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**Hot Springs, Arkansas
November 17–20, 2008**

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Proceedings of the 15th Biennial Southern Silvicultural Research Conference

Edited by
James M. Guldin

Hot Springs, Arkansas
November 2008

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Preface

The 15th Biennial Southern Silvicultural Research Conference was held November 17-20, 2008, at the Hot Spring Convention Center, Hot Springs, AR. The 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 pine and hardwood silviculture, wildlife, fire, and forest health. Two field trips focused on shortleaf pine/bluestem restoration and intensive loblolly pine silviculture, with both trips in the Ouachita Mountains. The conference was attended by over 200 people. Ten sessions included 94 oral and 50 poster presentations.

Sponsors for the conference included the Arkansas Forest Resources Center, University of Arkansas; School of Forest Resources, University of Arkansas at Monticello; Arkansas Forestry Association; Arkansas Forestry Commission; Arkansas Division of the Ouachita Society of American Foresters; Weyerhaeuser Company; USDA Forest Service, Ouachita and Ozark-St. Francis National Forests; and the USDA Forest Service, Southern Research Station. The steering committee devoted many hours to reviewing abstracts, establishing the program for oral and poster presentations, and making all necessary arrangements for the conference.

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We would also like to thank Robert Weih and Richard Kluender of the University of Arkansas at Monticello for their help in procuring some of the technology (especially the notebooks) for this conference.

Special recognition is given to the moderators. They included Andy Ezell, Dave Haywood, Nancy Herbert, Gordon Holly, Leda Kobziar, Gary Kronrad, John Kushla, Brian Roy Lockhart, Tom Lynch, Brian Oswald, Jamie Schular, Callie Schweitzer, Marty Spetich, Geoff Wang, Hans Williams, and Jimmie Yeiser. Thanks also goes to all those who helped judge student oral and poster presentations.

A special feature of this conference was the awarding of student travel scholarships. Individuals were selected for this award based on giving a paper at the conference, recommendations from their institutional sponsor, and willingness to assist with audio-visual duties.

The 85 papers published in these proceedings were submitted by the authors in electronic media. Limited editing was done to ensure a consistent format. Authors are responsible for content and accuracy of their individual papers.

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Inserted July 2015

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PLENARY SESSION

View of Compartment 36, an even-aged loblolly pine stand, at the Crossett Experimental Forest, Ashley County, Arkansas. (Photo by James M. Guldin)

ROLE OF THE USDA FOREST SERVICE EXPERIMENTAL FOREST: AN EXTENSION POINT OF VIEW

Eric L. Taylor, C. Darwin Foster, and Diomy Zamora[1]

Abstract—The expansive network of experimental forests (EF) facilitated by the U.S. Forest Service (Forest Service) encompasses a fairly complete representation of the forest ecotypes in the nation. The network, 101 years old this year (2009), has provided researchers with a wealth of long-term data on silviculture, watershed protection, and restoration. However, our nation's needs and expectations have changed dramatically in the last 100 years. A number of structural, financial, and policy challenges must be overcome if the current EF model is to continue to exist. The Forest Service must consider how EFs fit the needs of the public in the 21st century.

INTRODUCTION

The expansive network of experimental forests (EF) facilitated by the U.S. Forest Service (Forest Service) encompasses a fairly complete representation of the forest ecotypes in the nation. The network, 101 years old this year (2009), has provided researchers with a wealth of long-term data on silviculture, watershed protection, and restoration. More recently, research on many of these forests has been redirected toward climate change, greenhouse gas mitigation, and other nontraditional forest ecosystem services from local to global scale. However, our nation's needs and expectations have changed dramatically in the last 100 years. The World has changed. Numerous questions exist regarding the future of the EF concept; as well as the concept of forestry research within the Forest Service, universities, and private industry. In this article we reflect upon the Southern Research Station (SRS) EF design, it's effectiveness, and potential strategies for ensuring remarkable work and relevance well into the 21st century. We pay particular attention to societal changes that push the need for changes in the EFs network architecture and present some marketing strategies to increase positive public perception and visibility.

Section 1 Success Stories

The mission of the SRS is to create the science and technology needed to sustain and enhance southern forest ecosystems and the benefits they provide. They have done a remarkable job in this mission. However, after we present the following, it may become evident that key language that reflects the need for new direction and commitment is missing from the current mission statement.

The SRS, headquartered in Asheville, NC, has long made their network of EFs an integral part of the national infrastructure for scientific knowledge. Research conducted by the SRS 130 scientists, support staff, 19 EFs and partnerships with State forest services, universities, and industry across the 13 Southern States have made key discoveries with far reaching and significant impacts, if not appreciated, on environmental policy, resource management, and the well-being of public citizens. For the most part scientists in research units use these as sites for studies and demonstration projects in conjunction with the management of national forest units. Each SRS EF purposely represents a specific ecosystem that presents unique opportunities to study different strategies for sustaining forested ecosystems and rehabilitation of deteriorated soil. Overall, SRS forestry research emphasizes measuring and monitoring forest resources; understanding ecosystem structure, function, and processes; managing resources for sustained and enhanced productivity; and protecting environmental quality.

Following are examples of areas of "cutting edge" research that too frequently have not been recognized as accomplishments by EF network. Among the experiments conducted on these forests are studies relating to regeneration and management of upland forest ecosystems as on the Crossett EF, to Appalachian ecosystems with Bent Creek EF. Studies on the Delta EF and nearby Sharkey Restoration Research and Demonstration Site (Gardiner and others 2008) have provided fundamental knowledge on watershed studies and bottomland hardwood ecosystem restoration. These studies have led to a major shift in forest management policy in that region and beyond. The effects of pollution, climate change, and timber harvest on Pine Management and Disturbance Science are a major focus at the Hitchiti EF. Research at the Palustris EF played a pivotal role in the development of early reforestation techniques for the four major southern pines that help convert a region of once decimated forests to one where forestry is of leading economic importance. The Stephen F. Austin EF focuses its efforts primarily to understanding and maintaining populations of wildlife species that have, or are becoming threatened, endangered, or sensitive.

These examples demonstrate how long-term, interdisciplinary studies in all 77 EFs throughout the United States are key to new discoveries and innovations that serve as the seedbed to continued health and productivity of our nation's forest ecosystems.

A Record of Excellence

EFs have allowed researchers to excel in providing a wealth of long-term datasets instrumental to understanding

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the dynamics of trees, watersheds, and wildlife habitats; measuring and monitoring forest resources; understanding ecosystem structure, function, and processes; managing resources for sustained and enhanced productivity; protecting environmental quality; and a multitude of other areas and topics. From the knowledge gained from EFs, SRS researchers have also been able to develop many notable outreach efforts. For example, technical tools such as the Forest Inventory and Analyses Web site, field demonstrations and tours, well-maintained Web sites, well done and award-winning magazines, and the more than 26,000 online publications are all invaluable resources to forest researchers and practitioners.

In essence SRS scientists have done a noteworthy job in gaining new knowledge and making the knowledge available to other scientists and resource educators. However, the education/outreach component has not been well structured. As a result the "average" professional and general public is often unaware of these Forest Service efforts because Forest Service researchers have not adequately kept the chain of custody as materials are disseminated. In other words, researchers are not being recognized as the ones that developed the innovation or information and EFs are not recognized as the venue used by researchers to obtain the innovation.

The Big Question ... Relevant in the Future?

What is the role of a network of Forest Service, university, and other EFs in the 21st century? How do EFs fit, if they do fit, as an element of southern forestry research in the future? Should the mission of EFs be expanded to include dissemination, outreach, evaluation, and marketing success? A number of structural, financial, and policy challenges must be overcome if the current EF model is to continue to exist. Perhaps the most obvious obstacle is the restructuring of forest industry into timberland investment management organizations (TIMO) and real estate investment trusts (REIT). This change has led to the elimination of research divisions and exchange of timberland. Even though new owners may participate in forest research, TIMOs and REITs are less likely to significantly contribute funding for long-term, traditional forest silvicultural studies (Berry 2008). A more critical challenge, however, lies with the U.S. Department of Agriculture and other related agency bureaucracy and restrictive regulatory formalities. A long list of laws restricts management of Federal lands, e.g., National Environmental Policy Act, Endangered Species Act, and Clean Water Act, etc. has diminished Forest Service's ability to conduct research (Berry 2008). Another critical area resides with funding wildfire suppression efforts. Each year, a seemingly larger portion of Forest Service's flat line budget is redirected to catastrophic wildfire management. The resulting ebbed research budgets are hardly adequate to accomplish the mission of SRS.

Serious discussion of the role of EFs in the 21st century is moot if policy reform does not occur. If adequate policy reform is anticipated, e.g., budgets are reworked so that research budgets are not raided for wildfire suppression appropriations, additional challenges must still be addressed before the success and survival of the EF network is ensured.

The Forest Service must consider how EFs fit the needs of 21st-century public. Does the public see any value to EFs and the work being done? If sufficient value is not met or realized by overwhelming public backing, future funding and support will be difficult to obtain. This will be a daunting task because public desires and expectations have changed significantly since EF programs were established more than 100 years ago. In that short time the United States has gone from a rural agrarian economy, to an industrial economy, to an urban information and service-driven economy (Hammond 2003). It is very important to note that as of May 23, 2007, the World became urban (Wimberley and others 2007). More people now live in urban areas than in rural areas. With this "Urban Millennium Milestone" comes an unprecedented change in public attitude and understanding. Fewer and fewer people are aware of the existence of the EFs and the value they bring. People holding the purse strings and making policy decisions at the county, State, and Federal levels are likely not forest landowners and neither are their peers. The public does not "talk shop" about forestry and does not likely know about the complexities and necessities of well-managed forests. Much of the information they absorbed is probably not science based. Human nature makes it difficult to value and support what one does not understand. As the EF structure currently operates, the nonforest-owning public has few opportunities to know about the contributions and value of the Forest Service EF network. Politicians will only approve funding for EFs as long as their constituents support such expenditures.

With the proliferation of information and methods of receiving it, our clientele is too frequently inundated. The pressure to be more accessible, more useful, quicker, better, smarter, and cheaper grows seemingly exponentially. Evolving social issues and science questions call for increasingly broad scale and interdisciplinary research and different approach to how the message is delivered.

SOLUTIONS

Extension Can Help!

The common charge of each partner within the cooperative extension system is seemingly a simple one: improving the lives of people, businesses, and communities through high-quality, relevant education. Most have a vast network of county extension offices, agents, and subject-matter specialists to carry out this mission. Extension employees work very closely with communities. Because of this, each county extension agent, in counties with forested acres, likely knows and works with at least one large landowner who is willing to install and maintain long-term research/demonstration plots. Because of extension's vast network, a well-devised, funded memorandum of understanding with extension could establish cooperative projects, joint appointments, and Forest Service station-based extension specialists that would serve as invaluable resources to

ensure the reform and relevance of EFs. This partnership could develop methods and strategies to make EFs useful to a large and diverse customer base, build capacity through nontraditional partnerships and collaborative efforts, develop effective ways to rapidly move research into practice, evaluate results, and convey value to individuals and organizations.

The Forest Service excels in identifying research needs on a national and global scale and conducting the research to produce an innovation or new knowledge ... two of the critical innovation process steps. A collaborative agreement with cooperative extension system would assist the current EF network in addressing the remaining crucial steps of an effective innovation process (Leonard and Sensiper 1998):

1. Dissemination of knowledge to practitioners
2. Implementation of the innovation
3. Evaluation of the innovation
4. Identification of concomitant needs and/or contemporary highly visible societal problems and needs
5. Conveyance of the accomplishments and importance of the EF network to the public (marketing)

The successful completion of each of these steps is crucial to the success of the EF network, and each must be carefully considered during the planning stage. None can be just an afterthought.

Marketing .... Who Needs It?

The old adage of "the best marketing is excellence in research" is not enough and programs built around this philosophy are doomed. The need for marketing is a reality. Without it, we may not have the opportunity in the future to conduct research or practice our programmatic expertise. A strong, well-thought-out marketing plan offers many more benefits than just the obvious. For example, successful marketing can build a sense of team and pride. Morale and productivity is heightened, and a strong image is an effective recruiting tool. Marketing efforts also help secure public funds. Political audiences have to be "sold" on the value. A successful marketing plan will provide the compelling, easily absorbed stories and data required of time-constrained politicians. Marketing efforts are also essential to compete with the multiple information outlets encroaching on EF message and competing for the public's time. The EF network must still satisfy the needs of clientele because as good as any marketing might be, it should not be used as a substitute for good research and program delivery.

Future Experimental Forests

Extension's network may be utilized to develop smaller but more frequent demonstration sites designed to tell a story. Unlike the original requirement of EFs to be strategically located in the representative ecosystem, these "story telling" sites should be strategically located for visibility on lands obtained through partnerships with State, city, or private lands. Future EFs should showcase high profile projects that can tell the story in short, succinct bursts of information. Future EFs will serve as conduit where researchers and stakeholders can work collaboratively to answer and demonstrate pressing practical questions while, at the same time, developing a test bed to explore new techniques or theoretical developments. Future EFs will be located on neutral and secure area unrestrained by agency policies and red tape because neutrality attracts interest from and spawns collaboration between multiple organizations.

Future EFs will involve a wide range of stakeholders, i.e., scientists, managers and practitioners, forest landowners, woodlot owners, and general public, in the initial study design of research questions. This involvement promotes "buy in" by stakeholders and ensures that the resulting knowledge will be effectively and efficiently transferred. This is not a novel idea. The cooperative extension system has been employing this method since its conception. However, the idea does need a major renovation. Future EF outreach and evaluation efforts should implement information technology in order to reach a larger number of people, monitor the progress of research in real time, and capture and archive demonstrations for future use. Future EFs will use advisory groups made up of stakeholders who can help identify high profile issues for demonstrations to aid in adoption of appropriate policies. Sites should be representative of the problem and easily accessible on site or made virtually accessible on the Internet in a manner suitable to the way in which people currently use the Internet.

Future EFs will collaborate with other broad-scale observation programs, e.g., U.S. Geological Survey, National Aeronautic & Space Administration, Long-Term Ecological Research, to jointly study ecological issues, how they respond to human activities, and how humans respond to changing ecological pressure (Lugo and others 2006). In the future EF network, the push-pull dynamic that managers are currently imposing upon their scientists will no longer be directed specifically towards the practitioner (Rains 2006). Instead, the push-pull dynamics will be clearly defined between the Forest Service researcher and the extension specialists. Extension can repackage the message, disseminate the information or implement the innovation, evaluate the results, and provide feedback to the researcher. These are all the primary goals of extension.

