged from 79.9 percent to 89.6 percent.

Stem Defects

Stem defects were recorded only for those trees still alive at the time of sampling. Since all plots experienced some mortality, an analysis of simply the number of stems displaying defect in each plot would have been meaningless. We divided the number of stems with defects by the number of surviving stems in the plot to yield a rate which was analyzed. Mean overall stem defects rate for the study was 12.2 percent. Mean defect rate for the different families ranged from 6.9 percent to 19.6 percent (table 5).

Table 5--Mean defect rate for the nine loblolly pine families. Means followed by the same letter are not significantly different at $p < 0.05$

Family	Defect	Sample size
	percent	
ArborGen 579	19.6a	4
ArborGen 520	15.0ab	4
ArborGen 522	13.8ab	4
Weyerhaeuser 2^{d} gen.	13.0ab	4
Weyerhaeuser 1st gen.	12.6ab	4
AFC Imp. mix	12.5ab	4
AFC Nat. mix.	9.2b	4
ArborGen 34	7.3b	4
ArborGen MCP-22M	6.2b	4

DISCUSSION

Height Growth

When the tested loblolly pine families are listed from tallest to shortest (table 3), the top three families are two of ArborGen's Atlantic Coast families and one Arkansas sourced family. Two of those are elite families and one is a second generation family. The fourth family is an ArborGen elite family sourced from Arkansas. The ArborGen elite families produced greater height growth than the most of the second generation families. One ArborGen second generation family, AG 34, produced height growth equal to the elite families.

The Arkansas Forestry Commission's Natural Mix family and Weyerhaeuser's first generation family produced the least height growth of the nine families. Families derived from Atlantic Coast sources tended to produce superior height growth compared to those from western Gulf Coastal Plain sources. The Weyerhaeuser first generation family and the AFC Natural Mix family have been subjected to little or no selection for height growth; thus one would not expect superior height growth from these families.

Survival

The lack of significant differences among the top eight families in our study may be explained by either of two ideas. First, our sample size may have been too small to clearly define survival rates within the narrow range of survival rates of the top eight families. Unfortunately, the size of this study was limited by number of seedlings available to us and space available to install the variety trial. Second, the lack of significance may also be attributed to very good and consistent seedling quality. From an operational viewpoint, the difference between 79.9 percent survival and 89.6 percent survival will rarely matter. When considering the implications of these survival rates, one must keep in mind that these seedlings were planted during a year when annual rainfall was significantly above normal, even though subsequent years were abnormally dry. These survival rates may not accurately reflect the rates that would have been produced had the seedlings been planted during a year of normal or subnormal rainfall.

Stem Defects

Just as with survival, analysis of the stem defect rates was plagued by small sample sizes. The ANOVA divided the nine families into two broadly overlapping groups. Three families (ArborGen MCP-22M, ArborGen 34, and Forestry Commission Natural) had significantly lower defect rates than ArborGen 579 which had the highest defect rate. The remaining five families could not be statistically distinguished from each other or from either end of the defect rate spectrum. There was no clear differentiation between defect rates of Atlantic Coastal families and western Gulf Coastal Plain families. Some Atlantic Coast-derived families were found at each end of the defect rate spectrum.

CONCLUSION

Two loblolly pine families produced by ArborGen (34 and M-22M) ranked among the best families for height growth, survival rate, and defect rate in this variety trial. The remaining seven families tended to rank well in one or two of the tested parameters but poorly in the remaining one or two.

It has been postulated by some that Atlantic Coast families were not well suited to western Arkansas' hot and dry summers. The idea anecdotally presented by some is that height growth might be better but that poor survival would render the stand less productive than loblolly pine derived from western Gulf Coastal Plain sources. In spite of the dryer, or much dryer, than normal summers southwest Arkansas has experienced during the term of this variety trial, loblolly pine families derived from Atlantic Coast sources did not suffer reduced height growth or poorer survival compared to families derived from western Gulf Coastal Plain sources. This is consistent with the findings of Will and others (2010) in southeastern Oklahoma. These preliminary results present no evidence to suggest that one should select loblolly pine families derived from western Gulf Coastal Plain sources over those derived from Atlantic Coast sources based solely on height growth and survival.

Be aware that these are preliminary results after only four growing seasons. Much can change through the next 10 to 15 years. Also bear in mind that this study examines seedlings grown on one site. Different soils or moisture regimes may yield different results.

ACKNOWLEDGEMENTS

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LITERATURE CITED

iAIMS Climatic Data. 2013. iAIMS Climatic Data web site. https://beaumont.tamu.edu/climaticdata/ViewWeatherDataPage.aspx?Type=Daily&index=US_US3&state=AR&county=HEMPSTEAD&station=HOPE%203%20NE. [Date accessed: May 30, 2013].

SAS Institute Inc. 2008. SAS software for Windows. Release 9.2. Cary, NC: SAS Institute, Inc.

Will, R.; Hennessey, T.; Lynch, T. [and others]. 2010. Effects of planting density and seed source on loblolly pine stands in southeastern Oklahoma. Forest Science. 56(5): 437-443.

COMPUTER PROGRAMS FOR OPTICAL DENDROMETER MEASUREMENTS OF STANDING TREE PROFILES

Jacob R. Beard, Thomas G. Matney, and Emily B. Schultz[1]

Abstract--Tree profile equations are effective volume predictors. Diameter data for building these equations are collected from felled trees using diameter tapes and calipers or from standing trees using optical dendrometers. Developing and implementing a profile function from the collected data is a tedious and error prone task. This study created a computer program, Profile Data Calculator (www.timbercruise.com/Downloads/TProfile/TProfileSetup.exe) that calculates diameters and heights directly from dendrometer outputs, negating the necessity for intermediate calculations and allowing for improved functionality and immediate utility of the devices. Six different dendrometer models (calipers, calipers with Haglof Gator-Eyes® attachment, Wheeler Pentaprism, Spiegel Relaskop, Tele Relaskop, and Barr & Stroud FP15) were evaluated for incorporation into the computer program. The data processing program was written in Microsoft Visual Basic® Editor within Microsoft Excel®. TProfile© software reads the Microsoft Visual Basic® output and produces tree profile equations which are subsequently imported into TVolume© software to predict volume. Six program modules were created that allow for rapid deployment of profile equations for tree volume estimation.

INTRODUCTION

Tree profile equations are tools for quickly calculating tree volumes and heights to any top diameter limit using minimal input data, such as diameter at breast height (d.b.h.) and total or merchantable height. There are a variety of these equations that involve various methods of assessing stem shape. Max and Burkhart (1976) developed a profile function that is, perhaps, the best known. The utility of profile equations is their flexibility in determining tree taper, volume, and value for multiproduct inventories where merchantability specifications frequently change. However, the application of these functions is only valid for the specific tree grouping sampled during data collection (Grosenbaugh 1966). Adequate data must be available for each specific grouping (for example: species, species group, region, geographic area, physiographic provenance, stand density, or age) from which taper may be generalized to form an equation (Grosenbaugh 1966). The data collection process is tedious and requires height and diameter measurement at many points along the entire length of a standing or felled tree stem. Felled and standing tree data collection methods differ in regard to ease of measurement, accuracy, required time, and cost.

Felled Tree Data Collection

Felled tree data collection is destructive but allows for direct measurement of the stem. Measurements are generally acquired through the use of calipers, a bark gauge, and a logger's tape. Matérn (1990) discovered that when directly measuring diameter, tapes and calipers provided about the same bias, but use of diameter forks, such as the sector fork, resulted in a higher positive bias. Diameter tapes and calipers have been the preferred diameter measurement devices for the felled tree method, which has long been considered a standard and accurate procedure for developing profile equations. Once data are collected, additional calculations are not required other than organization into the proper format for analysis. The use of this method is limited to situations where a significant number of trees are available for destructive sampling, which may not be feasible if there are restrictions such as accessibility or cost.

Standing Tree Data Collection

Standing tree data collection may be achieved in several ways. If possible, the tree can be directly measured using climbing spikes and a harness. Climbing individual trees, however, is costly, time consuming, and requires additional safety procedures. Optical dendrometers are an alternative to tree climbing that allows visual measurement of diameter at any point along the stem (Avery and Burkhart 2002). Existing optical dendrometer models can be delineated into one of three classes (optical forks, optical calipers, and short base rangefinder dendrometer) according to the trigonometric functions from which their measurements are derived. Grosenbaugh (1963) details the history and theory of optical dendrometers and provides information on the geometry involved with each

[1]Graduate Research Assistant and Professors, respectively, Mississippi State University, Forest and Wildlife Research Center, Mississippi State, MS 39762.

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class and its associated benefits and disadvantages. The most notable disadvantage is that collected data are typically intermediate values for which post-processing requires a thorough understanding of the underlying principles of the instrument. Of the three classes of optical dendrometers, optical calipers are the simplest group to use because they do not require knowledge of height or vertical angle in order to calculate diameter (Grosenbaugh 1963). Optical forks and short base rangefinder dendrometers relate their respective diameter measurements back to recorded height to the point of diameter measurement.

OBJECTIVES

The focus of this study was to improve the usability of optical dendrometer data in developing standing tree profile functions by automating intermediate calculations and profile function construction within an easy to use computer interface. Because there will always be a need for accurate volume prediction from reliable profile functions, improving the usability of standing tree diameter measurements for this purpose would make them more widely applicable. Previous optical dendrometer programs were developed for single devices in non-Microsoft Windows® based applications (Arney and Paine 1972, Grosenbaugh 1967, Jager 1976, Parker 1997). The application resulting from this study included instruments from all dendrometer types and was delivered in a Web accessible Microsoft Excel® Visual Basic® Editor program.

METHODS

Dendrometers

Optical dendrometer devices from each of three categories (Grosenbaugh 1963) were evaluated for incorporation into a Microsoft Excel® Visual Basic® Editor program. The Wheeler Pentaprism® and calipers with a Haglof Gator-Eyes® attachment represented the optical calipers class while the Spiegel Relaskop® and Tele Relaskop® represented the optical fork category. The Barr & Stroud FP15® filled the short base rangefinder dendrometer category. In addition to these devices, calculations and input routines were constructed for felled tree caliper measurements.