CONCLUSION

The discussion of the role of the EF network is just one of a long list of discussions about the need for effective ways to move research into practice and the proper role of scientists and managers to accomplish this goal. Many challenges exist in the current EF network. Some of which are policy driving and must be remedied within the U.S. Department of Agriculture system. Many other challenges, however, can be addressed through a well thought-out collaborative agreement with the expansive network of extension professionals.

LITERATURE CITED

Berry, A. 2008. Commentary: a second century of Forest Service research. The forestry source. Society of American Foresters. 13(11): 10.

Gardiner, E.; Stanturf, J.; Leininger, T. [and others]. 2008. Establishing a research and demonstration area initiated by mangers: the Sharkey Restoration Research and Demonstration Site. Journal of Forestry. 106(7): 363–369.

Leonard, D.A.; Sensiper, S. 1998. The role of tacit knowledge in group innovation. California Management Review. 40(3): 112-132. [Reprint #CMR111]. [Reprinted with new introduction in Choo, C.W.; Bontis, N., eds. 2002. Strategic management of intellectual capital and organizational knowledge. New York: Oxford University Press]

Lugo, A.E.; Swanson, F.J.; Gonzãlez, O.R. [and others]. 2006. Long-term research at the USDA Forest Service's experimental forests and ranges. BioScience. 56(1): 39–48.

Rains, M.T. 2006. The future of long-term USDA Forest Service research sites in the Northeast. In: Irland, Lloyd C.; Camp, Ann E.; Brissette, John C.; Donohew, Zachary R., eds. Long-term silvicultural & ecological studies: results for science and management. New Haven, CT: Yale University: 226–228.

Varea-Hammond, S. 2003. Marketing cooperative extension at the local level. Salinas, CA: University of California Cooperative Extension, Monterey County, California.

Wimberley, R.; Morris, L.; Fulkerson, G. 2007. World population takes a turn. The News and Observer. May 23: 15A. [Reprinted in The Rural Sociologist. 27(3): 18–19.

PINE SILVICULTURE SESSION I

Release treatment with herbicide to remove hardwoods competing with desired pines; stand on left treated with imazapyr the previous fall, stand on right was not treated. Crossett Experimental Forest, Ashley County, Arkansas. (Photo by James M. Guldin)

BEFORE AND AFTER COMPARISONS OF TREE HEIGHT IN SUCCESSIVE LOBLOLLY PINE PLANTATIONS WITH INTERVENING MACHINE, WHOLE-TREE HARVESTING

Thomas J. Dean, D. Andrew Scott, and A. Gordon Holley[1]

Abstract—In 1993, several forest industries, the U.S. Forest Service Southern Research Station, Louisiana Tech University, and the School of Renewable Natural Resources in the Louisiana State University Agricultural Center formed a cooperative that came to be called Cooperative Research in Sustainable Silviculture and Soil Productivity. One of the objectives of the cooperative is to determine whether typical, high-production harvesting is detrimental to the growth of newly established plantations. Prior to harvesting, dominant and codominant trees were severed and discs cut from the stems at 0.5-m intervals up to 10 m in height. Annual heights of the trees were determined through stem analysis. Height comparisons were made between trees from the previous plantation with those of the current plantations up to age 12. For the four sites considered, no changes in productivity were evident as a result of harvesting as measured by the height of the tallest trees in the successive generations.

INTRODUCTION

At the beginning of the 1990s, the U.S. Forest Service began establishing a Long-Term Soil Productivity Study (LTSP) on national forests nationwide to identify the growth response of major forest types to extremes in soil compaction and organic matter removal. The study is motivated by a requirement in the National Forest Management Act that management practices must preserve site productivity. Powers (1999) identified these two factors as the principal causes of any potential loss of site productivity due to harvesting. Heavy harvesting equipment can compress the soil, and efficient harvesting usually requires complete removal of the tree.

The LTSP study is providing much interesting information (Powers and others 2005), but it does not directly address the question of whether high-production harvesting reduces site productivity. Between 1995 and 2003 a series of studies were installed as part of the Cooperative Research in Sustainable Silviculture and Site Productivity (CRiSSSP) to address this question in loblolly pine plantations (table 1). The CRiSSSP protocol incorporates two harvesting intensities. The first is high-production harvesting where the tree is cut with a feller buncher and the whole tree yarded with a grapple skidder. The second is minimum disturbance harvesting where only the commercial portion of the bole is removed leaving the associated vegetation and forest floor intact and the underlying soil nearly undisturbed.

The objective of this analysis is to compare, to the extent possible, the before and after effects of these harvesting treatments on site height. Site height is the mean height of some subset of the tallest trees on the site and is most reflective of site and environmental conditions. A conifer's leader meristem is largely insensitive to crowding (Lanner 1985). It sits in a preferred position with the hydraulic architecture of tree (Zimmerman 1983). Furthermore, site height is the only metric that can be analyzed across rotations since it can be reconstructed from stem analysis.

METHODS

Experimental Design

CRiSSSP installed experiments in six locations across the South to address the question of whether high-production harvesting affects the growth of loblolly pine (*Pinus taeda* L.). Data from four of these installations are used in this particular analysis (table 1). Each location was originally occupied by a stand of loblolly pine that was either artificially or naturally regenerated. The field design is described in detail by Carter and others (2006). To summarize, the two harvesting intensities are crossed with one or two other establishment treatments such as bedding or broadcast burning to create either $2 \times 2 \times 2$ or 2×3 factorial treatments. The eight or six treatments were randomly assigned to eight or six, approximately 0.12-ha plots within three or four blocks. A broadcast application of a mix of herbicides was applied at each location for competition control, especially for competing trees and shrubs. At the two youngest installations, planting was followed by herbicide release from herbaceous competition, based on the widespread positive response of tree growth to this treatment. Each level of establishment treatment included the absence of the treatment so that within each block, two plots represent only the effect of harvesting and competition control. Just these two treatments at four locations are included in this study (table 1).

Plots were planted with 1-year-old, half-sib, loblolly pine seedlings the winter after harvesting and site preparation. Each plot contained 14 rows of 14 seedlings. Rows were generally spaced 3 m apart, and seedlings within rows generally 2 m apart. The outer 2 rows and outer 2 seedlings on each row served as a buffer, leaving 100 potential measurement trees.

Measurements

Height accumulation of the previous stand at annual intervals was reconstructed with stem analysis of a subsample of dominant and codominant trees—the number of trees analyzed

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Table 1—Some physical characteristics of the sites used in this study

Location designation	Physical location	Soil subgroup	Mean annual precipitation	Site index	Harvests[a]
			inches	*feet*[b]	*number*
Fred	Fred, TX	Oxyaquic Paleudult	53.54	59	2
Bryceland	Bryceland, LA	Typic Hapludult	53.94	59	2
Black Lake	Campti, LA	Glossaquic Paleudalf	53.94	56	1
Lee Forest	Pine, LA	Typic Fragiudult	64.17	66	2

[a] Including the harvest treatment.

[b] At 25 years.

depended on the size of the trees. Disks were cut at regular intervals along the stem (cf. Carter and others 2006). Height was interpolated within disc interval with the method recommended and described by Dyer and Bailey (1987). Site height for the previous stand is block averages of the height at a given age.

Height of the surviving trees within the measurement plots was measured on a nearly annual basis using a hypsometer. Site height in response to the harvesting treatments is the average height of the tallest half of the trees within a plot. Age of the trees in the experimental plantations is age from germination.

Statistical Analysis

The effects of regeneration practices on height growth of the subsequent plantation were tested within individual installations by age using a randomized, complete-block analysis of site height for the previous stand and the two harvesting treatments combined with the "zero" level of the additional factorial treatment(s). The null hypothesis tested with this analysis is no effect of harvesting intensity on site height. A probability of a greater F statistic <10 percent for three treatments was considered evidence of a harvesting effect on potential productivity. The probability of erroneously rejecting the null hypothesis was set to 10 percent because of the potential consequences of not detecting a change in site productivity. Since the trees sampled from the previous stand were selected to represent the best trees in the stand, the comparison may be biased toward rejecting the hypothesis that harvesting has no effects on height growth.

RESULTS

During the interval height was measured on the current plantations, average height steadily increased with age for trees in the previous stand and trees planted on the low- and high-intensity harvesting treatments. During the early stages of current plantation, average height lagged behind the height of the sampled trees from the previous plantation. However, at each location, the average height of the current plantation eventually matched or exceeded the height of the trees sampled before harvesting (fig. 1). The differences were statistically significant during the first 3 years after planting at the Fred site and the first year after planting at Lee Forest (table 2). There were no detectable differences in average height before and after harvest nor as a result of harvesting intensity at the Black Lake site and for the Fred site with the exception of one age. At both the Fred and the Lee Forest sites, the average height of the trees planted on the mechanically harvested plots was higher than the average height of the trees in the previous stand and trees planted on the plots harvested with chainsaws and lifts. At Lee Forest the difference in mean height was significant at age 5 years.

Harvesting effects on average height at the Bryceland site were more consistent within treatment and somewhat different than those observed at the other three sites. For the first 4 years after planting, average height of the trees planted after low- and high-production harvesting was less than the height of correspondingly aged trees in the previous stand (table 3). Furthermore, the tallest half of the trees planted on the intensively harvested plots was significantly shorter than the tallest half of the trees planted on the plots harvested with hand tools and lifts. By age 9, the average height of the trees planted on the hand-felled plots was significantly taller than the average height of the trees in the previous stand and the trees planted on the intensively harvested plots. Trees on the hand-felled plots remained taller than trees from the other two treatments for the remainder of the study, though the differences were not statistically different after age 10 years.

DISCUSSION

Evaluating harvesting effects on site height is equivalent to evaluating harvesting effects on site index if the growth curves remain unchanged between generations. For the sites included in this study, no lasting harvesting effect on site quality is evident. Where height of the previous stand was significantly taller than the site height of the current stands, the differences occurred early in plantation and were short lived at Fred and Lee Forest (table 2). At Bryceland where the differences lasted longer, by age 6 years, the site height of the trees planted on the hand-felled plots were as tall as the site height of the previous stand (fig. 1).

One of the problems of using a site height as a measure of generational changes in productivity is the possible

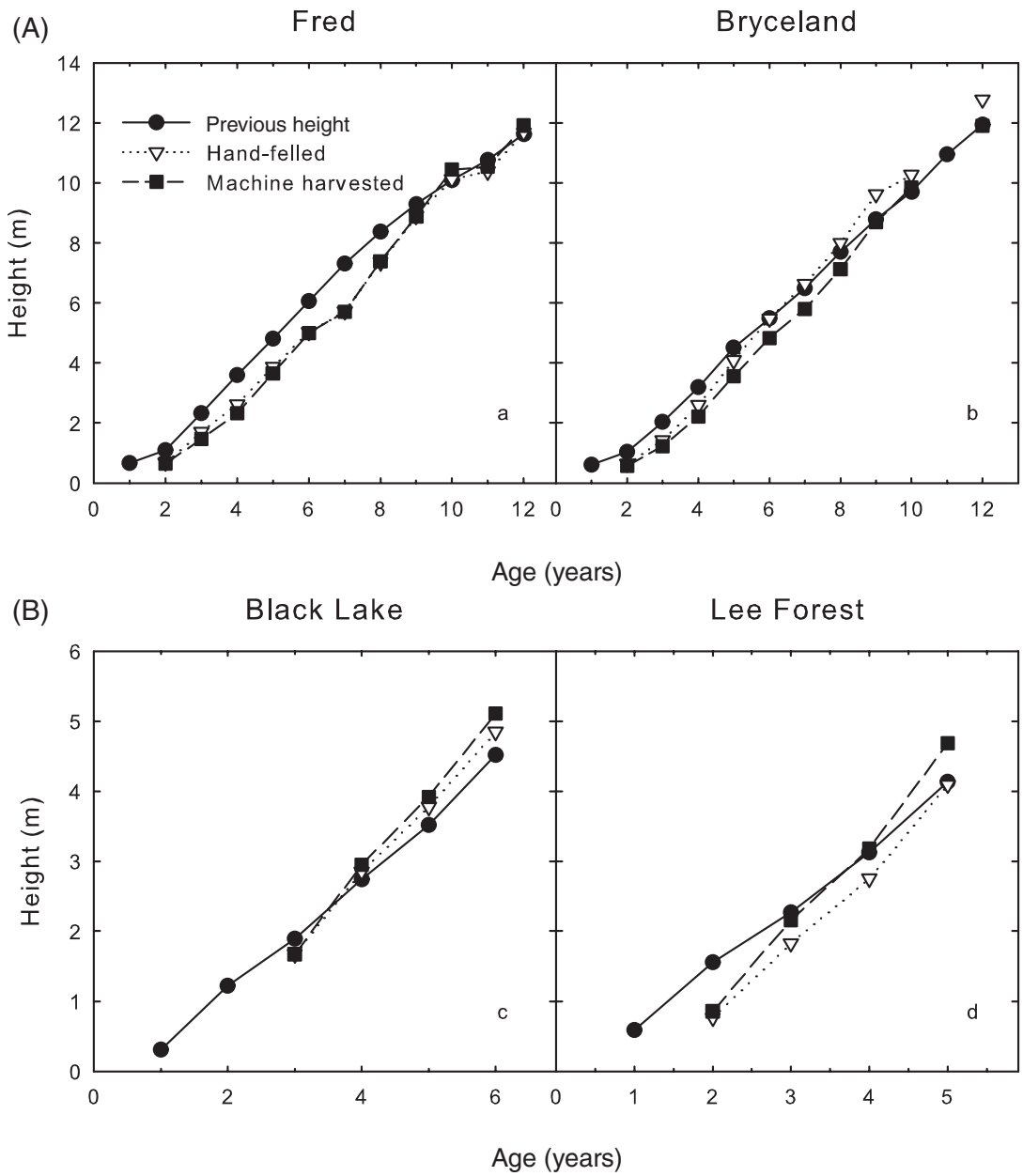

Figure 1—Site height as a function of tree age for loblolly pine at the four sites used in this study.

confounding influence of weather. In a study with slash pine (*P. elliottii* Engelm.) where the second rotation on a site duplicated the site preparation, seed source, planting style, and spacing of the first, Rose and Shiver (2002) observed significant reduction in the average height of the dominants and codominants between the first and second rotations. Since the study was replicated across a range of soil types, they attributed the difference to drought events and high temperatures that occurred during the first two growing seasons of the second rotation. Insufficient data were collected on the four sites used in this study to compare temperature and rainfall patterns between the generations; however, a regionwide, severe drought occurred during the year 2000 and the first half of 2001. At the Fred site, the drought caused only a temporary reduction in height increment at age 7 years; height growth appeared to accelerate after cessation of the drought (fig. 1A). The drought occurred during the 6th growing season at the Bryceland site: no lasting effects were evident (fig. 1B). Black Lake and Lee Forest were established after the drought. Height growth at these sites appears to be resilient against temporary reductions in rainfall.