Microsoft Excel® Visual Basic® Editor was used to code program modules to calculate tree profile diameters from each instrument's field observations according to their underlying trigonometric functions. Respective manuals and research articles for the devices provided the appropriate mathematical equations to derive results from intermediate data. With the exception of the Tele Relaskop®, where Parker's (1997) improved equations (through personal correspondence with Bitterlich) were used, each instruments manufacturer's manual was referenced for program development. Development of the Spiegel and Tele Relaskop® programs were based on the manufacturer's manuals, publication by Parker (1997), and Bitterlich's (1984) conspectus on a suite of devices that he engineered. The Wheeler Pentaprism® and calipers with Haglof Gator-Eyes® attachment read diameter in inches directly from the devices and do not require calculations, but do require separate height measuring instruments. The Barr & Stroud FP15® and Spiegel Relaskop® require both diameter and height calculations and presented the most complicated programming routines of all the dendrometers. For this reason, close attention and verification were exercised during program development to ensure that appropriate functions were written that produced good variable estimates. A routine for constructing profile equations from felled tree data was added to those for the five optical dendrometers, and the application was named Profile Data Calculator.

Profile Data Calculator produces diameter and height data in a format acceptable for input to TProfile© (Matney 1996a). TProfile© software fits 16 commonly used tree profile functions and outputs parameter estimates, fit statistics, and residual analysis files. TProfile© reads the Microsoft Visual Basic® output and produces tree profile equations that can be imported by TVolume© (Matney 1996b) or TCruise© (Matney 1996c) software to predict volume.

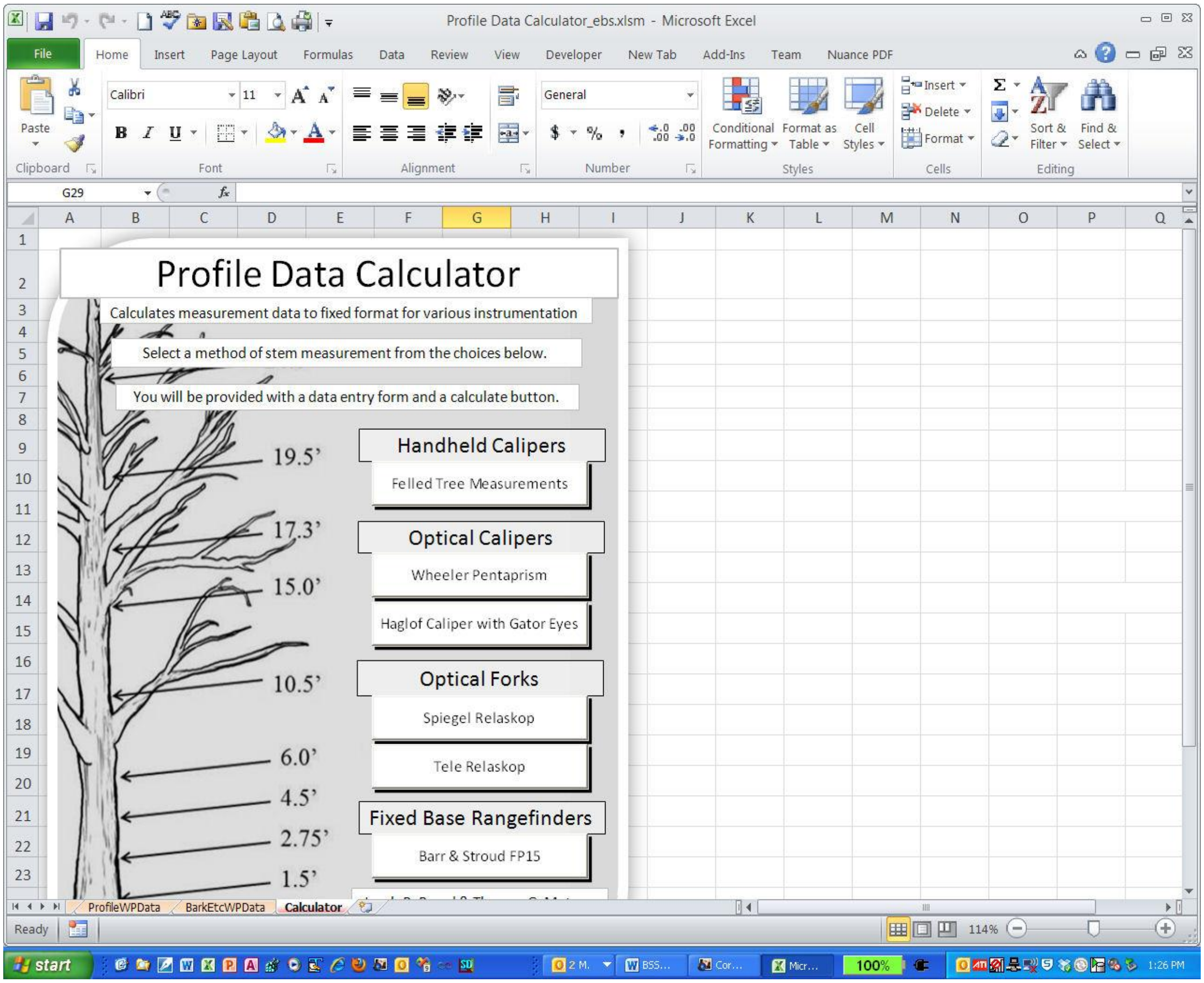

Figure 1--Welcome Microsoft Excel® worksheet for Profile Data Calculator software that calculates diameters and heights directly from outputs of five optical dendrometer instruments plus felled tree data.

Profile Data Calculator User Interface Construction

A standard user entry form was created for each optical dendrometer and for felled tree data collection. In the case of felled tree data collection, input fields were created for measurements along the felled bole at 0.5, 2.0, 3.5, 4.5, 6.0, and 8.0 feet and every 4.0 feet thereafter, including an additional measurement at form class height, 17.3 feet. For standing tree optical dendrometer data collection, input fields were created for measurements along the standing bole at groundline, 0.5, 1.5, 2.75, 4.5, and 6.0 feet, and every 4.5 feet thereafter, plus 17.3 feet for form class height. In both data collection methods, measurements were terminated at 3.0 inches diameter outside bark (DOB) top height. Total height was also an input field for both methods. Other data entry form elements pertain to specific units and data requirements for measuring diameters and heights that are associated with the underlying technology of the instrument.

A Microsoft Excel® application workbook was designed to open in a welcome worksheet (fig. 1), from which the user navigates to the appropriate data input form through the use of command buttons linked to separate instrument/data collection interface worksheets.

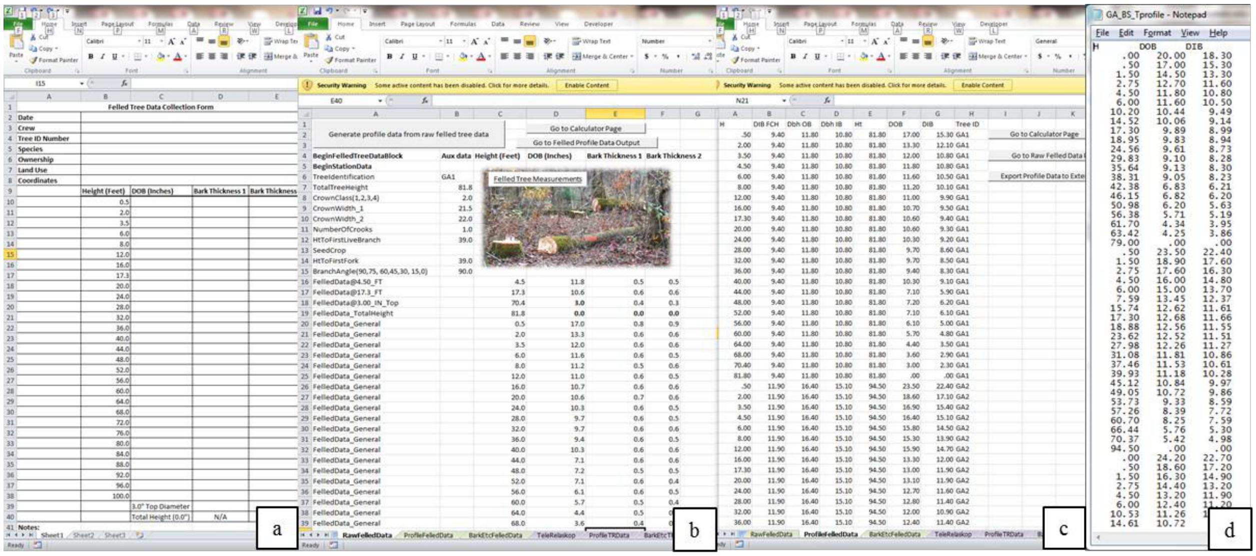

Figure 2--Profile Data Calculator screen shoots of the (a) field data collection form appropriate to the device, (b) spreadsheet data entry form appropriate to the device, (c) output form containing calculated tree profile data from entered field data, and (d) electronic output file in a format acceptable as input to TProfile©.

An output data worksheet is associated with each data collection form, and a simple return command button was implemented to link the user back to the main page. Error routines and messages were implemented to prevent calculations from easily discerned data entry (typographical) errors such as the misplacement of a decimal point or a missing digit. Detection of other errors requires visual inspection of graphical displays of individual tree profile data.

RESULTS

The Web accessible Microsoft Excel® Visual Basic® program (Profile Data Calculator) created in this study provided an automated process for producing formatted tree profile data from the field outputs of five optical dendrometer instruments plus felled tree data. Each instrument module within the program provided (1) a field data collection form appropriate to the device (fig. 2a); (2) a spreadsheet data entry form appropriate to the device (fig. 2b); (3) an output form containing calculated tree profile data from entered field data (fig. 2c); and (4) an electronic output file in a format acceptable as input to TProfile© (fig. 2d). Each program module incorporates the full functionality of its respective device to accommodate a range of user preferences. Instruments that record field data that are intermediate in form and require post-field calculations may now be considered as functional as instruments that provide direct measurements.