CONCLUSIONS

While high-production harvesting causes a significant perturbation with regards to the amount of organic matter

Table 2—Probabilities for a greater value of F for the null hypothesis: no harvesting effect on site height

	Site			
Age	Fred	Bryceland	Black Lake	Lee Forest
1				
2	0.005	<0.001		0.024
3	0.051	<0.001	0.854	0.163
4	0.046	0.001	0.753	0.046
5	0.128	0.009	0.822	0.066
6	0.195	0.004	0.415	
7	0.076	0.057		
8	0.375	0.085		
9	0.787	0.076		
10	0.823	0.341		
11	0.868			
12	0.847	0.169		

Table 3—Comparisons of site height by age at the Bryceland site using the Fisher's protected least significant difference procedure with alpha = 0.10

	Treatment		
Age	Previous	H0	H1
2	A	B	B
3	A	B	C
4	A	B	C
5	A	B	C
6	A	A	B
7	A	A	B
8	A	AB	B
9	B	A	B
10	A	A	A
12	A	A	A

H0 = hand-felled, bole-only harvest; H1 = high-production, whole-tree harvest.

Different letters within a row indicate significant differences. Significantly lower means indicated with higher letters in the alphabet.

displaced and compaction of the soil surface, for three-quarters of the sites used in this study, this was only the second tree harvest from these sites. Soil processes appear to recover quite quickly (Carter and others 2002). Furthermore, tree growth appears to be quite resilient to changes in soil properties (Powers 1999). The results of this analysis add further evidence that commercial harvesting of forests is largely benign to the long-term productivity of loblolly pine plantations.

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LITERATURE CITED

Carter, M.C.; Dean, T.J.; Wang, Z. [and others]. 2006. Impacts of harvesting and postharvest treatments on soil bulk density, soil strength, and early growth of *Pinus taeda* in gulf Coastal Plain: a long-term soil productivity affiliated study. Canadian Journal of Forest Research. 36(3): 601–604.

Carter, M.C.; Dean, T.J.; Zhou, M. [and others]. 2002. Short-term changes in soil C, N, and biota following harvesting and regeneration of loblolly pine (*Pinus taeda* L.). Forest Ecology and Management. 164(1-3): 67–88.

Dyer, M.E.; Bailey, R.L. 1987. A test of six methods for estimating true heights from stem analysis data. Forest Science. 33(1): 3–13.

Lanner, R.M.1985. On the insensitivity of height growth to spacing. Forest Ecology and Management. 13(1-2): 143–148.

Powers, R.F.1999. On the sustainable productivity of planted forests. New Forests. 17(1): 263–306.

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 soils research: theory, reality and its role in technology—selected and edited papers from the 10th North American forest soils conference. Forest Ecology and Management. 220(1-3): 31–50.

Rose, C.E., Jr.; Shiver, B.D. 2002. An assessment of first and second rotations average dominant/codominant height growth for slash pine plantations in south Georgia and north Florida. Southern Journal of Applied Forestry. 26(2): 61–71.

Zimmermann, M.H. 1983. Xylem structure and the ascent of sap. Berlin: Springer-Verlag. 143 p.

COMPARISON OF THREE SITE PREPARATION TECHNIQUES ON GROWTH OF PLANTED LOBLOLLY PINE 6 YEARS AFTER A SOUTHERN PINE BEETLE EPIDEMIC

Wayne K. Clatterbuck and Michael Carr[1]

Abstract—Three site preparation treatments: (1) complete removal of woody debris—drum chopped, raked, and disked; (2) drum chopping leaving woody debris; and (3) no site preparation—planting among dead standing trees were compared by evaluating the growth and survival of planted loblolly pine (*Pinus taeda* L.) after six growing seasons following a southern pine beetle (*Dendroctonus frontalis* Zimmermann) epidemic. Each treatment was replicated three times at one location on the Cumberland Plateau in Tennessee. Each treatment had the same number of planted seedlings (681) per acre, and was sprayed with herbicide to control hardwood residuals before planting and to release seedlings one growing season after planting. Results indicate that the growth (height and diameter) of seedlings was not significantly different between the treatments. However, survival was only slightly, but significantly different, for the no-mechanical-site-preparation (standing-dead) treatment which may be a reflection of difficult planting conditions. A cost evaluation of the different site preparation treatments is also discussed.

INTRODUCTION

The Cumberland Plateau and east Tennessee suffered a major southern pine beetle epidemic in 1999 to 2001. More than 30 percent of the pine (90,000 acres) on the Cumberland Plateau was impacted and killed by southern pine beetle (Clatterbuck and others 2006). With the presalvage and salvage operations that occurred during this time, pine was in high supply, but demand was low resulting in low stumpage prices. Some salvage of dying and dead pine stands occurred, but many dying stands were left uncut because harvest costs were greater than the potential revenue.

What is happening to this forest land where pines succumbed to southern pine beetle, especially those areas where trees were not harvested and dead standing trees remain? Three scenarios are possible: (1) some will be replanted to pine, (2) some will be left alone and through natural regeneration will transition to hardwood or mixed hardwood-pine forest types, and (3) some will be converted to nonforest uses. One of the major obstacles to replanting with pine is the cost of site preparation in these standing-dead, pine beetle areas. The objectives of this study were to (1) evaluate three site preparation treatments for survival and growth of planted loblolly pine after 6 years in uncut stands that had succumbed to southern pine beetle and (2) determine the cost-effectiveness of each treatment based on pine growth and survival.

METHODS

Study Area

This study was conducted on the Cumberland Plateau in Cumberland County, TN (longitude 84.46° W, latitude 35.54° N). The area is considered the "true plateau" where the surface is undulating and rarely exceeds slopes of 10 percent (Smalley 1982). The working unit is in several tracts composing an estimated 5,000 acres and was formerly owned by Bowater Incorporated. Presently, the area is in its third rotation of pine plantation. Southern pine beetle attacked the area in 1999 to 2000 during the second rotation when the trees were 18 years old. The third rotation was planted at 8- by 8-foot spacing during the spring of 2002. Soils are moderately productive (site index for yellow pine at 50 years ranged from 70 to 80 feet) and belong to the Lily-Gilpin-Jefferson soil series complex (mesic, typic hapludults) (McGowan 2006). Climate, geology, topography, and forest site classification of the study area may be referenced in Smalley (1982). A timeline of events that occurred in the study is presented in table 1.

Table 1—Timeline of events occurring for the site preparation study of standing-dead pine trees, Cumberland County, TN

Season and/or year	Event
1999–2000	Pine beetle infestation
Summer 2001	Mechanical-site preparation
Fall 2001	Initial herbicide application before planting
Early spring 2002	Hand planting pine
Late summer 2002	Pine-release herbicide application
Fall and winter 2007–08	Data collection

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Treatment Implementation

Three site preparation treatments were distributed in separate stands across the tracts: (1) drum chopping where residual material remained fairly evenly dispersed on the ground; (2) drum chopping, raking, and disking where little residual material was left on the ground surface; and (3) no site preparation where dead trees were left standing. Stands were only sampled where dead trees were left standing prior to the implementation of the site preparation treatments.

Experimental Design and Data Collection

The three site preparation treatments were located in separate stands within the working area. Three stands were sampled for each treatment with 6 plots per stand yielding 54 total plots.

Transects were established in each stand with plots taken every 150 feet starting at least 50 feet from the edge of the stand. Multiple, parallel transects were used (at least 100 feet apart) in a stand if all the plots could not be established on one transect. Each plot consisted of 4 rows of 7 trees (28 trees at 8- by 8-foot spacing) or a 32- by 56-foot rectangular plot (approximately 1/25-acre plot). Data collected at each plot were tree survival counts, total height of the four corner trees (if available, otherwise an adjacent tree was measured if a corner tree was missing), and diameters of the same four trees.

Site Preparation Costs

Average site preparation costs were formulated from standard regional data (Smidt and others 2005) and from surveys of contractors implementing the practices or treatments in the area. All tracts, and thus all treatments, incurred the following costs: initial herbicide application before planting, release herbicide application the first growing season after planting, planting labor, and seedling cost (table 2). There was no differentiation in planting costs between the three treatments even though planting was more difficult and time consuming in the standing-dead (control or no-site-preparation) treatment. The same number of trees (681 trees per acre at 8- by 8-foot spacing) was planted for each treatment. The cost of site preparation treatments is quite different ranging from no cost for the standing-dead or control treatment to $250 per acre for the most intensive treatment (drum chop, rake, and pile) (table 2).

RESULTS

Total height and diameter of loblolly pine were not different between treatments averaging 16 to 18 feet and 2.3 to 3.1 inches, respectively, after six growing seasons (table 3). However, tree survival did differ between the control (standing-dead) and the more intensive treatments. Tree survival averaged 78 percent in the control and 86 to 89 percent in the other two site preparation treatments (table 3). Survival was similar between the drum chop only and the drum chop, rake, and disk treatments.

DISCUSSION

The impact of competition control (whether mechanical, chemical, or both) and its positive effects on loblolly pine growth and development are well documented (Haines and others 1975, Minogue and others 1991, Neary and others 1990) and thoroughly reviewed (Fox and others 2008). Generally, chemical treatments for site preparation are used to deter herbaceous growth and hardwood sprouting and growth prior to pine planting. Herbicides are also used after planting for pine release. Mechanical methods of site preparation provide greater accessibility for planting through slash disposal as well as incorporating organic material into the soil and improving soil physical properties. Aerial chemical treatments were used on all stands in the study to control hardwoods and herbaceous growth both prior to and after planting pine seedlings (table 2). Thus, the question in this study was whether the cost of removing standing-dead trees through mechanical site preparation was justified through potential increases in growth and survival of planted pine trees. The cost of physically knocking down the standing-dead trees can be expensive and excessive (table 2) considering that the cost is compounded annually for at least 12 to 18 years before a return from the first thinning is attained.

Table 2—Average pine establishment costs for the site preparation study of standing-dead pine trees, Cumberland County, TN

Cost category	Practice	Cost per acre
		dollars
Costs incurred for all site-preparation treatments	Initial herbicide application (Imazapyr and Metsulfuron-methyl)	100
	Pine release herbicide application (Imazapyr)	60
	Planting labor	35
	Seedling cost	20
Costs of mechanical-site-preparation treatments	Standing-dead (control)	No cost
	Drum chop	100
	Drum chop, rake, and disk	250

Table 3—Survival, total height, and total diameter means of pine trees by site-preparation treatment after six growing seasons for the site-preparation study of standing-dead pine trees, Cumberland County, TN

Treatment	Survival[a]	Total height	Total diameter
	percent	*feet*	*inches*
Standing-dead (control)	78 a	16 a	2.3 a
Drum chop only	89 b	18 a	2.8 a
Drum chop, rake, and disk	86 b	18 a	3.1 a

[a] Treatment means with different letters within a column are significantly different at $P = 0.05$.

Results from this study indicate that diameter and height of loblolly pine were not affected by the site preparation treatments (table 3). The planted pines from a single nursery and on fairly uniform plateau sites grew similarly regardless of treatment. This outcome is in contrast to other research (Fox and others 1989, Morris and others 1983) where the topsoil and nutrients on the site were unevenly displaced by raking, piling, and burning windrows creating "waves" of different site productivities. The sites in this study were not windrowed and burned, but the residual, standing-dead material was raked, disked, and incorporated into the soil. The drum chop-only treatment left all residual material on the ground surface rather than incorporating the residual material into the soil. These differences in site preparation techniques as well as the differences between plateau, Piedmont, and Coastal Plain sites may contribute to the different results found in this study and the literature.

Survival of loblolly pine in the standing-dead control treatment was significantly lower (78 percent) compared to the two more intensive mechanical-site-preparation treatments (table 3). Poorer survival could be attributed to several factors. First, planting conditions were difficult. The control stands were within standing-dead pine trees (dead for 12 to 24 months), chemically treated hardwood midstory, and a dense understory of herbaceous vegetation with many briers that resulted from increased light penetration when the pine overstory died. These conditions may have affected the quality of the planting. The decaying standing-dead trees were safety hazards to planting crews. Second, crown debris and decaying dead stems often fell on new seedlings affecting their growth and survival. Planting among standing trees caused the planting rows to be more irregular and space between planted seedlings to be more variable than on the site-prepared stands. Third, the aerial herbicide application to control hardwoods and herbaceous vegetation in the control treatment prior to planting was probably not totally effective. The standing-dead pine overstory and the living hardwood midstory intercepted some of the herbicide such that the herbicide did not impact the ground vegetation as much as when the standing stems were removed.

A detailed, quantitative cost analysis was not performed in this study. The survival, diameter, and height results by treatment made the economic analysis rather intuitive. Removal of the dead-standing pine and the live hardwood midstory had little effect on the diameter and height growth of planted loblolly pine seedlings after six growing seasons. Thus, the added expense of removing these trees ($100 per acre for drum chopping only and $250 per acre for drum chopping, raking, and disking—table 2) is questionable. Seedling survival was less in stands that were not mechanically site prepared. However, at 78-percent average survival with more than 500 stems per acre after 6 years, the standing-dead, control stand has more than sufficient stocking for future management. The added expense of treating the overstory and midstory through mechanical-site preparation and the compounded interest before future revenues are received may not justify the expense. A fallacy of this study is that the cost of planting the control area was the same as planting the site-prepared area. The planters were paid per seedling regardless of ease of planting. The control, standing-dead stands made planting operations much more difficult and probably were more expensive and time consuming to plant.

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LITERATURE CITED

Clatterbuck, W.K.; Smalley, G.W.; Turner, J.A.; Travis, A. 2006. Natural history and land use history of Cumberland Plateau forest in Tennessee. Spec. Rep. 0601. Research Triangle Park, NC: National Council for Air and Stream Improvement, Inc. 37 p.

Fox, T.R.; Jokela, E.J.; Allen, H.E. 2007. The development of pine plantation silviculture in the Southern United States. Journal of Forestry. 105(7): 337–346.

Fox, T.R.; Morris, L.A.; Maimone, R.A. 1989. The impact of windrowing on the productivity of a rotation age loblolly pine plantation. In: Miller, J.H., ed. Proceedings of the fifth Biennial Silvicultural Research Conference. Gen. Tech. Rep. SO-74. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 133–140.

Haines, L.H.; Maki, T.H.; Sandeford, S.G. 1975. The effect of mechanical site preparation treatments on site productivity and tree (*Pinus taeda* L. and *P. elliottii* Engelm. var. *elliottii*) growth. In: Bernier, B.; Winget, C.H., eds. Forest soils and forest land management. Quebec, Canada: Les Presses de L'Universite Laval: 378–396.

McGowan, C. 2006. Soil survey of Cumberland County, Tennessee. Washington, DC: U.S. Department of Agriculture, Natural Resources Conservation Service. 229 p. + maps. http://soildatamart.nrcs.usda.gov/manuscripts/TN035/0/TNCumberland6_06Web.pdf. [Date accessed: September 2008].

Minogue, P.J.; Cantrell, R.L.; Griswold, H.C. 1991. Vegetation management after plantation establishment. In: Duryea, M.L.; Dougherty, P.M., eds. Forest regeneration manual. Dordrecht, The Netherlands: Kluwer Academic Publishers: 335–358.

Morris, L.A.; Pritchett, W.L.; Swindel, B.F. 1983. Displacement of nutrients into windrows during site preparation of a flatwoods forest soil. Soil Science Society of America Journal. 47: 951–954.

Neary, D.G.; Rockwood, D.L.; Comerford, N.B. [and others]. 1990. Importance of weed control, fertilization, irrigation, and genetics in slash and loblolly pine early growth on poorly drained Spodosols. Forest Ecology and Management. 30: 271–281.

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.

Smidt, M.; Dubois, M.R.; da Silveira Folegatti, B. 2005. Costs and cost trends for forestry practices in the South. Forest Landowner. 64(2): 25–31.

FIRST-YEAR SURVIVAL AND GROWTH OF FERTILIZED SLASH PINE IN SOUTH ALABAMA

Rebecca Barlow, Luben Dimov, Kris Connor, and Mark Smith[1]

Abstract—Early survival and growth rates are critical to the successful establishment of pine stands. Landowners need options to improve first-year growth on pine stands that will help them meet their land management objectives. One way to improve early stand survival and growth is through fertilization. In January 2008, approximately 5 acres of slash pine (*Pinus elliottii* Englem.) were planted on an old field site in south Alabama. Slash pine seedlings were treated with Accele-Grow-M™, a patent pending fertilizer supplement, to determine if there were any growth differences as a result of foliar application, root dip, or a combination of both and compared to a control group. Comparisons between initial seedling height and root-collar diameter measurements that were taken shortly after planting and first-year growth measurements showed that the control seedlings had increased growth in height and groundline diameter when compared to treated seedlings. In addition to the early effect on growth, it is possible that the fertilized seedlings also invested more in increased foliar density and root mass, parameters that were not measured. If this is the case, we expect to see acceleration in the growth rates of the fertilized seedlings in subsequent years.

INTRODUCTION

Landowners have keen interest in the options available to them that could improve survival and growth on their pine plantations and help them meet their management objectives. One way to improve early pine stand survival and growth is through fertilization (Jokela 2004, Jokela and Stearns-Smith 1993, Jokela and others 1991). Typically, pine stand fertilization recommendations are postplanting, between ages 5 and 10 at the time of canopy closure, and post thinning (Jokela and Stearns-Smith 1993, Jokela and others 1991). Postplanting fertilization is usually used to reduce the time until pines reach sawlog size and to increase pulpwood production. However, in some cases fertilization may actually reduce early slash pine (*Pinus elliottii* Englem.) growth on poorly drained soils (Haywood 1983).

In 1952, the Alabama Forestry Commission established E.A. Hauss Nursery. Located in Escambia County, this 400-acre tree seedling nursery grew an average of 37 million seedlings each year, producing southern pine and hardwood seedlings that were sold to the public. Seedling production ceased in 2006 and the mission of the nursery was realigned the following year. During the winter of 2008, the Alabama Forestry Commission renamed the former nursery the E.A. Hauss Demonstration Forest to reflect growing interest and demand for forestry research and demonstration in Alabama.

As part of the goals for the E.A. Hauss Demonstration Forest, demonstration areas were established that highlight different management techniques landowners could employ to improve returns from their small-scale forest operations. Of these forest research and demonstration plots, 5 acres were planted as a slash pine fertilization demonstration area on the E.A. Hauss Demonstration Forest where slash pine seedlings were treated with a liquid fertilizer supplement, Accele-Grow-M™, prior to planting. Accele-Grow-M™ has been used to enhance growth in agricultural crops such as soybeans and corn. However, limited testing on timber stands has been completed to date.

METHODS

In January 2008, Accele-Grow-M™ treated slash pines were planted on approximately 5 acres at the E.A. Hauss Demonstration Forest to determine differences in growth and survival relative to fertilizer application method. Soils on the site consist predominantly of Greenville fine sandy loam with 0.0 to 2 percent slopes. Since the area has a history of heavy cultivation, the site was subsoiled on 12-foot centers during the fall of 2007. Slash pine seedlings received one of four Accele-Grow-M™ treatments (foliar, root, foliar and root application, and a no-treatment control) prior to planting. Two rows (treatment strip) of trees from each treatment were then hand planted on a 6- by 12-foot spacing. Treatment strips were systematically alternated, maintaining the 12-foot spacing between rows, throughout the remainder of the 5-acre block; resulting in two full replicates.

Initial seedling height and root-collar diameter measurements were taken after planting in February 2008. During the first growing season, the stand was released using a rate of 5-ounce Arsenal per acre to control morning glory (*Ipomoea violacea*). First-year growth measurements of height, root-collar diameter, and survival were taken during October of that same year.

RESULTS

First-year measurements indicate that there is indeed some growth response of Accele-Grow-M™ treated seedlings relative to untreated seedlings on the demonstration forest site. Table 1 illustrates first-year growth comparison of groundline diameter measurements in mm. Growth of foliar-treated trees on the site ranged from an average of 1.7 mm more than the untreated control trees to 3.1 mm more than the trees that were root treated. Of the fertilized trees, foliar-only treatments performed much better than those that had fertilized applied only to the roots or to both roots and foliage with an average groundline diameter growth of 22.4 mm (table 1).

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Table 1—First-year growth comparison of groundline diameter for Accele-Grow-M™ treated slash pine trees on the E.A. Hauss Demonstration Forest

Groundline diameter growth	Root and foliar	Root only	Foliar only	Control
	mm			
Average	11.7	9.6	12.7	11.0
Maximum	21.6	15.7	22.4	18.8
Minimum	1.2	1.3	4.3	3.9

Results were similar when comparing first-year average height growth (table 2). Foliar-treated trees averaged 26.1 cm of growth compared to 23.2 cm for the control trees and 24.2 cm for the root-and-foliar-treated trees. Root-only treatments averaged 20.5 cm of height growth. As with diameter growth, foliar treatments had more height growth than other treatments (table 2). In some cases root-and-foliar and root-only treatments lost height growth as there was Nantucket pine tip moth [*Rhyacionia frustrana* (Comstock)] damage on many trees.

There were little differences in survival among treatment types (table 3). Foliar-only and root-and-foliar treatments had slightly higher survival rates (97.8 and 97.4 percent, respectively) than the control and root-only treatments.

CONCLUSIONS

First-year measurement results indicate that there is some benefit to diameter and height growth from treating slash pine seedlings prior to planting with Accele-Grow-M™ fertilizer when comparing height and groundline diameter growth. However, there are several factors that need further investigation to determine long-term growth effects. Follow-up measurements and research installations are suggested to further investigate Accele-Grow-M™ effects on southern pine growth.

Intermediate treatments are planned to determine how tree growth, form, and cone and seed production is impacted when trees are treated over the course of a rotation. This would include examination of aboveground biomass production. In addition, work is ongoing to determine if there are any differences in root development and growth by treatment type.

Further study is also needed to determine if there is increased damage by tip moth to young Accele-Grow-M™ fertilized slash pines planted in that region. Results of prior research that examines tip moth infestations on young fertilized southern pines has been mixed (Ross and others 2005). For this current study, incidence of tip moth by treatment type was not measured. However, decreases in tree height were noted for some treatments, and would suggest that future studies should measure this specifically.

Finally, there is interest in how other southern pines, such as loblolly (*P. taeda* L.), will respond to treatment.

Today the E.A. Hauss Demonstration Forest offers a valuable opportunity for forestry research and demonstration among partners from the Alabama Forestry Commission, Alabama A&M University, Auburn University, and the U.S. Forest Service. Projects such as this current study will provide real world demonstrations for landowners and land managers who are increasingly seeking information to help them better identify and meet their land management goals.

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Table 2—First-year height growth comparison for Accele-Grow-M™ treated slash pine trees on the E.A. Hauss Demonstration Forest

Total height growth	Root and foliar	Root only	Foliar only	Control
	cm			
Average	24.2	20.5	26.1	23.2
Maximum	42.5	41.0	42.5	56.0
Minimum	–4.0	–6.0	7.5	4.0

Table 3—First-year survival comparison of Accele-Grow-M™ treated slash pine trees on the E.A. Hauss Demonstration Forest

Survival comparison	Root and foliar	Root only	Foliar only	Control
	---------- *percent* ----------			
First growing season	97.4	96.0	97.8	96.3

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LITERATURE CITED

Haywood, J.D. 1983. Small topographic differences affect slash pine response to site preparation and fertilization. Southern Journal of Applied Forestry. 7: 145–148.

Jokela, E.J. 2004. Nutrient management of southern pines. In: Dickens, E.D.; Barnett, J.P.; Hubbard, W.G.; Jokela, E.J., eds. Slash pine: still growing and growing. Proceedings of the slash pine symposium. Gen. Tech. Rep. SRS-076. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 33–41.

Jokela, E.J.; Allen, H.L.; McFee, W.W. 1991. Fertilization of southern pines at establishment. In: Duryea, M.L.; Dougherty, P., eds. Forest regeneration manual. Dordrecht, the Netherlands: Kluwer Academic Publishers: 263–277.

Jokela, E.J.; Stearns-Smith, S.C. 1993. Fertilization of established southern pine stands: effects of single and split nitrogen treatments. Southern Journal of Applied Forestry. 17: 135–138.

Ross, W.G.; Kulhavy, D.L.; Sun, J.H. 2005. Effects of fertilization and herbicides on growth of young loblolly pine and infestations of Nantucket pine tip moth (Lepidoptera: Tortricidae). Insect Science. 12: 367–374.

PLANT COMMUNITY RESPONSES TO SOIL DISTURBANCE AND HERBICIDE TREATMENTS OVER 10 YEARS ON THE TEXAS LTSP STUDY

D. Andrew Scott and Richard H. Stagg[1]

Abstract—Determining how anthropogenic disturbances affect site productivity through bioassays requires a complete understanding of both overstory and understory vegetation. This study was installed in 1997 to determine how soil compaction and intensive harvesting affected the inherent site productivity of pine stands on the western boundary of loblolly pine's (*Pinus taeda* L.) natural range. We measured the plant communities at ages 5 and 10 on plots receiving a factorial combination of three levels of soil compaction and three levels of organic matter removal at harvest. Soil compaction had little impact on plant communities. Intensive harvesting, especially when the forest floor was removed, greatly reduced pine survival and growth and somewhat reduced woody understory growth, while increasing herbaceous understory growth. The reduction in woody understory biomass did not improve pine growth because forest floor removal reduced soil fertility and water content, which affected the pines in addition to the understory vegetation.

INTRODUCTION

The North American Long-Term Soil Productivity (LTSP) study was created to determine how forest management affected the long-term productivity of a site (Powers and others 1990). The most consistent finding across the 62 locations of the core LTSP study and across the many affiliate installations has been the large and sustained positive impact of noncrop vegetation control on forest productivity (Powers and others 2004). Seldom if ever has the total aboveground biomass of the nontreated plots been equal or greater than the plots receiving competition control. Although soil compaction has reduced tree growth on some heavy textured soils, it has also led to increased growth on several sites. The increased growth on some sites has been attributed to increased water holding capacity (Gomez and others 2002), but also to the indirect control of noncrop vegetation (Stagg and others 2006). Intensive organic matter removals have not consistently reduced productivity but they have reduced productivity on inherently low nutrient soils through induced nutrient deficiencies (Scott and others 2007). A final lesson that has emerged from synthesizing the overall LTSP project has been that various site-specific anomalies in site type, treatment installation, or post-treatment events have at times overshadowed the main treatments in their effect on measured productivity.

An excellent example of this effect was observed on the Mississippi plots at age 10, which are dominated in the understory by inkberry (*Ilex glabra* L.), a very persistent and dense-growing woody species that is difficult to control even with herbicides. On these sites, planted loblolly pine (*Pinus taeda* L.) productivity was 21 percent greater on the plots that had been compacted compared to uncompacted plots, apparently due to reduced understory biomass on those plots (Stagg and Scott 2006). The compaction effect was apparent whether herbicides were applied or not; chemical vegetation control was not 100 percent effective on understory biomass and was greater on compacted plots compared to the noncompacted plots. Since the soil was loamy and had a moderate initial bulk density, it was not likely that the compaction increased pine growth through increased soil water holding capacity as found on the loose sands in California (Gomez and others 2002).

The Texas LTSP sites provide a unique environment to test how differences in site and soil conditions may alter the relationships between soil compaction, organic matter removal, chemical vegetation control, and planted tree productivity. Accordingly, the objectives of this study were to determine how organic matter removal, soil compaction, and chemical vegetation control affected understory plant abundance and growth over the first 10 years since plantation establishment, and how any differences in understory dynamics affect planted loblolly pine dynamics.