CONCLUSION

The Web assessable Profile Data Calculator, www.timbercruise.com/Downloads/TProfile/TProfileSetup.exe (case sensitive), software created in this study provides profile data outputs for five optical dendrometer instruments (calipers with Haglof Gator-Eyes® attachment, Wheeler Pentaprism®, Spiegel Relaskop®, Tele Relaskop®, and Barr & Stroud FP15®) representing three classes of dendrometers (optical forks, optical calipers, and short base rangefinder dendrometer) plus felled tree data. Many optical dendrometer instruments record intermediate data in the field that must undergo trigonometric computations to produce diameter and height measurements. This process is tedious, time consuming, and can hamper the instrument's use and functionality. In situations where tree species are too valuable to collect felled data or there are logistical obstacles to felling trees, optical dendrometers provide a means of collecting data for constructing tree profile equations. The application is executed from within an easy-to-use Microsoft Excel® Visual Basic® application and produces files formatted for subsequent input into TProfile©, TVolume©, or TCruise© software for profile equation construction and volume estimation.

LITERATURE CITED

Arney, J.D.; Paine, D.P. 1972. Tree and stand volumes using height-accumulation and the telescopic Spiegel-Relaskop. Forest Science. 2: 159-163.

Avery, T.; Burkhart, H. 2002. Forest measurements. Boston: McGraw-Hill. 456 p.

Bitterlich, W. 1984. The relascope idea: relative measurements in forestry. Slough (Buckinghamshire): Commonwealth Agricultureal Bureaux. 242 p.

Grosenbaugh, L.R. 1963. Optical dendrometers for out of reach diameters: a conspectus and some new theory. Washington, DC: Society of American Foresters. 47 p.

Grosenbaugh, L.R. 1966. Tree form: definition, interpolation, extrapolation. The Forestry Chronicle. 42(4): 444-457.

Grosenbaugh, L.R. 1967. STX-Fortran-4 program for estimates of tree populations from 3P sample-tree-measurements. Berkeley: U.S. Department of Agriculture Forest Service, Pacific Southwest Forest and Range Experiment Station. 47 p.

Jager, D.J. 1976. Pocket electronic calculator for Barr and Stroud dendrometer. Journal of Forestry. 7: 438.

Matérn, B. 1990. On the shape of the cross-section of a tree stem. An empirical study of the geometry of mensurational methods. Report 28. Umeá, Sweden: Swedish University of Agricultural Sciences, Section of Forest Biometry. S-901 83. 47 p.

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. 1996c. TCruise©. Starkville, MS: World Wide Heuristic Solutions.

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

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

PRO-B SELECTION METHOD FOR UNEVEN-AGED MANAGEMENT OF LONGLEAF PINE FORESTS

Dale G. Brockway, Edward F. Loewenstein, and Kenneth W. Outcalt[1]

Abstract-- Interest in uneven-aged silviculture has increased since advent of ecosystem management programs, which place greater emphasis on ecological values and ecosystem services while also harvesting timber from the forest. However, traditional uneven-aged approaches (e.g., BDq) are often criticized as too complex, costly, and requiring highly-trained staff. The Proportional-B method (Pro-B) addresses these concerns, making uneven-aged silviculture a practical management option. In an operational-scale study, Pro-B was successfully used, by forest staff from a range of professional backgrounds, following less than 3 hours of training, to apply single-tree selection and group selection in longleaf pine (*Pinus palustris* Mill.) forests. Field crews achieved precision levels within 3 to 5 percent of the target residual basal area. By aggregating many diameter size-classes into only three size-class groups, Pro-B improves efficiency by requiring tree markers to remember only three fractions while making a single pass through the stand. Not being restricted by maximum-diameter rules also allows flexibility to retain larger trees for enhancing structural diversity. Trees of specific species and with good form, broad crowns and cavities can be retained, while adjusting spacing to release residuals. Systematic quantification of tree removal enables different individuals to obtain consistent results. A stable structure is maintained with characteristics of a mature forest, while regeneration is initiated and timber is removed through a periodic cutting cycle. With a focus on forest sustainability and flexibility to retain large trees and biological legacies by mimicking small-scale natural disturbances, Pro-B might be implemented to achieve the production objectives and stewardship goals of retention forestry.

INTRODUCTION

Ecosystem management policies emphasizing biodiversity and long-term sustainability have, in recent years, increased the interest in and practice of uneven-aged silviculture (O'Hara 1998). This is true nowhere more than in southern pine forests, where even-aged methods for timber production lead to adverse consequences for other ecosystem values (Guldin 2006). Protecting native plant communities, maintaining continuous forest canopy, and facilitating development of large old trees are among the desirable habitat features resulting when uneven-aged silviculture is applied in an adaptive management framework (Brockway and others 2006). Uneven-aged silviculture also affords a major advantage, in that natural regeneration is more or less continuous through time, as late-successional stand dynamics are emulated (Guldin 1996).

Despite an historical decline, longleaf pine (*Pinus palustris* Mill.) forests are highly valued for a variety of resources having ecological, economic and cultural importance and substantial interest has recently emerged in best management approaches for sustaining and restoring them (Brockway and others 2005a, Van Lear and others 2005). Although longleaf pine has been mostly managed with even-aged methods and was formerly thought to be too intolerant for uneven-aged silviculture (Croker and Boyer 1975), recent evidence suggests this to be a viable management alternative (Brockway and Outcalt 1998, Farrar 1996, Gagnon and others 2003, McGuire and others 2001, Palik and others 1997). Longleaf pine can grow in pure stands and also in association with numerous trees species across a wide range of ecosystem types, including slash pine (*P. elliottii* Engelm.) on flatwoods, loblolly pine (*P. taeda* L.) and shortleaf pine (*P. echinata* Mill.) on mesic uplands, and various hardwoods on xeric sandhills, mountains and other site types (Boyer 1990). This natural variety indicates that no single prescription is appropriate for sustaining longleaf pine everywhere. Prudent managers will select approaches suited for their specific environment, which lead to: (1) an overstory dominated by mature longleaf pine occurring as uneven-aged stands or even-aged patches across an uneven-aged landscape, with a lesser component of other tree species; (2) a midstory that is generally absent or mostly composed of ascending longleaf pine in scattered, modestly-sized canopy gaps; and (3) an understory with abundant longleaf pine seedlings and groundcover dominated by native grasses and

[1]Research Ecologist, USDA Forest Service, Southern Research Station, Auburn, AL 36849; Associate Professor, School of Forestry and Wildlife Sciences, Auburn University, AL 36849; and Emeritus Research Ecologist, USDA Forest Service, Southern Research Station, Athens, GA 30602.

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forbs with lesser cover of shrubs and vines (Brockway and others 2005a). Research has fostered improved technology for the establishment, recovery, and maintenance of longleaf pine ecosystems (Jose and others 2006). Private sector interest and public sector direction now emphasize improved management of existing longleaf pine and, on suitable sites, eventual expansion of the area occupied by longleaf pine. To these ends, the principal goal of all sustainable forest management should be application of silvicultural methods that ensure maintenance of longleaf pine ecosystems in perpetuity. Such methods will incorporate natural regeneration and, to the degree possible, simulate disturbance events and other ecological processes (e.g., lightning strikes, periodic surface fire, windstorms) that contributed to maintaining longleaf pine ecosystems prior to settlement. However rather than relying upon random chance, management will deliberately manipulate the ecosystem to achieve specific stewardship objectives (Brockway and others 2006).

Uneven-aged silviculture is not new in the South and has been practiced in loblolly pine and shortleaf pine using the Volume Guiding Diameter Limit (VGDL) (Reynolds 1969, Reynolds and others 1984) and Basal Area-Maximum Diameter-q (BDq) procedures (Guldin 2006, Shelton and Cain 2000). Following field tests, guidance for uneven-aged silviculture in longleaf pine recommended the use of BDq, with B = 60 square feet per acre, D_{max} = 20 inches, and q = 1.44 for 2-inch size-classes (Farrar 1996). These parameters define a target structure with a “reverse-J” diameter distribution, against which stands are evaluated for cutting. In expert hands, BDq can be adeptly applied. However, the requirement to mark and tally trees by 2-inch size-classes may require multiple passes through a stand before the tally by size-class satisfies the cutting prescription. Such a challenge is often prohibitive in the field, thus limiting the applicability of BDq.

Frustration with the complexity of BDq and suspicion that such a procedure might constitute an unnecessary over-control of stand structure, led to efforts to expedite field application of selection silviculture. Recognition that basal area (B) is the most important of these variables, resulted in deletion of the requirements to identify a D_{max} and adhere to a specific q-value. Basal area is biologically linked to the cross-sectional area of sapwood, hence, leaf area index and total foliar display of the stand (O'Hara and Gersonde 2004, O'Hara and others 2001, O'Hara and Valappil 1999). Basal area may also be used for expressing stand condition relative to stocking, competition and prospects for regeneration success. Unless maximum timber production is desired, there is no need to set a tree-diameter size limit, since basal area occurring in a small number of trees in the larger size-classes can be compensated for through adjustments in the smaller size-classes. The q-value is informative concerning the relative balance among tree size-classes, but those near 1.4 or 1.5 may not be needed to support stand development, and values as low as 1.2 or 1.3 may be adequate. Thus, B was retained as the principal index for this method. Rather than tallying individual trees by numerous 2-inch size-classes, the many size-classes were combined into three size-class groups, each representing a stage in stand development and potential product [< 6, 6 to12, and > 12 inches at diameter breast height (d.b.h.)]. Tree marking was also changed to simply mark the fraction of trees that should be removed in each size-class-group. To attain the target stand structure, tree markers need to remember at most only three fractions that represent the rate of tree removal in each of the three size-class groups. This process systematically apportions residual basal area among size-class groups, thus the name arose as Proportional-B (for “proportional basal area”), or more simply Pro-B.