MATERIALS AND METHODS

Study Sites

The study site is located in the Davy Crockett National Forest in Trinity County, TX. The soil is a Kurth series, which is in the fine-loamy, siliceous, semiactive, thermic family of oxyaquic glossudalfs. A relatively thick (0.5-m) sandy loam surface caps a deep clay loam subsoil. This series is moderately well drained, forming in loamy clay sediments in uplands (Steptoe 2003). The preharvest stand was a well-stocked, naturally regenerated, even-aged loblolly and shortleaf (*P. echinata* Mill.) pine stand with a few scattered longleaf pines (*P. palustris* Mill.) and associated upland hardwoods, such as oaks (*Quercus* spp.) and sweetgum (*Liquidambar styraciflua* L.). The site had received prescribed burns at 3- to 5-year intervals during the previous rotation, although the site had not been burned for 10 years prior to the establishment of the LTSP study.

Twenty-seven 0.2-ha study plots were established in a split-plot, randomized complete block design (RCBD). The general plot layout is a 3×3 factorial arrangement of organic matter removal and soil compaction. Following harvest, main plot treatment applications and planting, each plot was split in half. On half of each main plot, the understory vegetation community was allowed to recover naturally. On the other half,

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the understory was intended to be controlled chemically to create a virtual planted pine monoculture.

Treatments

The compaction treatments of the LTSP study consisted of no compaction (C0), moderate (C1), and severe (C2) compaction. No equipment traffic was allowed during harvest on the C0 plots; only foot traffic was allowed. The trees were removed by chainsaw felling and lifting the logs off of each plot with a log loader or crane. The C1 and C2 compaction plots were logged by normal logging methods, which consisted of chassis-mounted shears and grapple skidders, although no repetitive trafficking was allowed on these plot treatments. Prior to harvesting, the compaction treatments were applied by towing a pneumatic-tire roadbed compactor with a crawler tractor over the designated plots. The roadbed compactor had a rolling width of 1.52 m. The ballast in the roadbed compactor was adjusted to meet the requirements of the treatment. Field trial tests determined that a load of 3.0 mt was required to initiate soil compaction in the Kurth soil series. The severe load was set at 6.4 mt and the moderate load was set as a logarithmic average between these two points or 3.6 mt. To ensure complete coverage and uniform compaction, each plot received three passes in one direction and then three more passes in a perpendicular direction for a total of six passes.

The organic matter removal treatments consisted of a bole-only harvest (BOH), a whole-tree (aboveground) harvest (WTH), and a whole-tree harvest followed by the removal of the forest floor (WTFF). Following the compaction and organic matter removal treatments the study plots were hand planted with containerized loblolly pine seedlings from 10 known families on a 2.5- by 2.5-m spacing. Four annual applications of glyphosate (3.9 L/ha each) were applied to the pine-only half of each main treatment plot beginning in the middle of the fourth growing season.

Measurements

Prior to harvesting, the understory biomass was destructively sampled within five randomly placed 1-m^2 square sampling areas and separated by lifeform (woody or herbaceous). The understory was collected, oven dried, and weighed. All understory species <7.62 cm in diameter at breast height (d.b.h.) were collected, but the biomass was not separated by species. Percent cover by species was determined visually along six 30.5-m transects per plot.

At age 5, understory vegetation biomass (woody and herbaceous) was collected from four randomly placed 1.56-m^2 sample areas on each subplot, but was separated only by lifeform, not species.

A more intense understory vegetation biomass measurement was made at age 10. All woody understory species <1.37 m tall were clipped, bagged, and tallied by species and number of stems in each of three 6.25-m^2 sampling areas on each subplot. All woody understory species >1.37 m tall were tallied by species, height, and number of stems per rootstock within three 56-m^2 sampling areas randomly located within each study plot. Plot level biomass was determined from these measurements using biometric equations developed from the LTSP installations and other nearby forests (Scott and others 2006).

Data Analysis

The study was analyzed as a split-plot RCBD with three replicate blocks using analysis of variance (ANOVA) (repeated measures for understory biomass) and an alpha of 0.1. Interaction terms were statistically insignificant unless otherwise stated. When interaction terms were significant, the least-squares means were sliced by each effect in the interaction. When the ANOVA showed a significant model, the means were separated with the Ryan-Einot-Gabriel-Welsch multiple range test (SAS Institute 1999).

RESULTS AND DISCUSSION

Initial planted pine survival was very good following the first growing season after planting, averaging 93 percent across all plots. Survival was not affected by compaction or intensive organic matter removal in the first growing season (table 1). About 40 percent of the planted trees died in the second growing season, and survival averaged 51 percent across all plots. Few trees died following the second growing season, and overall survival through age 10 was 46 percent (data not shown). Neither compaction nor WTH had an impact on survival compared to uncompacted or BOH, but removing the forest floor reduced survival to only 34 percent. In fact, two WTFF plots had such poor initial survival (<20 percent), they were replanted and failed again and were not included in the calculations.

The poor survival of planted pines in the second growing season appeared to be due largely to rainfall patterns. Rainfall was normal following planting until autumn of the first growing season (data not shown). From November through January, the site received almost twice the normal rainfall, creating overly wet conditions for good root growth. This was followed by extremely dry conditions in the early part of the second growing season, and wet conditions in the latter part of the second growing season. These periods of wetness and dryness likely inhibited early root growth through increased soil strength when the soil was dry and low aeration when the soil was wet. While the original hypothesis of the LTSP study proposed that compaction would exacerbate these types of effects through increasing soil strength and reducing aeration (Greacen and Sands 1980), compaction had no impact on survival at the Texas sites. Some sandy sites have benefited from compaction by increasing soil water holding capacity, but the Texas sites showed no improvement in survival or growth that could be attributed to improved water-holding capacity. It's possible that while water-holding capacity may have been improved, which would have improved root growth and survival during the droughty months, the corresponding decrease in macroporosity would have hindered root growth and survival during the wet months.

Woody understory biomass was unaffected by compaction at age 10 (fig. 1), even though the C1 plots had only about one-third the biomass at age 5 and grew from just over 2 mt/ha to about 8 mt/ha. Herbaceous biomass was unaffected by compaction

Table 1—Planted loblolly pine initial survival and 10-year mean height and d.b.h. response to three levels of compaction and harvest intensity and two levels of herbicide application on a sandy soil in east Texas

	Levels of compaction			Harvest intensity			Herbicide application	
	C0	C1	C2	BOH	WTH	WTFF	H0	H1
Survival (%)								
Year 1	95 a	92 a	92 a	94 a	92 a	91 a	93[a]	N/A
Year 2	50 a	57 a	46 a	64 a	56 a	34 b	51	N/A
Height (m)								
Age 5	3.8 a	3.8 a	3.8 a	4.1 a	3.8 b	3.5 c	3.8 a	3.8 a
Age 10	9.5 a	9.6 a	9.7 a	10.0 a[b]	9.7 a	8.9 b	9.5 a	9.7 a
D.b.h. (cm)								
Age 5	5.0 a	5.0 a	5.1 a	5.5 a	5.0 b	4.4 c	4.8 b	5.2 a
Age 10	12.9 b	13.0 b	13.5 a	13.8 a	13.2 b	12.2 c	12.8 b	13.5 a

N/A = not applicable; C0 = no compaction; C1 = moderate compaction; C2 = severe compaction; BOH = bole-only harvest; WTH = whole-tree harvest; WTFF = whole-tree and forest floor harvest; H0 = hand-felled, bole-only harvest; H1 = high-production, whole-tree harvest.

[a] Statistics were not performed for survival differences between herbicide treatments because those treatments had not yet been implemented.

[b] The compaction by organic matter removal by herbicide interaction term was significant, but was caused by one specific treatment combination and not in any discernible pattern.

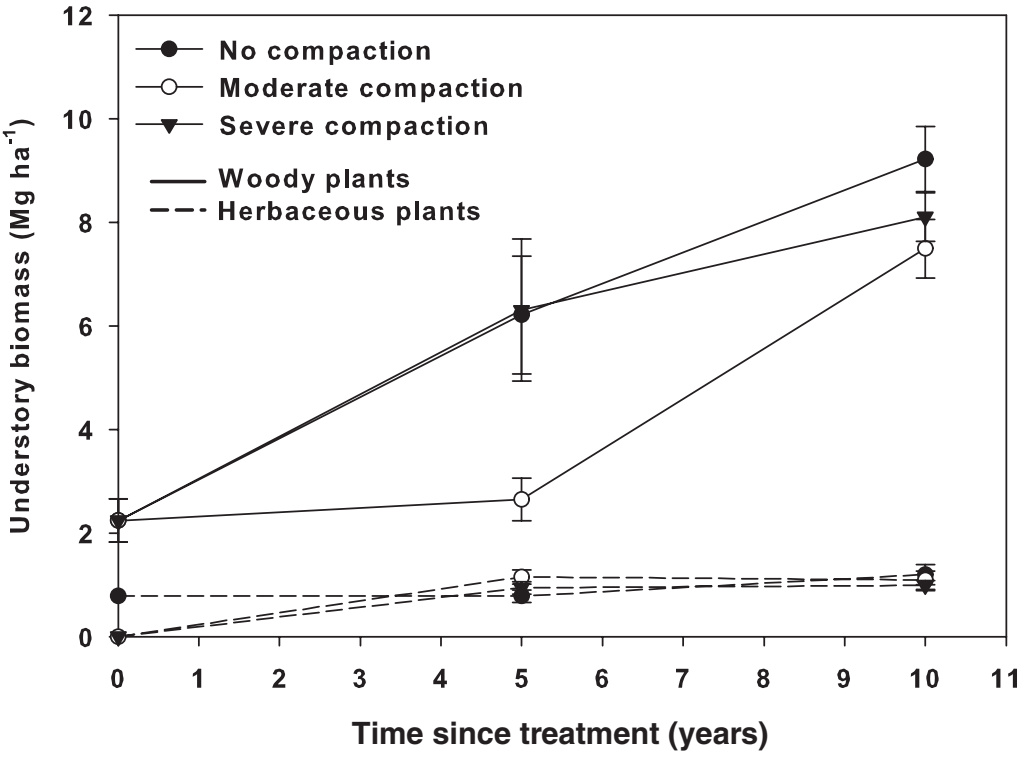

Figure 1—Woody and herbaceous understory biomass response to three levels of experimentally applied soil compaction in loblolly pine plantations. Time 0 years refers to preharvest conditions.

at either age, and averaged about 1 mt/ha from age 5 through age 10. These results are in contrast to the results found on the Mississippi LTSP sites, where soil compaction reduced overall understory biomass by 66 percent at age 5 years, and at age 10 years the understory biomass on the C1 plots was the same as on the C0 plots, but remained 40 percent less on the C2 plots (Stagg and Scott 2006). The compaction treatments on the Texas sites were effective in increasing soil bulk density, but not to exceptionally high levels; both the C1 and C2 treatments increased bulk density of the surface 10 cm, from 1.18 to 1.33 mt/m^3 (Scott and others 2004). Thus, the change in bulk density and concomitant increase in soil strength and decrease in macroporosity may have been too slight to affect root growth and water relations.

Similarly, compaction had little impact on planted pine growth at age 5 or 10, even though the trees on the C2 plots were about 0.5 cm greater in d.b.h. than the trees on the C0 and C1 plots at age 10 (table 1). In Mississippi, pine growth increased in proportion with the decrease in understory biomass (Stagg and Scott 2006). Stagg and Scott (2006) did not find evidence of soil improvement caused by compaction; the compaction apparently acted as a vegetation control treatment. Since understory biomass was not affected by compaction in Texas at age 10, it was not surprising that tree growth was also largely unaffected. The slight increase in d.b.h. on the C2 plots may have been due to differences in understory biomass; woody understory biomass was lower, although not statistically significant, on the C2 plots (fig. 1). It is unclear why the understory was reduced by compaction in Mississippi but not in Texas at age 10. In general, the understory composition was quite similar between the two sites except for the dominant understory species. The Texas sites were dominated by yaupon (*I. vomitoria* Aiton), while the Mississippi sites were dominated by inkberry. Although both yaupon and inkberry are hollies, it is possible that inkberry is more susceptible to reduced sprouting and growth caused by direct crushing than yaupon.

Both woody and herbaceous understory biomass were also largely unaffected by harvesting treatment. Although the understory woody biomass on the WTFF plots was about 37 percent lower than the biomass on the BOH plots (fig. 2), this difference was not significant. In contrast, the herbaceous biomass declined slightly from age 5 to 10 on the BOH and WTH plots but doubled on the WTFF plots due to the higher light availability on these plots that had poor pine survival and poor woody biomass colonization. Several areas on these plots had open areas up to about 0.08 ha in size with little if any woody cover of any sort. Forest floor removal may have removed some of the seed source for many of the woody species, but it is unclear why, after 10 years, these areas have not been colonized by woody plants.

Planted pine height was about 1 m less on the WTFF plots compared to the BOH or WTH plots at age 10, and d.b.h. was 0.6 and 1.6 cm less on the WTH and WTFF plots, respectively, than on the BOH plots at age 10, and the relative differences among treatments were similar at age 5 (table 1). This difference in pine response to competition between the Texas and Mississippi sites was probably due to the mechanism by which the treatments controlled the vegetation. In Mississippi,

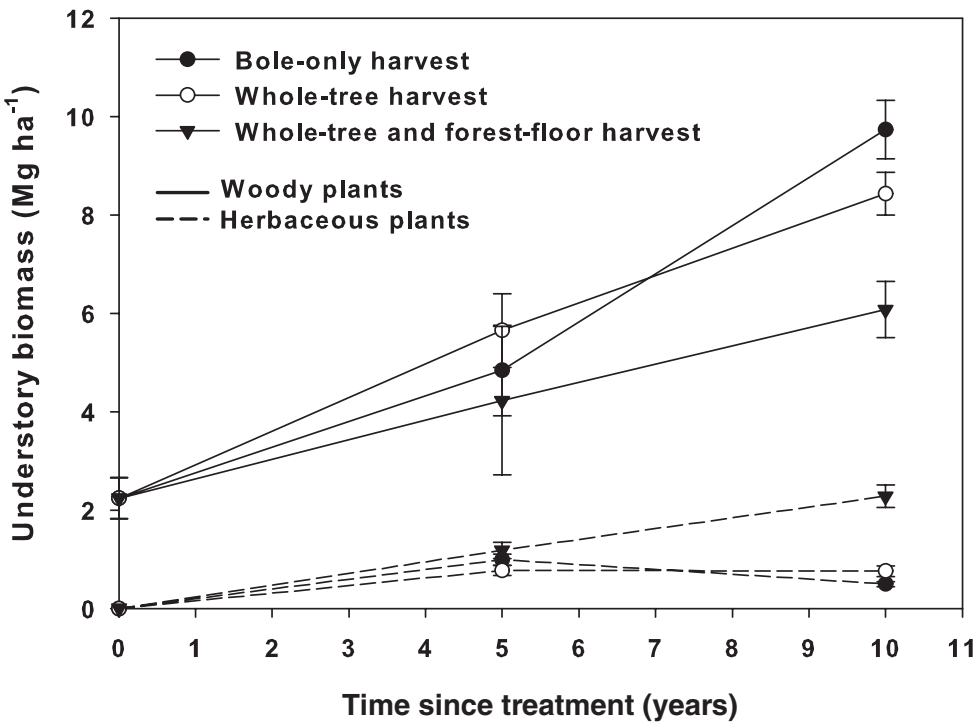

Figure 2—Woody and herbaceous understory biomass response to three levels of harvest intensity. Time 0 years refers to preharvest conditions.

the control mechanism was likely the direct impact of compaction against the vegetation. In Texas, the WTH and forest floor removal had much less of a direct impact against the understory vegetation, but instead probably affected its growth through organic matter and nutrient removal.

Previous research has shown that across seven soil types from Mississippi to Texas, site productivity loss caused by intensive harvesting (WTH or WTFF) is directly related to soil phosphorus content (Scott and others 2007). The Kurth soils in Texas had the lowest soil P content of the seven soil series studied. Thus, it is likely that the additional removal of nutrients, especially P, by the WTH and forest floor removal reduced pine growth. In addition, following the second growing season the sites experienced a 4-year drought that may have reduced the ability of the woody plants to expand or invade into the open areas. Grasses and other herbaceous plants were able to competitively invade these open areas. The removal of the forest floor may also have exacerbated the drought stress on these plots by removing the mulch effect normally provided by an intact forest floor which again would likely favor herbaceous plants over woody species.

Herbicides had much less of an effect on either understory biomass or on pine tree growth at the Texas sites than expected or in comparison to similar studies. By age 5, the herbicide treatments had been applied twice; once during the fourth and once during the fifth growing seasons. Woody understory biomass was reduced from 6.4 to 3.6 mt/ha by the herbicide applications, while herbaceous biomass was reduced from 1.3 to 0.6 mt/ha (fig. 3) at age 5. After age 5, two more applications were made; one each in the sixth and seventh growing seasons. At age 10, woody understory biomass was essentially unaffected by the additional herbicide treatments. The difference between the treated and untreated subplots at age 10 (2.7 mt/ha) was the same as at age 5 (2.8 mt/ha). The herbaceous understory was no longer different between the treatments at age 10. Planted pines were significantly but only slightly taller and larger in d.b.h. on the plots receiving herbicide treatment compared to untreated plots at age 5 or 10 years (table 1). Although the herbicides reduced woody understory biomass, the early drought and poor soil fertility conditions hindered the ability of the pines to take advantage of the reduction in competition.

Woody species composition changed dramatically from preharvest to age 10. Prior to harvest, American beautyberry (*Callicarpa americana* L.) was the most dominant species in the understory, followed by yaupon and sweetgum. Various oaks, mostly southern red oak (*Q. falcata* Michx.), cherrybark oak (*Q. pagoda* Raf.), water oak (*Q. nigra* L.), and white oak (*Q. alba* L.) and wax myrtle [*Morella cerifera* (L.) Small] composed the other common species. Neither compaction nor any of the organic matter removal treatments affected the composition of the major woody species (data not shown). The herbicide applications reduced the biomass and relative dominance of American beautyberry, sweetgum, oaks, and wax myrtle, but more than doubled the relative dominance of yaupon (fig. 4). Apparently, because of their relatively low susceptibility to glyphosate, the herbicide treatments

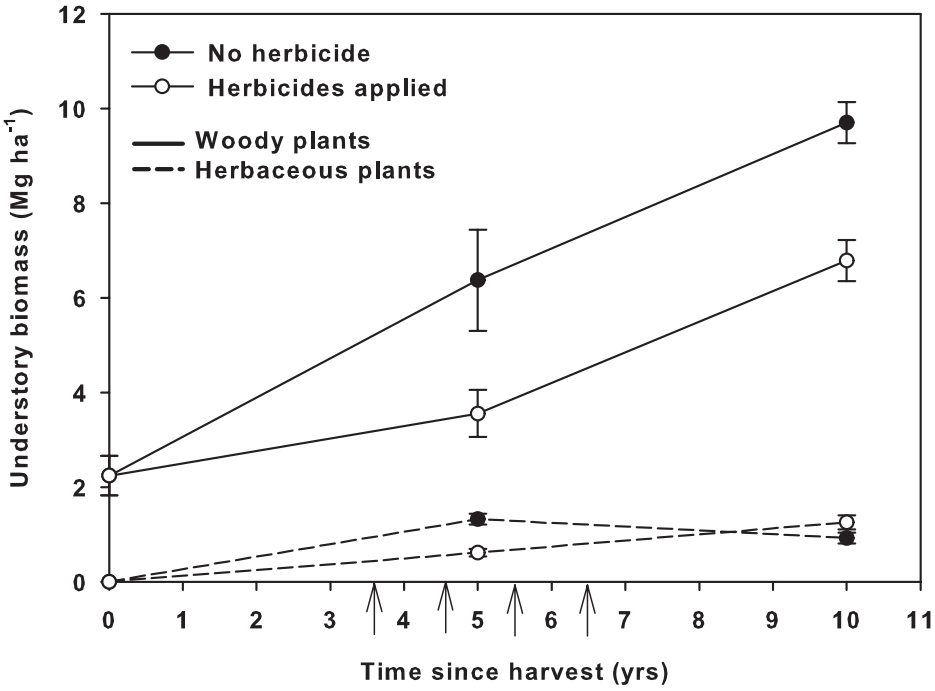

Figure 3—Woody and herbaceous understory biomass response to four applications of herbicide (glyphosate) in loblolly pine plantations. Arrows indicate approximate dates of application. Time 0 years refers to preharvest conditions.

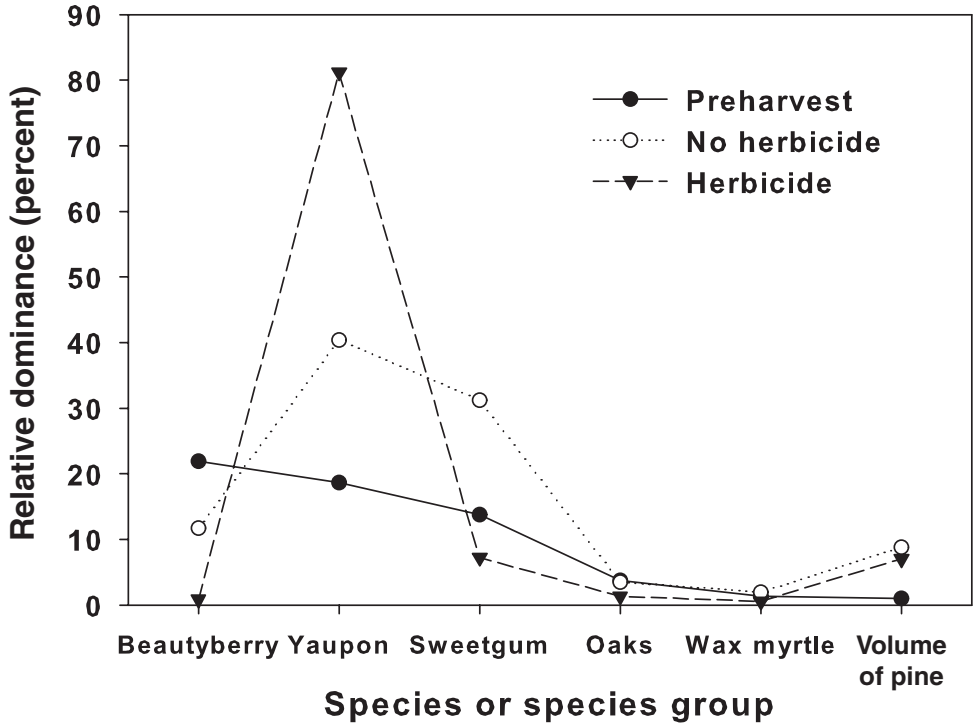

Figure 4—Relative dominance of the six most dominant woody species or species groups across the Texas LTSP sites before treatments and at age 10 years following two vegetation control treatments (no treatment and herbicide).

essentially provided a chemical release to both loblolly pine and yaupon.

CONCLUSIONS

This study found that in contrast to other LTSP sites, compaction had little impact on the understory or overstory vegetation, while both organic matter removal and herbicide treatments did affect the vegetation. Intensive organic matter removal reduced the growth of planted pine even where woody understory biomass was somewhat reduced by the treatments. The poor soil moisture conditions that developed due to both very low and very high rainfall during the second growing season likely caused the poor pine survival, but treatments such as whole-tree and forest floor removal that further reduced soil fertility on these infertile sites caused further declines in pine tree growth. These results support caution when harvesting to leave the forest floor and slash scattered on site on soils with limited nutrients.

ACKNOWLEDGMENTS

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LITERATURE CITED

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

Greacen, E.L.; Sands, R. 1980. Compaction of forest soils. A review. Australian Journal of Soil Research. 18: 163–189.

Powers, R.F.; Alban, D.H.; Miller, R.E. [and others]. 1990. Sustaining site productivity in North American forests: problems and prospects. In: Gessel, S.P.; Lacate, D.S.; Weetman, G.F.; Powers, R.F., eds. Sustained productivity of forest soils. Proceedings of the seventh North American forest soils conference. Vancouver, B.C., Canada: University of British Columbia: 49–79.

Powers, R.F.; Sanchez, F.G.; Scott, D.A.; Page-Dumroese, D. 2004. The North American Long-Term Soil Productivity experiment: coast-to-coast findings from the first decade. In: Sheppard, W.D.; Eskew, L.G., eds. Silviculture in special places: Proceedings of the national silviculture workshop. Gen. Tech. Rep. RMRS-34. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 191–206.

SAS Institute Inc. 1999. SAS/STAT user's guide. Version 8. Cary, NC: SAS Institute Inc.

Scott, D.A.; Novosad, J.; Goldsmith, G. 2007. Ten-year results from the North American Long-Term Soil Productivity study in the western Gulf Coastal Plain. In: Furniss, M.; Clifton, C.; Ronnenburg, K., eds. Advancing the fundamental sciences: Proceedings of the Forest Service national earth sciences conference. Gen. Tech. Rep. PNW-689. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station: 331–340.

Scott, D.A.; Stagg, R.H.; Smith, M.A., Jr. 2006. A non-destructive method for quantifying small-diameter woody biomass in southern pine forests. 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: 358.

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

Stagg, R.H.; Scott, D.A. 2006. Understory growth and composition resulting from soil disturbances on the Long-Term Soil Productivity study sites in Mississippi. 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: 52–56.

Steptoe, Levi J., Jr. 2003. Soil survey of Trinity County, Texas. [Place of publication unknown]: U.S. Department of Agriculture, Natural Resources Conservation Service. In cooperation with: Texas Agricultural Experiment Station and U.S. Department of Agriculture, Forest Service. 261 p.

IMPACTS OF FOUR DECADES OF STAND DENSITY MANAGEMENT TREATMENTS ON WOOD PROPERTIES OF LOBLOLLY PINE

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Abstract—Stand density management is a powerful silvicultural tool for manipulating stand volumes, but it has the potential to alter key wood properties. At a site in northcentral Louisiana, five density management regimes were conducted over a 45-year period. At age 49, a stratified sample of trees was destructively harvested for crown length, taper, and specific gravity determination. No differences among these tree and wood characteristics were found, suggesting that forest managers have great flexibility in density management options for such a site from a wood quality perspective.

INTRODUCTION

Manipulation of stand density is integral to the management of loblolly pine (*Pinus taeda* L.) plantations. Silvicultural tools for manipulating stand density consist of initial stocking density and thinning. Initial stocking density strongly influences the timing of crown closure, the rate at which stand-level yield accrues, mean tree size at rotation, and the timing and intensity of midrotation treatments such as thinning (Harms and others 2000, Sharma and others 2002, Zhang and others 1996). Thinning similarly affects tree- and stand-level growth rates and mean tree size at rotation (Brix 1982, Visser and Stampfer 2003, Zahner and Whitmore 1960). Due to these profound effects on tree- and stand-level growth and development, Clason (1994) asserted that biological, financial, and ecological performance of pine plantation management is determined by combinations of initial stocking densities and thinning schedules.

In addition to their influences on tree- and stand-level growth and yield, initial stocking density and thinning schedules can influence wood properties and the resulting quality of timber produced (Clark and others 1994). As such, stand density regulation is the most profound silvicultural practice for controlling wood formation and quality (Larson and others 2001). Low initial stocking densities tend to increase branch retention and the size of the live crown as suppression of branches is delayed, which can degrade log quality via the presence of larger and more frequent knots (Macdonald and Hubert 2002). Poor stem straightness has also been observed in response to relatively low initial stocking densities (Brazier 1986, Macdonald and Hubert 2002). The relatively fast stem diameter growth promoted by low initial densities can result in larger volumes of juvenile wood characterized by low specific gravity (Clark and others 2008, Larson and others 2001). Low specific gravity is of particular merit due to its close correlation with structural properties of dry wood and importance as a component of grading rules for southern pine lumber and timber (Larson and others 2001). Similar to low initial stockings, heavy and/or frequent early thinnings promote greater retention of lower branches, increase stem taper, and increase the proportion of juvenile wood within stems. However, the effect of thinning on wood properties is variable depending on initial planting densities, tree age, time of year, and site conditions (Larson and others 2001, Macdonald and Hubert 2002).

While the effects of initial stocking density and thinning on loblolly pine wood properties have been studied, rotation-length information on the effects of combinations of these practices on wood properties is lacking. Such study is vital for determining the quality of wood that can be generated by a wide range of rotation-length density management regimes. The objective of this study was to determine the effects of initial stocking density and thinning regimes on key wood properties of loblolly pine.