Extensive research performed with BDq has resulted in adaptations that improve its field application in a variety of forest types (Farrar 1996; Guldin 2011; Guldin and Baker 1998; Leak 1964; Leak and Filip 1975; Leak and others 1987; Nyland 1987, 1998, 2007). Pro-B is somewhat different, in that it was more recently developed to serve both timber production and biodiversity conservation purposes, while specifically retaining larger trees in the forest. The Pro-B stand structure is not rigidly defined by a q-value, recognizing that a diameter distribution may correspond to multiple q-values which can vary through time (Leak and Filip 1975). Tending an uneven-aged stand, so that its diameter distribution exactly matches a q-defined curve is an exercise of imposing arbitrary human values on the ecosystem (O’Hara 1998). Doing so ignores the wider range of structures that can be sustainable (O’Hara 1996, Seymour and Kenefic 1998). A substantial problem with q-based approaches is that reliance on them can create a false sense of stability, where imbalances in age structure may not become evident until well after they can be easily addressed (Seymour and Kenefic 1998). Alternatively, Pro-B stand structure is defined by the proportion of basal area distributed among size-class groups (typically 1:2:3, although other ratios may be valid). Also, the Excel-based Pro-B Calculator makes the complicated computation of marking guides much easier and provides an opportunity for the projected results to be assessed before field application. This study was undertaken to assess suitability of Pro-B for applying uneven-aged silviculture by: (1) ascertaining the consequences of applying single-tree selection and group selection in longleaf pine forests, including effects on stand structure, growth, and regeneration; (2) evaluating its effectiveness as a single-pass method; and (3) discerning the efficiency with which it can be learned and applied in the field by managers from a wide range of backgrounds.

MATERIALS AND METHODS

Study Sites and Management History

Goethe State Forest flatwoods are located 15 miles east of the Gulf of Mexico (29° 13' N, 82° 33' W), on the Lower Coastal Plain of the Florida peninsula. Temperatures in the humid subtropical climate range from a maximum of 91 °F in summer to a minimum of 41 °F in winter. Annual precipitation averages 57 inches, arriving mostly from April to September. At 50 feet above sea level, surface topography is nearly level and dominated by Smyrna fine sand, which is very deep, poorly-drained, low in organic matter and nutrients, and low in water holding capacity (Slabaugh and others 1996). The overstory was dominated by longleaf pine, with lesser amounts of slash pine and very few hardwoods. Tree seedlings were few and mostly comprised of slash pine and longleaf pine. Understory plants were dominated by shrubs, primarily saw-palmetto [*Serenoa repens* (W. Bartram) Small] and gallberry [*Ilex glabra* (L.) A. Gray]. Because of shrub dominance, the herbaceous layer was poorly developed, with wiregrass (*Aristida beyrichiana* Trin. & Rupr.) and broomsedge bluestem (*Andropogon virginicus* L.) the most prominent grasses and few herbs. The area was cutover about 100 years ago and then subjected to a 50-year period of fire exclusion, during which trees recovered and saw-palmetto expanded to now dominate the understory. Since 1992, active programs of prescribed burning and timber harvest have been implemented to foster multiple-use management and restore the ecosystem. Stands received improvement cuts between 1997 and 2004, and winter-season prescribed fire has been applied on a 3-year cycle. Mature pines were 48 through 74 years in age. Site index ranges from 70 through 80 feet at 50 years.

Blackwater River State Forest uplands are located 30 miles north of the Gulf of Mexico (30° 47' N, 86° 44' W), on the Middle Coastal Plain of the Florida panhandle. Average temperatures range from 80 °F in summer to 54 °F in winter. Annual precipitation averages 65 inches, with about half arriving from June to September. At 200 feet above sea level, topography is nearly level to gently inclined and occupied by Troup, Orangeburg, Lucy and Dothan sands, which are well-drained, low in organic matter and nutrients, and low to moderate in water holding capacity (Weeks and others 1980). The overstory was dominated by longleaf pine, with a smaller component of hardwoods and slash pine. Tree seedlings were abundant, with southern red oak (*Quercus falcata* Michx.), bluejack oak (*Q. incana* W. Bartram), post oak (*Q. stellata* Wangenh.), persimmon (*Diospyros virginiana* L.), and longleaf pine most common. Dangleberry [*Gaylussacia frondosa* (L.) Torr. & A. Gray], blueberries (*Vaccinium* spp.), and blackberries (*Rubus* spp.) were the most prominent shrubs. The herbaceous layer was well developed and species-rich, with wiregrass and broomsedge bluestem dominating the grasses and silverthread goldaster [*Pityopsis graminifolia* (Michx.) Nutt.] the most abundant herb. The area was occupied by second-growth longleaf pine that naturally regenerated after cutover of the original forest in the 1920s. This site has received numerous prescribed fires since 1970, on a 3-year cycle, initially during the dormant season and then changing to growing-season fire since 1995. Improvement cutting in 1981 and 1991 and hurricane-salvage in 2004 were followed by multiple waves of natural regeneration. Most of the mature pines were about 66 years, with the oldest being age 80. Site index is 80 feet at 50 years.

Experimental Design and Treatments

In summer 2004, a randomized complete block design was installed as three replications of the two silvicultural treatments, plus control, at each site. In spring 2005, treatments were assigned within the three replications that were grouped as blocks to topographically account for moisture gradient or spatial differences. The nine plots (stands) are each 22.2 acres (984- by 984-feet) and totaled 200 acres at each forest. Within each treatment plot, five 0.25-acre subplots were randomly located, each 66- by 164-feet with the long axis oriented in a north-south direction. The stand reproduction alternatives examined were the uneven-aged techniques of: (1) single-tree selection and (2) group selection, plus (3) no harvest to serve as the experimental control.

Single-tree selection has the advantage of maintaining a high level of canopy cover while periodically allowing removal of some trees from the forest. However, since longleaf pine is known to be intolerant of competition for light and soil resources, it is unclear whether it can regenerate and fully develop in the small space resulting from the death of a single overstory tree (Brockway and others 2005b). Most evidence indicates that several longleaf pine trees must fall from the canopy before sufficient space is available to allow longleaf pine juveniles to begin recruiting into the canopy (Brockway and others 2006). Hence, the importance of this comparative analysis experiment. Group selection simulates the natural gap-phase regeneration process of longleaf pine, by simultaneously tending the forest matrix and creating small canopy gaps (Brockway and Outcalt 1998). Although natural regeneration often occurs widely in the forest, young longleaf pine are usually more concentrated and better developed in canopy gaps ranging from 0.25- to 2-acres in size. Gaps may resemble very small clearings or contain scattered mature trees and typically regenerate as even-aged cohorts, in an uneven-aged matrix. Thus, the resulting forest eventually becomes an uneven-aged mosaic of even-aged patches. Patches with similar age cohorts need to be sufficiently dispersed to achieve the desired result (Brockway and others 2006). As long as herbaceous plants dominate the periodically-burned gaps, longleaf pine seed should germinate and seedlings will become established when good seed years are followed by favorable weather. Creating group openings at locations where regenerating seedlings already exist is an effective way to promote their release and eventual recruitment into the canopy. During treatment, the forest matrix was tended using Pro-B, and three 0.25- to 0.5-ha gaps were created in each 22-acre stand. Gap width ranged from 1.4 to 2 times the height of adjacent dominant trees.

Pro-B apportions the target residual basal area into a structure consistent with a ratio of 1:2:3 for small (< 6 inches), medium (6 to 12 inches), and large (> 12 inches) size-classes, respectively. This ratio was developed for forests having trees no > 24 inches, with most < 20 inches, at d.b.h. In BDq terms, this ratio would approximate a q-value of about 1.3. When Pro-B is applied in forests containing trees of larger diameter, a different ratio among and different boundaries for size-class groups may be more appropriate. These can be established only after a stable stand structure is defined. Alternative ratios and boundaries for size-class groups may also apply under circumstances where forest management seeks to maintain habitat conditions for at-risk species. For example, a residual basal area ratio of 1:3:6 for small (< 10 inches), medium (10 to 14 inches), and large (> 14 inches) size-classes is implied in the recovery plan for red-cockaded woodpeckers [*Picoides borealis* (Vieillot)] (U.S. Fish and Wildlife Service 2003). Developing new ratios and alternative size-class groupings suitable for differing forest conditions and management concerns will be areas of future study.

Measurements and Analysis

In winter 2005, tree data were collected on all subplots to establish pretreatment stand conditions. Species was recorded, and diameter of all trees >1 inch d.b.h. was measured to the nearest 0.05 inch. The total height of trees in a subsample representing the full range of size-classes was also measured to the nearest 4 inches to establish the height-diameter relationships for longleaf pine and slash pine. Repeated post-treatment measurements of trees were then completed following the 2006, 2007, and 2008 growing seasons. Following the 2005 and 2008 growing seasons, the number of slash pine seedlings and grass-stage (height < 6 inches) and bolt-stage (height 6 inches to 6 feet) longleaf pine seedlings were recorded on all subplots. Tree and seedling data were summarized as the mean for each plot and analyzed by treatment and change through time. Stand density and basal area were calculated from tree diameter data. Height-diameter relationships for pine were computed with regression analysis, using height and diameter data (Hintze 2007). Stand volumes (cubic feet per acre) were calculated for each species, by summing individual tree volumes to a 4-inch top outside bark on an area basis and using height and diameter data in regional equations (Saucier and others 1981). Means of dependent variables for each plot were used to estimate the means and variances for the treatment units. A repeated measures analysis of covariance (ANCOVA), using initial conditions as covariates, was used to evaluate time and treatment effects and interactions (Hintze 2007). Responses of treatments were compared using pairwise contrasts. The trend through time after treatment was analyzed using orthogonal polynomials. Significant differences were discerned at the 0.05 level.

RESULTS

Pro-B Application

Tree-marking guides were computed using the Pro-B Calculator, a Microsoft Excel-based tool with tabular fields for input of stand data and output of marking guides and graphic displays of pre-cutting and post-cutting structures. Pro-B apportioned the target of 50 square feet per acre into 8.3 square feet per acre for small (< 6 inches), 16.7 square feet per acre for medium (6 to 12 inches), and 25 square feet per acre for large (> 12 inches) size-class groups. During winter 2006, training workshops were held at the Goethe State Forest and Blackwater River State Forest, where less than 3 hours of instruction were presented about selection silviculture, the Pro-B method, and field considerations when applying the computed marking guides. In the field, newly-trained practitioners arrayed themselves 66 feet apart in a line along the edge of each stand and then made a single pass through, covering the width of their assigned lane, marking trees they identified for removal in accordance with the marking guides. These practitioners, ranging from administrators to field foresters, wildlife biologists, recreation specialists, and GIS specialists demonstrated remarkable skill in easily marking stands to a high level of precision (to within 3 to 5 percent error of the target residual basal area). Within the Pro-B numeric marking guides, they easily incorporated marking rules, such as: (1) take the worst and leave the best trees, (2) remove less desirable species, (3) adjust spacing to release residual trees, and (4) retain snags, live cavity trees, and large trees with broad crowns that can benefit wildlife. During November and December 2006, marked trees were removed by private logging contractors, and the resulting residual stands met prescription specifications.

Stand Structure

On the Goethe State Forest flatwoods, harvest reduced density from 126 to 74 trees per acre with single-tree selection and from 130 to 67 trees per acre with group selection. Declines in tree density were 41 percent following single-tree selection and 49 percent after group selection, with harvested stands significantly less dense than uncut controls. Harvest also reduced stand basal area from 70.9 to 50.9 square feet per acre with single-tree selection and from 72.6 to 44.8 square feet per acre with group selection. Declines in basal area were 28 percent following single-tree selection and 38 percent after group selection, again with lower residual basal areas than control stands. On the Blackwater River State Forest uplands, harvest reduced tree density from 137 to 107 trees per acre with single-tree selection and from 200 to 155 trees per acre with group selection. Declines in tree density were 22 percent following both single-tree and group selection, with harvested stands significantly less dense than controls. Harvest also reduced stand basal area from 60.9 to 47.8 square feet per acre with single-tree selection and from 73.5 to 48.7 square feet per acre with group selection. Declines in basal area were 22 percent after single-tree selection and 34 percent after group selection. An example of stand structural dynamics for group selection can be seen in figure 1.

Tree Volume and Growth

On Goethe State Forest flatwoods, total pine volume in treated stands prior to harvest averaged 1,959 cubic feet per acre, which was similar to the 1,933 cubic feet per acre in controls. Volumes were divided between longleaf pine and slash pine on an 80 to 20 percent basis in treated stands and a 59 to 41 percent basis in controls. Both selection methods reduced volume to levels significantly less than in controls (2,036 cubic feet per acre). Single-tree selection reduced volume by 26 percent to 1,438 cubic feet per acre, while group selection reduced volume by 36 percent to 1,257 cubic feet per acre. By the second post-treatment growing season, the annual volume growth rate was 2 to 4 percent. On Blackwater River State Forest uplands, total pine volume in treated stands prior to harvest averaged 1,409 cubic feet per acre, which was not significantly different from the 1,276 cubic feet per acre in controls. These volumes were mostly longleaf pine, with slash pine comprising only 4 percent

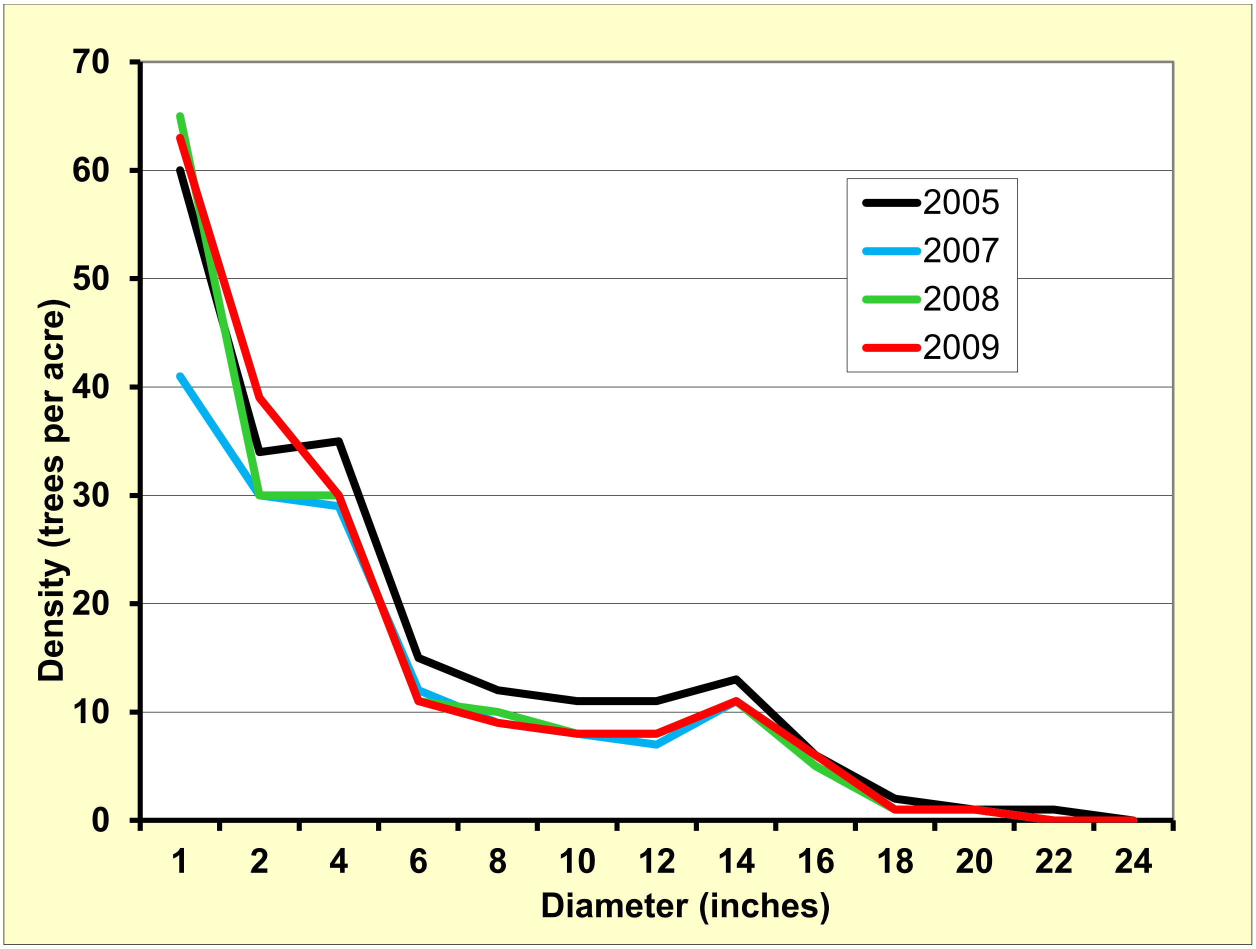

Figure 1--Changes in longleaf pine stand structure after application of group selection with the Pro-B method on uplands at the Blackwater River State Forest. Harvest in fall 2006 caused a decline in tree density by winter 2007, mostly in size-classes below 16 inches at d.b.h. Increased density was first noted for 1-inch trees in winter 2008, which translated into increased density for 2-inch trees by winter 2009. Tending across a broad range of diameters released a new wave of recruits for the canopy.

in single-tree selection stands and less than 1 percent in controls. Single-tree selection reduced volume by 28 percent to 1,035 cubic feet per acre, while group selection reduced volume by 25 percent to 1,040 cubic feet per acre, both significantly less than controls. By the second post-treatment growing season, the annual volume growth rate was 3 percent.

Pine Regeneration

On Goethe State Forest flatwoods, grass-stage longleaf pine prior to treatment averaged 20 seedlings per acre, with only those in the group selection stands significantly lower, at 9 seedlings per acre. Two years after treatment, grass-stage longleaf pine significantly increased (140 percent) to 48 seedlings per acre, overall. Without significant differences among stands, including controls, the increase in density could not be attributed to treatment, but was more likely the result of larger-scale factors, such as weather, seed dispersal, and fire cycles. Bolt-stage longleaf pines were present in as-yet very low densities (0 to 3 seedlings per acre). Conditions favoring a rising density in grass-stage longleaf pines have not yet facilitated bolting. Slash pine densities, initially 10 to 14 seedlings per acre, while rising in all stands, significantly increased only in group selection stands to 70 seedlings per acre. On Blackwater River State Forest uplands, grass-stage longleaf pine prior to treatment averaged 1,849 seedlings per acre. Two years after treatment, grass-stage longleaf pine significantly declined by 69 percent to an overall average of 597 seedlings per acre. With no significant differences among stands, the decrease in density could not be attributed to treatment. This decrease was perhaps related to mortality from drought stress, given the incidence of several dry years during that time period. Bolt-stage longleaf pines were initially present at 35 seedlings per acre, with significantly higher densities in group selection stands (73 seedlings per acre). Two years following treatment, bolt densities broadly improved so as to become comparable in all treatments (66 seedlings per acre overall). Increased bolt density likely resulted from the release of grass-stage seedlings already present. Slash pine densities initially ranged from 0 to 9 seedlings per acre and increased significantly only in single-tree selection stands to 52 seedlings per acre.

DISCUSSION

Goethe State Forest Flatwoods

Stand dynamics reflect ecosystem maintenance with prescribed fire and tree removal, causing reductions in density, basal area, and volume. Although treatment produced temporary disturbance, all stands soon stabilized and grew at normal rates. This finding fits the pattern of no growth loss for periodically-burned longleaf pine above sapling size (Boyer and Miller 1994). These stands were characteristic of seldom-burned forests, with low-density regeneration of less than 30 grass-stage and 5 bolt-stage seedlings per acre. Such low levels resulted from competition with saw-palmetto that creates a shrub-canopy with very few openings for seedling establishment. Burning and mechanical disturbance of tree harvest sufficiently diminished shrub cover, so that grass-stage seedling density more than doubled. This is encouraging, since regeneration is a key requirement for sustaining longleaf pine forests (Brockway and others 2006). A strong relationship exists between disturbances like fire and the composition of understory vegetation (Outcalt 2000), with frequently burned stands having fewer woody and more herbaceous plants (Glitzenstein and others 2003). Although fire can reduce shrubs like gallberry (Brockway and Lewis 1997), many burning cycles are needed to reduce shrubs like saw-palmetto, with its extensive rhizomes and capacity for rapid regrowth. Since longleaf pine ecosystems are prone to and highly resilient to disturbances like surface fire and partial canopy reduction (Outcalt 2008, Stanturf and others 2007), they are well suited for management using periodic prescribed burning and the regular cutting cycles of selection.