METHODS

In 1958, a loblolly pine plantation was planted with 1,200 trees per acre at the Louisiana State University AgCenter Hill Farm Research Station in northwest Louisiana (32°44' N, 93°03' W) on a gravelly, fine sandy loam Darley-Sacul soil (an association of a fine, kaolinitic, thermic Hapludult and a fine, mixed, active, thermic Aquic Hapludult). This well-drained soil type is common in upland forests of northwestern Louisiana, southwestern Arkansas, and eastern Texas (Kilpatrick and Henry 1989). Site index for loblolly pine on a 25-year basis for this site was 65 feet. Prior to planting, the area had reverted from an agricultural field to a 40-year-old mixed pine-hardwood stand. When the previous stand was harvested, merchantable trees were harvested and remaining vegetation was piled and burned 2 years prior to planting.

In 1962, the stand was precommercially thinned to approximate initial stocking densities. Five stocking density treatments, each replicated four times, were conducted: 1,000, 600, 300, 200, and 100 trees per acre. Treatments were arranged in a randomized complete block design, with slope as the blocking factor, and applied to 0.5-acre plots. At age 21, thinning regimes were superimposed on the initial stocking density treatments (table 1). Thinning regimes followed correlated curve trend study protocol (Craib 1947, O'Conner 1935), in which thinning occurred when annual d.b.h. growth of a thinning regime declined relative to that of a thinning regime with lower stand density. The goal of the

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Table 1—Stand density throughout a series of five density management regimes conducted in a loblolly pine plantation in northcentral Louisiana

	Density management regime				
Thinning age	1	2	3	4	5
year	*trees per acre*				
4	1,000	600	300	200	100
21	300	300	200	100	50
26	200	200	100	50	—
31	100	100	50	—	—
36	50	50	—	—	—
41	25	25	25	25	25

thinning regimes was to sequentially reduce initial stocking densities to 25 trees per acre via the correlated curve trend approach.

In July and September 2007, a destructive harvest of trees was conducted. Sampling was stratified by stem d.b.h. distributions in each plot, based on stem d.b.h. measurements collected in June 2007. In each plot, d.b.h. distributions were divided into three classes (upper, middle, lower), and one tree from each d.b.h. class was felled using a feller buncher. Upon felling, total stem length and length of the stem to the lowest live whorl of branches were measured, and crown length was calculated as the difference between these measurements. Inside-bark diameter was measured at the base of the stem and 16, 32, and 48 feet along the stem. Taper was calculated for each 16-foot length of stem from the inside-bark diameter measurements as the change in height per change in radius (Bohannan and others 1974) to provide taper for three logs of each tree. A disk of 1.5-inch thickness was cut from the stem at 4.5 feet for specific gravity determination. A 0.006-inch thick strip was cut through the center of each disk and dried at 50 °C. Specific gravity of each annual ring was determined at 0.0002-inch intervals using an x ray densitometer (Quintek Measurement Systems, Inc., Knoxville, TN) with a resolution of 0.00001 (Clark and others 2006). Specific gravity for each ring was weighted by its basal area. Weighted ring specific gravities of each tree were averaged for the 1962 through 1978 period to isolate the effects of the initial stocking densities, averaged for the 1978 through 2007 period to isolate the effects of thinning regimes, and averaged for the entire life of each tree to determine the effects of the density management regimes on specific gravity.

Analyses of all variables were conducted by analysis of variance (ANOVA) using the MIXED procedure of the SAS system (SAS Institute Inc., Cary, NC). When an ANOVA indicated significant ($P < 0.05$) treatment effects, treatment means were calculated and separated by the DIFF option of the LSMEANS procedure. The models for all analyses consisted of density regime as a fixed effect and block and the block × treatment interaction as random effects.

RESULTS AND DISCUSSION

Among all variables measured, only d.b.h. was significantly affected by the density management regimes (table 2). The lowest density (regime 5) had greater d.b.h. than all other treatments, and d.b.h. of regime 4 exceeded that of regimes 1, 2, and 3. Clason (1994) found that d.b.h. growth of regimes 4 and 5 exceeded that of the other regimes as early as age 7 at this site. These final d.b.h. measurements indicate that diameter growth differentials were very persistent in response to these regimes, so much so that d.b.h. differences among regimes remained even though all regimes were at the same density from 1999 through 2007. This indicates that at this site diameter growth of trees managed at relatively higher densities could not reach growth trajectories of stands managed at relatively low densities, even when thinned to comparable levels.

Larger crowns are associated with low-density management regimes (Larson and others 2001, Macdonald and Hubert 2002), and persistently larger crown areas would likely induce the long-term differences in d.b.h. seen among regimes at this site. However, although crown lengths of regimes 4 and 5 were 7 to 21 percent greater than those of the higher density regimes, they did not significantly differ among regimes (table 2). It is probable that crown damage sustained in an ice storm in 2001 mediated differences in crown area among treatments.

Taper was unaffected by the density management regimes (table 3). In accordance with the uniform stress theory, trees taper to equalize bending stress from wind drag acting on the crown; as such, there is a close linkage between crown dimensions and taper (Dean and Long 1986, Metzger 1893). The lack of differences in crown lengths among regimes in this study may have thus contributed to the lack of taper

Table 2—Crown length, d.b.h., weighted average specific gravity of the entire stem at d.b.h., weighted average specific gravity of wood formed prior to the first commercial thinning, weighted average specific gravity of wood formed during commercial thinnings, and taper in response to five density management regimes conducted in a loblolly pine plantation in northcentral Louisiana

Regime	Crown	D.b.h.	SG	SG-PRE	SG-THIN
	feet	*inches*			
1	37.4 a	19.1 c	0.456 a	0.456 a	0.468 a
2	34.4 a	18.7 c	0.480 a	0.478 a	0.495 a
3	35.0 a	20.2 c	0.465 a	0.472 a	0.466 a
4	40.3 a	23.1 b	0.454 a	0.447 a	0.468 a
5	43.6 a	25.6 a	0.451 a	0.449 a	0.447 a

Means within a column followed by a different letter differ significantly at $P < 0.05$.

Crown = crown length; D.b.h. = diameter at breast height; SG = weighted average specific gravity of the entire stem at d.b.h.; SG-PRE = weighted average specific gravity of wood formed prior to the first commercial thinning; SG-THIN = weighted average specific gravity of wood formed during commercial thinnings.

Table 3—Taper of the lowest, middle, and uppermost logs (defined as 16 feet of stem length) in response to five density management regimes conducted in a loblolly pine plantation in northcentral Louisiana

Regime	LOW	MID	UPPER
	---------- *feet per inch* ----------		
1	7.9 a	24.3 a	17.2 a
2	7.4 a	27.1 a	18.3 a
3	8.3 a	22.4 a	17.7 a
4	5.9 a	24.6 a	20.5 a
5	6.0 a	20.0 a	12.9 a

Means within a column followed by a different letter differ significantly at $P < 0.05$.

LOW = taper of the lowest logs; MID = taper of the middle logs; UPPER = taper of the uppermost logs.

differences among density regimes. Given the correlation between stem taper and characteristics such as strength, stiffness, and dimensional stability, it is possible that these regimes produced wood with similar key characteristics.

Specific gravity values were within average values for loblolly pine in previous studies (Zobel 1972, Zobel and McElwee 1958), and no differences in specific gravity were observed among treatments (table 2). Larson and others (2001) noted that specific gravity of loblolly pines planted at close spacings seldom differ from that of wider spacings because summerwood formation is greatly restricted by close spacings. Such an effect would likely be pronounced at this site because summer precipitation of the region is typically substantially below potential evapotranspiration of the same period (Blazier and Clason 2006).

CONCLUSIONS

The lack of differences in crown length, taper, and specific gravity among the wide gradient of density management regimes in this study provides evidence that many key wood properties were relatively resistant to change at this site. This may indicate that forest managers have great flexibility in density management from a wood quality perspective at such a site. However, before a broad assertion about density management can be made with greater certainty, more wood

quality characteristics should be studied and a greater array of sites must be observed.

LITERATURE CITED

Blazier, M.A.; Clason, T.R. 2006. Eleven-year loblolly pine growth in response to site preparation and seedling type in northern Louisiana. In: Conner, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 167–170.

Bohannan, B.; Habermann, H.; Lengel, J.E. 1974. Taper of wood poles. Gen. Tech. Rep. FPL-2. Madison, WI: U.S. Department of Agriculture Forest Service, Forest Products Laboratory. 9 p.

Brazier, J.D. 1986. Growth features and structural wood performance. In: Proceedings of the 18th IUFRO World Congress. Ljubljana, Yugoslavia: IUFRO World Congress: 37–49.

Brix, H. 1982. Effects of thinning and nitrogen fertilization on growth of Douglas-fir: relative contribution of foliage quantity and efficiency. Canadian Journal of Forest Research. 13: 167–175.

Clark III, A.; Daniels, R.F.; Jordan, L. 2006. Juvenile/mature wood transition in loblolly pine as defined by annual ring specific gravity, proportion of latewood, and microfibril angle. Wood and Fiber Science. 38(2): 293–299.

Clark III, A.; Jordan, L.; Schimleck, L.; Daniels, R.F. 2008. Effect of initial planting spacing on wood properties of unthinned loblolly pine at age 21. Forest Products Journal. 58(10): 78–83.

Clark III, 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.

Clason, T.R. 1994. Impact of intraspecific competition on growth and financial development of loblolly pine plantations. New Forests. 8: 185–210.

Craib, I.J. 1947. The silviculture of exotic conifers in South Africa. British Empire Forestry Conference. Pietermaritzburg, South Africa: City Printing Works. 35 p.

Dean, T.J.; Long, J.N. 1986. Validity of constant-stress and elastic-instability principles of stem formation in *Pinus contorta* and *Trifolium pretense*. Annals of Botany. 58: 833–840.

Harms, W.R.; Whitesell, C.D.; DeBell, D.S. 2000. Growth and development of loblolly pine in a spacing trial planted in Hawaii. Forest Ecology and Management. 126: 23–24.

Kilpatrick, W. Wayne; Henry, Charles, Jr. 1989. Soil survey of Claiborne Parish, Louisiana. [Place of publication unknown]: U.S. Department of Agriculture, Soil Conservation Service. In cooperation with: U.S. Department of Agriculture Forest Service; Louisiana Agricultural Experiment Station; and Louisiana Soil and Water Conservation Committee. 147 p.

Larson, L.; Kretschmann, D.D.; Clark III, A.; Isebrands, J.G. 2001. Formation and properties of juvenile wood in southern pines: a synopsis. Gen. Tech. Rep. FPL-GTR-129. Madison, WI: U.S. Department of Agriculture Forest Service, Forest Products Laboratory. 42 p.

Macdonald, E.; Hubert, J. 2002. A review of the effects of silviculture on timber quality of Sitka spruce. Forestry. 75(2): 107–138.

Metzger, K. 1893. Der wind als massgebender faktor fur das wachtsum der baume. Mundener Forstliche Hefte. 3: 35–86.

O'Connor, A.J. 1935. Forest research with special references to planting distances and thinning. British Empire Forestry Conference. Pietermaritzburg, South Africa: City Printing Works. 30 p.

Sharma, M.; Burkhart, H.E.; Amateis, R.L. 2002. Modeling the effect of density on the growth of loblolly pine trees. Southern Journal of Applied Forestry. 26(3): 124–133.

Visser, R.; Stampfer, K. 2003. Tree-length system evaluation of second thinning in a loblolly pine plantation. Southern Journal of Applied Forestry. 27(2): 77–82.

Zahner, R.; Whitmore, F.W. 1960. Early growth of radically thinned loblolly pine. Journal of Forestry. 58: 628–634.

Zhang, S.; Burkhart, H.E.; Amateis, R.L. 1996. Modeling individual tree growth for juvenile loblolly pine plantations. Forest Ecology and Management. 89: 157–172.

Zobel, B.J. 1972. Three-rings-per inch dense southern pine: can it be developed. Journal of Forestry. 70: 333–335.

Zobel, B.J.; McElwee, R.L. 1958. Natural variation in wood specific gravity of loblolly pine, and an analysis of contributing factors. TAPPI Journal. 41: 158–161.

SHORT-TERM EFFECTS OF SILVICULTURE ON BREEDING BIRDS IN WILLIAM B. BANKHEAD NATIONAL FOREST

Jill M. Wick, Yong Wang, and Callie Jo Schweitzer[1]

Abstract—We evaluated the changes in the bird community in relation to six disturbance treatments in the William B. Bankhead National Forest, AL. The study design is randomized complete block with a factorial arrangement of three thinning levels [no thin, 11 m^2/ha residual basal area (BA), and 17 m^2/ha residual BA] and two burn treatments (burn and no burn), with three replications. We collected data from pre- and post-treatment avian line-transect surveys. We found that the silvicultural treatments appear to create habitat for early successional bird species.

INTRODUCTION

The decline of neotropical migratory songbirds in Eastern North America has been a subject of much discussion among ornithologists over the past two decades (Askins and others 1990, Finch 1991, James and others 1996, Rappole and McDonald 1994, Robbins and others 1989). Although some evidence of declines is conflicting, it is generally accepted that due to general trends of habitat loss and degradation, and their importance to the ecosystems, giving priority conservation status to neotropical songbirds is justified. In recent studies, the decline of birds associated with early successional breeding habitat has been noted (Askins and others 1990, Hunter and others 2001, Litvaitis and others 1999). Trani and others (2001) reported that, according to Forest Inventory and Analysis data, young forest habitats are declining due to forest maturation and the absence of timber removal on much public land. Tree removal creates early successional habitat by removing trees to create an environment favorable for tree growth or regeneration (Smith and others 1997). As forest management evolves to employ multiple silvicultural tools to meet a myriad of objectives, it is important to understand how such management affects the bird community and if quality early successional wildlife habitat is produced.

Prescribed burning has garnered heightened awareness on public lands as a silvicultural technique since fire suppression in eastern forests has been questioned (Brose and others 2001, Van Lear and Waldrop 1989). Although the effect of silviculture and fire on birds has been studied individually in eastern forests, there is little research assessing the effect of thinning and prescribed burning (Greenberg and others 1995, 2007) and only one study reports the effects when tree reduction and burning are combined (Wilson and others 1995). It is important to understand how these treatments will affect the bird community when compared to other silvicultural techniques.

The objective of this portion of the study was to quantify the bird community on six silvicultural treatments in the William B. Bankhead National Forest.