Blackwater River State Forest Uplands

Stand dynamics were dominated by tree removal during harvest, with reductions in density, basal area, and volume. Regardless of treatment, pine growth continued at normal rates. These stands were typical of forests that are regularly tended and burned with prescribed fire, having a well-developed longleaf pine overstory and a grass-dominated understory with abundant pine regeneration. With no significant difference among treatments, the decline of grass-stage longleaf pine could not be attributed to the reproduction methods. When considering the occurrence of several dry years during this time, the decline in grass-stage seedlings is most likely a result of drought-induced mortality. Although grass-stage longleaf pine seedlings may persist for many years beneath the canopy, the longer they remain unreleased, the greater the probability they will die by being weakened by competition, drought, and/or fire (Boyer 1990, Brockway and Outcalt 1998, Brockway and others 2006). Conversely, the rise in bolt-stage seedlings was encouraging but occurred across all stands and could not be attributed to the reproduction treatments. Conditions causing the grass-stage decline did not impair development of the bolt-stage seedlings. Under conditions of less stress, perhaps a greater number of grass-stage seedlings might have initiated height growth and moved into the bolt stage. Competition intensity can influence the number of seedlings emerging from the grass stage (Haywood 2000, Ramsey and others 2003). Continued management with prescribed fire and periodic selection cutting should maintain conditions favorable for regeneration (Glitzenstein and others 1995, Kush and others 2004, Outcalt and Brockway 2010) and ecosystem resiliency to a variety of disturbance agents (Stanturf and others 2007).

Comparing Selection Methods

Single-tree selection differs from group selection by foregoing deliberate creation of canopy gaps when tending the forest matrix. During each cutting cycle, the stand is reduced to a basal area low enough to initiate regeneration, by harvesting trees across a range of size-classes. Long-term application results in a forest with an irregular canopy, many very small gaps (< 0.25 acre) and a stable uneven-aged structure. While seedlings readily establish among overstory trees, they do not recruit to the canopy until released by disturbances that sufficiently reduce the inhibitory influence of nearby competitors.

Lightning and tree harvest are common disturbances that augment the size of such gaps, thus releasing suppressed seedlings (Moore 2001, Outcalt 2008). With repeated entries, removal of adjacent overstory trees can progressively enlarge very small gaps so they approach the dimensions of those created through group selection (Brockway and others 2006). In actual practice, these two selection methods may seamlessly blend together in the field through time. Ecological forestry provides a useful context for practicing selection silviculture (Franklin and others 2007). By using natural disturbance regimes as a template for management that creates and maintains complex structures, natural processes, biological legacies, and recovery intervals, selection can be used to address concerns about biodiversity, wildlife habitat, productivity, and ecosystem services (Palik and others 2002).

Group selection mimics natural gap-phase regeneration in longleaf pine ecosystems (Brockway and Outcalt 1998). This results in an uneven-aged mosaic of even-aged patches, where a continuous canopy is maintained, and seedlings regenerate in small gaps created by lightning and other local disturbances. Because competition from the overstory limits resource availability, seedling growth benefits most in 0.5-acre gaps at locations distal from overstory trees (Palik and others 2003). Pre-settlement longleaf pine forests were largely uneven-aged, where continuous tree recruitment occurred in areas of $\leq$ 3 acres (Pederson and others 2008). Group selection creates gaps ranging from 0.25 to 2 acres distributed throughout the forest to simulate the desired uneven-aged structure (Brockway and others 2006). Ideally, as the forest matrix is tended, gaps should be cut where advanced regeneration is already present, thereby decreasing the likelihood that they will become occupied by competing woody species. This method is compatible with prescribed fire on a 3-year cycle to control competing vegetation and maintain appropriate conditions for regeneration (Farrar 1996). Gap-based approaches, like group selection, can be used to sustain an uneven-aged forest structure that achieves a range of ecosystem stewardship objectives (Coates and Burton 1997).

The initial overall effect of applying single-tree and group selection with Pro-B was reduction in tree density, basal area and volume. On flatwoods, this was followed by an increase in grass-stage regeneration. Here, selection harvest had low impact on the shrub-dominated understory, with only small reductions (< 10 percent) in saw-palmetto cover that soon recovered. Logging did not diminish shrub cover sufficiently to stimulate expansion of herbaceous plants. Achieving regeneration success with selection is challenging on sites with severe competition, such as flatwoods dominated by saw-palmetto (Farrar 1996, Kush 2002). On uplands, tree density recovered in group selection stands within 2 years of treatment, and grass-stage seedlings declined while bolt-stage longleaf pine increased. Single-tree selection stands were less changed than group selection stands, since deliberately cutting gaps in the canopy alters the spatial pattern of overstory retention and creates a somewhat different understory environment.

Selection Silviculture with Pro-B

As Guldin (2006) noted, BDq can be more easily applied if the number of size-classes is reduced by basing them on five broad product-classes (i.e., small pulpwood, large pulpwood, small sawlogs, medium sawlogs, large sawlogs), and the tree tally is performed as a percent reduction within each product-class rather than as a numeric count of individual trees for each 2-inch size-class. Similar steps for improving BDq efficiency have also been suggested by Leak and others (1987) and Nyland (1987). For example, one long-used target structure for northern hardwoods, with a 20-year cutting cycle, consists of 10 square feet per acre for the 2- to 5-inch size-classes, 20 square feet per acre for 6- to 11-inch size-classes, 30 square feet per acre for 12- to 16-inch size-classes, and 10 square feet per acre for $\geq$ 17-inch size-classes, resulting in a basal area ratio of 1:2:3:1 among residuals (Nyland 2007). However, BDq, with its traditional focus on producing timber, tends to reduce important ecological structures, such as live trees for cavity-nesting species and snags, unless marking rules include retaining older trees and those with cavities (Kenefic and Nyland 2007). Since high vertical structural diversity and a range of cavity heights and sizes can be characteristic of uneven-aged forests, selection is best applied so as to conserve these ecological assets (Kenefic and Nyland 2000).

Pro-B represents a different approach, intended to simultaneously meet biodiversity goals and timber objectives. As a streamlined easy-to-apply method, Pro-B was developed for upland hardwoods in southern Missouri (Loewenstein 2005). However, recent application in riparian hardwoods, loblolly pine, and longleaf pine represented opportunities to try Pro-B in a wider variety of southern forests, and we encourage broader testing on this and other continents. Our application of Pro-B was a successful test for cutting longleaf pine stands to a target basal area, leaving a desirable diameter distribution. Pro-B was easily learned and adeptly applied by practitioners from a range of professional disciplines. The single-pass feature of Pro-B makes it a time-efficient method for practicing selection. With Pro-B, managers achieved the target basal area with a high level of precision. Basal area stabilized early after treatment and then steadily increased through time. Thus, Pro-B is an effective method for implementing selection silviculture. These tests highlighted the need to develop computational technology to quickly and accurately calculate marking guides for field use, following input of inventory data. Thus, the Pro-B Calculator was produced in an English series (with multiple versions from 40 to 100 square feet per acre in increments of 10 square feet per acre) and a metric series (with versions from 10 to 24 m^2 ha^{-1} in increments of 2 m^2 ha^{-1}). The Pro-B Calculator, Pro-B instructions, and selection silviculture literature will eventually be hosted on an internet website (http://www.Pro-B.auburn.edu) and made accessible on a 24/7 basis. Pro-B technology is in the public domain and offered to a wide audience, in hopes of encouraging broader use of selection silviculture and contributing to improved forest management.

CONCLUSIONS

Implementing selection silviculture through Pro-B can harvest high-quality timber, moderately reduce the overstory, and free growing space for the next wave of longleaf pine regeneration. As an easy-to-learn, accurate, efficient, effective, single-pass method, it can be used to improve stand structure by progressively adjusting the size-class distribution through time. With its focus on long-term sustainability through stability of structure, composition and function in residual stands, and flexibility to retain large trees and biological legacies by mimicking small-scale disturbances followed by adequate time intervals for recovery, the Pro-B method might be implemented to attain the timber production objectives and ecological stewardship goals of retention forestry (Gustafsson and others 2012).

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LITERATURE CITED

Boyer, W.D. 1990. *Pinus palustris*, Mill. Longleaf pine. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America: Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 405-412. Vol. 1.

Boyer, W.D.; Miller, J.H. 1994. Effect of burning and brush treatments on nutrient and soil physical properties in young longleaf pine stands. Forest Ecology and Management. 70(1/3): 311-318.

Brockway, D.G.; Lewis, C.E. 1997. Long-term effects of dormant-season prescribed fire on plant community diversity, structure and productivity in a longleaf pine wiregrass ecosystem. Forest Ecology and Management. 96(1/2): 167-183.

Brockway, D.G.; Outcalt, K.W. 1998. Gap-phase regeneration in longleaf pine wiregrass ecosystems. Forest Ecology and Management. 106(2/3): 125-139.

Brockway, D.G.; Outcalt, K.W.; Boyer, W.D. 2006. Longleaf pine regeneration ecology and methods. In: Jose, S.; Jokela, E.J.; Miller, D.L., eds. The longleaf pine ecosystem: ecology, silviculture and restoration. New York: Springer Science: 95-133.

Brockway, D.G.; Outcalt, K.W.; Guldin, J.M. [and others]. 2005b. Uneven-aged management of longleaf pine forests: a scientist and manager dialogue. Gen. Tech. Rep. SRS-78. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 38 p.

Brockway, D.G.; Outcalt, K.W.; Tomczak, D.J.; Johnson, E.E. 2005a. Restoration of longleaf pine ecosystems. Gen. Tech. Rep. SRS-83. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 34 p.

Coates, K.D.; Burton, P.J. 1997. A gap-based approach for development of silvicultural systems to address ecosystem management objectives. Forest Ecology and Management. 99: 337-354.

Croker, T.C.; Boyer, W.D. 1975. Regenerating longleaf pine naturally. Res. Pap. SO-105. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 21 p.

Farrar, R.M. 1996. Fundamentals of uneven-aged management in southern pine. Misc. Pub. 9. Tallahassee, FL: Tall Timbers Research Station. 63 p.

Franklin, J.F.; Mitchell, R.J.; Palik, B.J. 2007. Natural disturbance and stand development principles for ecological forestry. Gen. Tech. Rep. NRS-19. Newton Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 44 p.

Gagnon, J.L.; Jokela, E.J.; Moser, W.K.; Huber, D.A. 2003. Dynamics of artificial regeneration in gaps within a longleaf pine flatwoods ecosystem. Forest Ecology and Management. 172: 133-144.

Glitzenstein, J.S.; Platt, W.J.; Streng, D.R. 1995. Effects of fire regime and habitat on tree dynamics in north Florida longleaf pine savannas. Ecological Monographs. 65: 441-476.

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: 22-37.

Guldin, J.M. 1996. The role of uneven-aged silviculture in the context of ecosystem management. Western Journal of Applied Forestry. 11(1): 4-12.

Guldin, J.M. 2006. Uneven-aged silviculture of longleaf pine. In: Jose, S.; Jokela, E.J.; Miller, D.L., eds. The longleaf pine ecosystem: ecology, silviculture and restoration. New York: Springer Science: 217-241.

Guldin, J.M. 2011. Experience with the selection method in pine stands in the southern United States, with implications for future application. Forestry. 84(5): 539-546.

Guldin, J.M.; Baker, J.B. 1998. Uneven-aged silviculture, southern style. Journal of Forestry. 96(7): 22-26.

Gustafsson, L.; Baker, S.C.; Bauhus, J. [and others]. 2012. Retention forestry to maintain multifunctional forests: a world perspective. BioScience. 62(7): 633-645.

Haywood, J.D. 2000. Mulch and hexazinone herbicide shorten the time longleaf pine seedlings are in the grass stage and increase height growth. New Forests. 19: 279-290.

Hintze, J.L. 2007. Number cruncher statistical system, version 7.1.1. Kaysville, UT: NCSS.

Jose, S.; Jokela, E.J.; Miller, D.L., eds. 2006. The longleaf pine ecosystem: ecology, silviculture and restoration. New York: Springer Science: 438 p.

Kenefic, L.S.; Nyland, R.D. 2000. Habitat diversity in uneven-aged northern hardwood stands: a case study. Res. Pap. NE-714. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station. 4 p.

Kenefic, L.S.; Nyland, R.D. 2007. Cavity trees, snags and selection cutting: a northern hardwood case study. Northern Journal of Applied Forestry. 24(3): 192-196.

Kush, J.S. 2002. Natural regeneration of longleaf pine: adaptations to site conditions and management systems. Auburn, AL: Auburn University. 107 p. Ph.D. dissertation.

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(1): 6-10.

Leak, W.B. 1964. An expression of diameter distribution for unbalanced, uneven-aged stands and forests. Forest Science. 10: 39-50.

Leak, W.B.; Filip, S.M. 1975. Uneven-aged management of northern hardwoods in New England. Res. Pap. NE-332. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 15 p.

Leak, W.B.; Solomon, D.S.; DeBald, P.S. 1987. Silvicultural guide for northern hardwood types in the Northeast. Res. Pap. NE-603. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 31 p.

Loewenstein, E.F. 2005. Conversion of uniform broadleaf stands to an uneven-aged structure. Forest Ecology and Management. 215(1/3): 103-112.

McGuire, J.P.; Mitchell, R.J.; Moser, E.B. [and others]. 2001. Gaps in a gappy forest: plant resources, longleaf pine regeneration, and understory response to tree removal in longleaf pine savannas. Canadian Journal of Forest Research. 31: 765-778.

Moore, J.H. 2001. Managing the forest and the trees: a private landowner's guide to conservation management of longleaf pine. Zebulon, NC: Theo Davis & Sons, Inc. 36 p.

Nyland, R.D. 1987. Selection system and its application to uneven-aged northern hardwoods. In: Nyland, R.D., ed. Managing northern hardwoods: proceedings of a silvicultural symposium. Misc. Pub. No. 13. Syracuse, NY: State University of New York, Faculty of Forestry: 49-80.

Nyland, R.D. 1998. Selection system in northern hardwoods. Journal of Forestry. 96(7): 18-21.

Nyland, R.D. 2007. Silviculture: concepts and application. Long Grove, IL: Waveland Press, Inc. 682 p.

O'Hara, K.L. 1996. Dynamics and stocking-level relationships of multi-aged ponderosa pine stands. Forest Science. 42(4) Monograph 33. 34 p.

O'Hara, K.L. 1998. Silviculture for structural diversity: a new look at multiaged systems. Journal of Forestry. 96(7): 4-10.

O'Hara, K.L.; Gersonde, R.F. 2004. Stocking control concepts in uneven-aged silviculture. Forestry. 77(2): 131-143.

O'Hara, K.L.; Lahde, E.; Laiho, O. [and others]. 2001. Leaf area allocation as a guide to stocking control in multi-aged, mixed-conifer forests in southern Finland. Forestry. 74(2): 171-185.

O'Hara, K.L.; Valappil, N.I. 1999. Masam - a flexible stand density management model for meeting diverse structural objectives in multiaged stands. Forest Ecology and Management. 118: 57-71.

Outcalt, K.W. 2000. Occurrence of fire in longleaf pine stands in the southeastern United States. In: Proceedings, 21st Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station. 21: 178-182.

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

Outcalt, K.W.; Brockway, D.G. 2010. Structure and composition changes following restoration of longleaf pine forests on the Gulf Coastal Plain of Alabama. Forest Ecology and Management. 259: 1615-1623.

Palik, B.; Mitchell, R.J.; Pecot, S; Battaglia, M.; Pu, M. 2003. Spatial distribution of overstory retention influences resources and growth of longleaf pine seedlings. Ecological Applications. 13(3): 674-686.

Palik, B.J.; Mitchell, R.J.; Hiers, J.K. 2002. Modeling silviculture after natural disturbance to sustain biodiversity in the longleaf pine (*Pinus palustris*) ecosystem: balancing complexity and implementation. Forest Ecology and Management. 155: 347-356.

Palik, B.J.; Mitchell, R.J.; Houseal, G.; Petersen, 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.

Ramsey, C.L.; Jose, S.; Brecke, B.J.; Merritt, S. 2003. Growth response of longleaf pine *(Pinus palustris* Mill.) seedlings to fertilization and herbaceous weed control in an old field in southern USA. Forest Ecology and Management. 172: 281-289.

Reynolds, R.R. 1969. Twenty-nine years of selection timber management on the Crossett Experimental Forest. Res. Pap. SO-40. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 19 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: Arkansas Agricultural Experiment Station. 43 p.

Saucier, J.R.; Phillips, D.R.; Williams, J.G. 1981. Green weight, volume, board-foot and cord tables for the major southern pine species. Georgia Forest Res. Pap. No. 19. Dry Branch, GA: Georgia Forestry Commission, Research Division. 63 p.

Seymour, R.S.; Kenefic, L.S. 1998. Balance and sustainability in multiaged stands: a northern conifer case study. Journal of Forestry. 96(7): 12-17.

Shelton, M.G.; Cain, M.D. 2000. Regenerating uneven-aged stands of loblolly and shortleaf pines: the current state of knowledge. Forest Ecology and Management. 129: 177-193.

Slabaugh, J.D.; Jones, A.O.; Puckett, W.E.; Schuster, J.N. 1996. Soil survey of Levy County, Florida. Washington, DC: U.S. Department of Agriculture Natural Resources Conservation Service. 297 p.

Stanturf, J.A.; Goodrick, S.L.; Outcalt, K.W. 2007. Disturbance and coastal forests: a strategic approach to forest management in hurricane impact zones. Forest Ecology and Management. 250: 119-135.

U.S. Fish and Wildlife Service. 2003. Recovery plan for the red-cockaded woodpecker (*Picoides borealis*): second revision. Atlanta, GA: U.S. Department of Interior Fish and Wildlife Service. 296 p.

Van Lear, D.H.; Carroll, W.D.; Kapeluck, P.R.; Johnson, R. 2005. History and restoration of the longleaf pine-grassland ecosystem: implications for species at risk. Forest Ecology and Management. 211: 150-165.

Weeks, H.H.; Hyde, A.G.; Roberts, A. [and others]. 1980. Soil survey of Santa Rosa County, Florida. Washington, DC: U.S. Department of Agriculture Soil Conservation Service. 150 p.

MORE PRACTICAL CRITICAL HEIGHT SAMPLING

Thomas B. Lynch and Jeffrey H. Gove[1]

Critical Height Sampling (CHS) (Kitamura 1964) can be used to predict cubic volumes per acre without using volume tables or equations. The critical height is defined as the height at which the tree stem appears to be in borderline condition using the point-sampling angle gauge (e.g. prism). An estimate of cubic volume per acre can be obtained from multiplication of the sum of the critical heights at a sample point by the point sampling basal area factor. One of the most serious problems with practical implementation of critical height sampling is that trees near the sample point have a very high critical height, which can be difficult to view from the sample point. It is proposed to correct this by obtaining the "antithetic variate" associated with each tree which is:

$$(1\text{-}u^*) = (1 - b/B) \tag{1}$$

where b is the cross-sectional area at critical height of the stem (equal to stem cross-sectional area at borderline) and B is the basal area of the tree.

The value of b can be computed from the distance to the tree, while the value of B will be based on measurements of diameter at breast height (d.b.h.). This 1-u* is used to perform importance sampling on sample trees. This will result in measurement of height to an upper-stem diameter on each sample tree, which will be lower on trees near the sample point and elevated as trees are more distant from the sample point.

Importance sampling is a method of obtaining unbiased tree-volume estimates using randomly selected upper-stem tree heights or diameters. Under importance sampling, a proxy taper function which approximates actual tree shape is used to sample tree dimensions with probability density proportional to proxy volume. We used an importance sampling individual tree volume estimator developed by Lynch and others (1992) using a paraboloid as a proxy taper function. Importance sampling can be combined with critical height sampling by using the following uniform random variate:

$$u^* = b/B \tag{2}$$

where b is the cross-sectional area at critical height for tree i and also the "borderline" cross-sectional area for a tree located at the same distance from the sample point as tree i.

This uniform random variate was then used in developing an estimator for volume per unit area which uses the importance sampling individual tree volume estimate in place of actual individual tree volume in the classic Horizontal Point Sampling (HPS) estimator. We refer to this estimator as the Importance sampling Critical Height Sampling (ICHS) estimator.

With ICHS, the upper-stem height measurement, h(b), is located high on the stem for trees close to the sample point and low on the stem for trees distant from the sample point. However, we can reverse that trend by using the antithetic variate, 1-u*, in Lynch and others (1992) importance sampling individual tree volume estimator. We refer to this estimator as the Antithetic Importance sampling Critical Height Sampling (AICHS) estimator. This solves the problem of steep viewing angles for CHS sample trees near the sample point because the relationship between distance-to-tree and upper-stem viewing height is reversed when 1-u* is used instead of u.

The sampling surface simulator of Gove (2012) was adapted and used to perform simulations to compare the precisions of CHS, ICHS, and AICHS to ordinary HPS. HPS is the "gold standard" because individual tree volumes in HPS are assumed known without error.

[1]Professor, Oklahoma State University, Department of Natural Resource Ecology and Management, Stillwater, OK 74078; and Research Forester, USDA Forest Service, Northern Research Station, Durham, NH 03824.

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Sampling surfaces are generated by evaluating each estimator at every point on a fine grid system covering the simulation tract. Sampling simulations with metric F = 4 were conducted on a reconstruction of a mature shortleaf pine (*Pinus echinata* Mill.) forest having 90 square feet of basal area per acre at age 80 and site index 50 feet at age 50, based on data for mature shortleaf from Huebschmann (2000).

HPS, CHS, AICHS, and ICHS are known to be unbiased from theoretical results since, for each of these methods, the mathematical expected value of the estimator is equal to forest volume for the sampled tract. However it should be noted that the lack of bias in HPS depends on the assumption that individual sample tree volumes come from an unbiased volume table or equation. CHS, AICHS, and ICHS do not require this assumption since they estimate tree volumes based on upper-stem stem measurements in the field. Simulation results empirically confirmed the lack of bias of each estimator. Simulation results indicated that ICHS, AICHS, and HPS had lower standard deviations and therefore were more precise than CHS. Very notable is that the new methods, ICHS and AICHS, are equally as precise as HPS in which individual tree volumes are known without error. This is very important because the new methods, like critical height sampling, avoid bias inherent in volume equations or tables to which HPS is subject.

LITERATURE CITED

Gove, J.H. 2012. Sampling surface simulation. https//r-forge. Rproject.org/projects/sampsurf/. [Date accessed: May 15, 2013].

Huebschmann, M.M. 2000. Restoring the shortleaf pine-bluestem grass ecosystem on the Ouachita National Forest: an economic evaluation. Stillwater, OK: Oklahoma State University. 141 p. Ph.D. dissertation.

Kitamura, M. 1964. Theoretical studies concerning the estimation of stand volume via the sum of ‘Deckpunkt’ heights [in Japanese]. Yamagata University Agricultural Sciences Bulletin No. 4: 365-403.

Lynch, T.B.; Chang, S-T; Chandler, J.P. 1992. Estimation of individual tree volume by importance sampling and antithetic variates from the cylindrical shells integral. Canadian Journal of Forest Research. 22: 326-335.

SAMPLING FOR COMPLIANCE WITH USDA FOREST SERVICE GUIDELINES USING INFORMATION DERIVED FROM LIDAR

Bogdan M. Strimbu, Daniel Cooke, and Samuel Strozier[1]

Forest resources are traditionally assessed using field measurements. The USDA Forest Service developed a series of guidelines for planning and executing the measurements, specifically the significance level and maximum allowed sampling error. The sampling process outlined by the Forest Service includes a pre-sampling phase to supply some of the information needed for the inventory dedicated to assess the resource. The advent of remote sensing techniques, especially LIDAR, reduced the field effort while increasing estimation accuracy. The objective of this research was to determine the sample size needed for assessing forest resources using prior information derived from remote sensing sources. Remote sensing data is available at very attractive prices: LIDAR can be < $2 per acre; stereo images can reach $0.50 per acre.

Traditionally, a forest inventory is executed in two phases: first a pre-cruise with at least five plots, one being a boundary plot, is carried out in the field to determine the coefficient of variation; secondly, the actual cruise is performed using the information from the pre-cruise. The sample size, determined using sampling without replacement [the procedure recommended by the Forest Service (Robertson 2000)] is:

$$n_{pre-cruise} = \frac{1}{\left(\frac{SE}{t_{n-1;\alpha} \times CV_{plot}}\right)^2 + \frac{1}{N_{plots}}} \qquad (1)$$

where $n_{pre\text{-}cruise}$ is sample size executed as recommended by the Forest Service guidelines; SE is sampling error, which is imposed by the Forest Service guidelines; N_{plots} is the number of plots for census, computed as $N_{plots}=A_{stand} / A_{plot}$; CV_{plot} is plot level coefficient of variation; and $t_{n-1,\alpha}$ is the t value for $n-1$ degrees of freedom and significance level α [according to the Forest Service guidelines, α =0.05 (Robertson 2000)].

The availability of remote sensing data allows precise and accurate determination of tree height, either total or to base of crown. Height has been documented to be correlated with total volume (Zeide 1995); therefore, one could argue that the coefficient of variation for volume, when volume is the objective of the forest inventory, can be replaced by the coefficient of variation for heights. The advantage of using heights instead of volumes rests not only in an increase in accuracy and precision but also in using populations and not samples, which adds to the accuracy of the estimates. The validity of the replacement of volume with height is warranted by the linear relationship that exists between them, which allows the translation of the results obtained for height to volume. Using mild distributional assumptions and the linear equation:

$$Volume_{tree} = k \times height_{tree} \text{ (i.e., linearity)} \qquad (2)$$

where k is a coefficient, the coefficient of variation, CV, for plot volume is

$$CV_{plot} = CV_{tree} / TPP^{0.5} \qquad (3)$$

where TPP represents the average trees per plot. This research uses LIDAR data to compute the coefficient of variation of tree height.

Considering the relationship between the coefficient of variation for volume and for tree height (equation 2), the volume of a stand can be estimated from a sample of size:

[1]Assistant Professor and Undergraduate Students, respectively, Louisiana Tech University, School of Forestry, Ruston, LA 71272.

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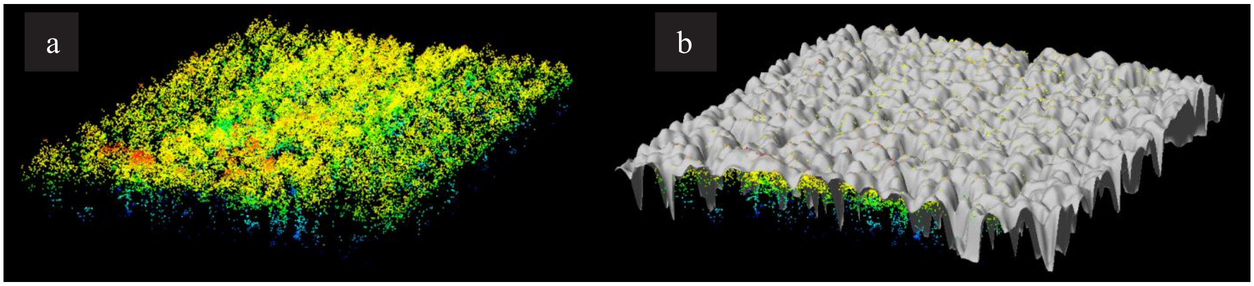

Figure 1--LIDAR point cloud (a) and identification of individual trees (b) using Fusion (McGaughey 2012).

$$n_{LIDAR} = \frac{1}{TPP \times \left(\frac{SE}{t_{n-1;\alpha} \times CV_{treeheight}} \right)^2 + \frac{1}{N_{plots}}} \quad (4)$$

where n_{LIDAR} is sample size determined according to the Forest Service specifications but the *CV* was determined using LIDAR. CV_{tree} is the coefficient of variation of tree height (fig. 1).

The maximum sampling error is established according to the value of the resources; higher values require higher accuracy and therefore smaller sampling error, expressed as percentage from the expected volume (Robertson 2000). Considering that in the sample size formula (i.e., equations 2 and 4) the size of the inventoried stand is a variable, the computations used a fictional stand of 100 acres cruised with fixed area plots of 0.10 acre. This stand size was selected as being large enough to be feasible from the forest operations perspective and easy to adjust to stands of different sizes. The size of the plot was chosen as being widely used in estimating the volume of merchantable timber. The rest of the parameters from equation 4 were selected following a factorial design, with *CV* having three values, (10, 20, and 30 percent); trees per acre (TPA), also with three values (200, 300, and 400); and sampling error, 10 and 20 percent. The *TPP* from equation 4 can be computed as $TPP = TPA \times A_{stand} / N_{plots}$.

The largest number of plots to be ground measured using LIDAR data is at most two (table 1), when stand variability is large (i.e., CV ≥ 20 percent) and number of trees is reduced (i.e., ≤ 300 TPA); otherwise one plot suffices to obtain a forest inventory within the required limits. Alternatively, a forest inventory executed with a pre-cruise has at least one plot, but only for homogeneous stands (i.e., CV = 10 percent) and reduced values, as the sampling error should be 20 percent. For valuable stands, cruising without prior information can require 33 plots, a disproportionate field effort compared with the measurements executed using remote sensing-derived information. The main difference between the two approaches is in the computation of the expected values, with one using the entire population (i.e., the approach using LIDAR data) while the other one uses a sample (i.e., the approach using a pre-cruise) (fig. 1).

Table 1--Sample size using LIDAR data and a ground pre-cruise

CV	TPA	# plots using LIDAR	# plots using pre-cruise
10	200	1	4
10	300	1	4
10	400	1	4
20	200	1	16
20	300	1	16
20	400	1	16
30	200	2	33
30	300	2	33
30	400	1	33
10	200	1	1
10	300	1	1
10	400	1	1
20	200	1	4
20	300	1	4
20	400	1	4
30	200	1	9
30	300	1	9
30	400	1	4

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LITERATURE CITED

McGaughey, R.J. 2012. FUSION. Version 3.30. Seattle, WA: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. http://forsys.cfr.washington.edu/fusion/fusionlatest.html. [Date accessed: Jan 16, 2012]

Robertson, F.D. 2000. Timber cruising handbook. FSH 2409.12. Washington, DC: U.S. Department of Agriculture Forest Service. 268 p.

Zeide, B. 1995. A relationship between size of trees and their number. Forest Ecology and Management. 72(2-3): 265-272.

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A range of issues affecting southern forests are addressed in 108 papers or extended abstracts. Papers are grouped in 13 sections that cover the topics of hardwood regeneration, best management practices for stream crossings, pine regeneration and genetics, fire effects, eco-physiology, forest nutrition, vegetation management, forest threats, biomass, biometrics, growth and development, conservation, and methods.