METHODS

Study Sites

The study was located in the northern one-third of William B. Bankhead National Forest (BNF), located in Lawrence and Winston Counties, northwestern Alabama. The forests in this region have a diverse species composition due to a variety of past disturbances—agriculture in the 1800s, heavy cutting and wildfire in the early 1900s, fire suppression in the last decade, and the recent infestation of the southern pine beetle (*Dendroctonus frontalis* Zimmerman) (Gaines and Creed 2003). In the 1930s, abandoned farmland and other open lands were reestablished with loblolly pine (*Pinus taeda* L.) (Gaines and Creed 2003). This has resulted in 31 600 ha of loblolly pine throughout BNF. Once established, intensive pine plantation management was not implemented, and subsequently, a variety of hardwood species voluntarily invaded these sites. Over the past decade, southern pine beetle infestations have killed a major portion of loblolly pine, increasing fuel loads and the risk of wildfires (Gaines and Creed 2003). BNF has initiated a Forest Health and Restoration Project to promote healthy forest growth via thinning and fire disturbance. The thinning and fire prescriptions were administered to return the forest to a more healthy state and to promote regeneration of native species. Our research was conducted in conjunction with BNF's restoration project.

The study design consisted of a randomized complete block design with two factors: three thinning levels [no thin, 11 m^2/ha residual basal area (BA), and 17 m^2/ha residual BA] and two burn treatments (no burn and burn). Each treatment was replicated three times and blocked by year. Treatments were assigned randomly to delineated stands. After the treatments were completed, we collapsed the thinned treatments together because there was no difference in BA between the two thinning levels ($F = 0.07$, $df = 1$, $P = 0.8$). This created three replicates each of the control and burn, and six replicates each of the thin and the thin/burn. The research stands were located on upland sites composed of 20- to 35-year-old loblolly pine. Stands were comprised of a minimum of 60 percent pine [loblolly pine or Virginia pine

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(*P. virginiana* Mill.)], with the remainder mainly oak species (*Quercus* spp.). Average stand size was 12 ha and plots had similar age and stand density. Thinning favored the retention of hardwood species and was done before fire prescriptions. Prescribed burning was completed in the dormant season (January through March) with low-burning surface fires. Treatments on block one were completed between August 2005 and February 1, 2006; blocks two and three were treated between April 2006 and March 2007. Pre-treatment data was collected from all stands between April and June 2005. Post-treatment data was collected from block one between April and June 2006, and from blocks two and three between April and June 2007.

Sampling

We sampled the bird community using line-transect surveys and distance sampling methods (Buckland and others 2001). Line transects were established on each of the stands and flagged every 25 m. Each transect was 50 m from the edge of the stand and 100 m wide; the observer slowly walked down the middle of the transect and recorded all birds heard or seen within 50 m on either side. The observer recorded the following: species, sex, age, and the location of the bird in relation to the transect.

All stands were surveyed three times during the breeding season (May 15 through June 30) between 0530 and 1030 central daylight savings time. Surveys were done in random order and the transects walked in a different order and direction at each visit. All surveys were conducted by JMW to avoid observer bias.

To create a relative bird abundance index, we divided the number of detections by the transect length for each stand. Stands differed in size and shape and transect lengths differed among stands as well. We used the greatest number of individuals detected among the three surveys to estimate the relative abundance of each species. We grouped species into four guilds based on their habitat association (Blake and Karr 1987, Freemark and Collins 1992) (tables 1 and 2).

RESULTS

Before treatment, a total of 1,185 birds were detected, representing 35 species (table 1). The most abundant species were the red-eyed vireo (*Vireo olivaceus* Linnaeus), comprising 20.9 percent of total individuals, and the pine warbler (*Dendroica pinus* Wilson), comprising 11.6 percent of total individuals.

A total of 983 birds were detected 1 year after treatment, representing 40 species (table 2). The most abundant species were the red-eyed vireo, comprising 16.5 percent of total individuals, and the pine warbler, comprising 14.0 percent of total individuals. Species detected post-treatment that were not detected before treatment were the brown-headed nuthatch (*Sitta pusilla* Latham), eastern phoebe (*Sayornis phoebe* Latham), eastern towhee (*Pipilo erythrophthalmus* Linnaeus), eastern wood-pewee (*Contopus virens* Linnaeus), mourning dove (*Zenaida macroura* Linnaeus), ruby-throated hummingbird (*Archilochus coulbris* Linnaeus), and yellow-throated vireo (*V. flavifrons* Vieillot). Two species [blue grosbeak (*Guiraca caerulea* Linnaeus) and red-bellied woodpecker (*Melanerpes carolinus* Linnaeus)] detected before treatments were not detected post-treatment.

DISCUSSION

The overall structure of the bird community before treatment appears to be a midsuccessional forest. The bird community consisted of a majority of shrub-nesting species and interior/edge dwelling species. Optimal habitat for these guilds was created by the presence of wildlife openings, roads, and southern pine beetle damaged areas within many of the plots, which create small pockets of open areas and increase the amount of edge.

One year after treatment there was a treatment effect on some aspects of the bird community; it is likely a result of changes in microhabitat among treatments. Seven species were detected after silvicultural treatment that were not detected before treatment; six of these species prefer early successional forests. This suggests that silvicultural treatments that leave trees are viable options for creating habitat for early successional birds if clearcutting is not an option or if retaining mature forest birds is also a management goal. Many other studies have found that when some trees are retained, as in shelterwood and selection cuts, edge and open habitat bird species use the habitat for a short time and many mature forest birds remain (Campbell and others 2007, Greenberg and others 2007, Holmes and Pitt 2007, Lanham and others 2002, Vanderwel and others 2007, Weakland and others 2002). However, Costello and others (2000) suggest that there may be a minimum opening size requirement for some species associated with early successional habitat. Treatments that retain some trees may not create openings large enough to support all species that use early successional habitat for breeding.

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LITERATURE CITED

Askins, R.A.; Lynch, J.F.; Greenberg, R. 1990. Population declines in migratory birds in Eastern North America. Current Ornithology. 7: 1–15.

Blake, J.G.; Karr, J.R. 1987. Breeding birds of isolated woodlots: area and habitat relationships. Ecology. 68: 1724–1734.

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. 32: 30–35.

Table 1—Species detected before treatment on 18 upland pine-hardwood stands in Bankhead National Forest, AL, classified by habitat association

Common name	Scientific name	Habitat guild
Acadian flycatcher	*Empidonax virescens* Vieillot	I
Black-and-white warbler	*Mniotilta varia* Linnaeus	I
Blue-gray gnatcatcher	*Polioptila caerulea* Linnaeus	I/E
Brown-headed cowbird	*Molothrus ater* Boddaert	O/E
Blue-headed vireo	*Vireo solitarius* Wilson	I/E
Blue grosbeak	*Guiraca caerulea* Linnaeus	O/E
Blue jay	*Cyanocitta cristata* Linnaeus	I/E
Brown thrasher	*Toxostoma rufum* Linnaeus	O/E
Black-throated green warbler	*Dendroica virens* Gmelin	I
Carolina chickadee	*Poecile carolinensis* Audubon	I/E
Carolina wren	*Thryothorus ludovicianus* Latham	O/E
Downy woodpecker	*Picoides pubescens* Linnaeus	I/E
Tufted titmouse	*Baeolophus bicolor* Linnaeus	I/E
Great crested flycatcher	*Myiarchus crinitus* Linnaeus	I/E
Hairy woodpecker	*Picoides villosus* Linnaeus	I
Hooded warbler	*Wilsonia citrina* Boddaert	I
Indigo bunting	*Passerina cyanea* Linnaeus	O/E
Kentucky warbler	*Oporornis formosus* Wilson	I/E
Louisiana waterthrush	*Seiurus motacilla* Vieillot	I/E
Northern cardinal	*Cardinalis cardinalis* Linnaeus	I/E
Northern mockingbird	*Mimus polyglottos* Linnaeus	E
Northern parula	*Parula americana* Linnaeus	I/E
Ovenbird	*Seiurus aurocapillus* Linnaeus	I
Pine warbler	*Dendroica pinus* Wilson	I/E
Pileated woodpecker	*Dryocopus pileatus* Linnaeus	I
Prairie warbler	*Dendroica discolor* Vieillot	O/E
Red-bellied woodpecker	*Melanerpes carolinus* Linnaeus	I/E
Red-eyed vireo	*Vireo olivaceus* Linnaeus	I/E
Scarlet tanager	*Piranga olivacea* Gmelin	I
Summer tanager	*Piranga rubra* Linnaeus	I/E
White-breasted nuthatch	*Sitta carolinensis* Latham	I
White-eyed vireo	*Vireo griseus* Boddaert	O/E
Worm-eating warbler	*Helmitheros vermivorus* Gmelin	I
Wood thrush	*Hylocichla mustelina* Gmelin	I/E
Yellow-breasted chat	*Icteria virens* Linnaeus	O/E
Yellow-billed cuckoo	*Coccyzus americanus* Linnaeus	I/E

E = edge; I = interior; I/E = interior-edge; O/E = open/edge.

Source: Blake and Karr (1987), Freemark and Collins (1992).

Table 2—Species detected after silvicultural treatment on 18 upland pine-hardwood stands in Bankhead National Forest, AL, classified by habitat association

Common name	Scientific name	Habitat guild
Acadian flycatcher	*Empidonax virescens* Vieillot	I
Black-and-white warbler	*Mniotilta varia* Linnaeus	I
Blue-gray gnatcatcher	*Polioptila caerulea* Linnaeus	I/E
Brown-headed cowbird	*Molothrus ater* Boddaert	O/E
Brown-headed nuthatch	*Sitta pusilla* Latham	I
Blue-headed vireo	*Vireo solitarius* Wilson	I/E
Blue jay	*Cyanocitta cristata* Linnaeus	I/E
Brown thrasher	*Toxostoma rufum* Linnaeus	O/E
Black-throated green warbler	*Dendroica virens* Gmelin	I
Carolina chickadee	*Poecile carolinensis* Audubon	I/E
Carolina wren	*Thryothorus ludovicianus* Latham	O/E
Downy woodpecker	*Picoides pubescens* Linnaeus	I/E
Eastern phoebe	*Sayornis phoebe* Latham	I/E
Eastern towhee	*Pipilo erythrophthalmus* Linnaeus	I/E
Eastern wood-pewee	*Contopus virens* Linnaeus	I/E
Tufted titmouse	*Baeolophus bicolor* Linnaeus	I/E
Great crested flycatcher	*Myiarchus crinitus* Linnaeus	I/E
Hairy woodpecker	*Picoides villosus* Linnaeus	I
Hooded warbler	*Wilsonia citrina* Boddaert	I
Indigo bunting	*Passerina cyanea* Linnaeus	O/E
Kentucky warbler	*Oporornis formosus* Wilson	I/E
Louisiana waterthrush	*Seiurus motacilla* Vieillot	I/E
Mourning dove	*Zenaida macroura* Linnaeus	O/E
Northern cardinal	*Cardinalis cardinalis* Linnaeus	I/E
Northern parula	*Parula Americana* Linnaeus	I/E
Ovenbird	*Seiurus aurocapillus* Linnaeus	I
Pine warbler	*Dendroica pinus* Wilson	I/E
Pileated woodpecker	*Dryocopus pileatus* Linnaeus	I
Prairie warbler	*Dendroica discolor* Vieillot	O/E
Red-eyed vireo	*Vireo olivaceus* Linnaeus	I/E
Ruby-throated hummingbird	*Archilochus coulbris* Linnaeus	O/E
Scarlet tanager	*Piranga olivacea* Gmelin	I
Summer tanager	*Piranga rubra* Linnaeus	I/E
White-breasted nuthatch	*Sitta carolinensis* Latham	I
White-eyed vireo	*Vireo griseus* Boddaert	O/E
Worm-eating warbler	*Helmitheros vermivorus* Gmelin	I
Wood thrush	*Hylocichla mustelina* Gmelin	I/E
Yellow-breasted chat	*Icteria virens* Linnaeus	O/E
Yellow-billed cuckoo	*Coccyzus americanus* Linnaeus	I/E
Yellow-throated vireo	*Vireo flavifrons* Vieillot	I/E

I = interior; I/E = interior-edge; O/E = open/edge.

Source: Blake and Karr (1987), Freemark and Collins (1992).

Buckland, S.T.; Anderson, D.R.; Burnham, K.P. [and others]. 2001. Introduction to distance sampling. Oxford, United Kingdom: Oxford University Press. http://www.ruwpa.st-and.ac.uk/distance.book/intro.html. [Date accessed: June 13, 2011].

Campbell, S.P.; Witham, J.W.; Hunter, M.L., Jr. 2007. Long-term effects of group-selection timber harvesting on abundance of forest birds. Conservation Biology. 21: 1218–1229.

Costello, C.A.; Yamasaki, M.; Penkins, 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.

Finch, D.M. 1991. Population ecology, habitat requirements, and conservation of neotropical migratory birds. Gen. Tech. Rep. RM-205. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Forest and Range Experiment Station. 26 p.

Freemark, K.; Collins, B. 1992. Landscape ecology of birds breeding in temperate forest fragments. In: Hagen, J.M. III; Johnston, D.W., eds. Ecology and conservation of neotropical migrant landbirds. Washington, DC: Smithsonian Institution Press: 443–454.

Gaines, G.D.; Creed, J.W. 2003. Forest health and restoration project. National forests in Alabama, Bankhead National Forest Franklin, Lawrence and Winston Counties, Alabama. Final environmental impact statement. Manage. Bull. R8-MB 110B. http://www.fs.fed.us/outernet/r8/alabama/forests/bankhead/documents/finalbankheadreport.pdf. [Date accessed: June 13, 2011].

Greenberg, C.H.; Harris, L.D.; Neary, D.G. 1995. A comparison of bird communities in burned and salvage-logged, clearcut, and forested Florida sand pine scrub. Wilson Bulletin. 107: 40–54.

Greenberg, C.H.; Tomcho, A.L.; Lanham, J.D. [and others]. 2007. Short term effects of fire and other fuel reduction treatments on breeding birds in a Southern Appalachian upland hardwood forest. Journal of Wildlife Management. 71: 1906–1916.

Holmes, S.B.; Pitt, D.G. 2007. Response of bird communities to selection harvesting in a northern tolerant hardwood forest. Forest Ecology and Management. 238: 280–292.

Hunter, W.C.; Buehler, D.A.; Canterbury, R.A. [and others]. 2001. Conservation of disturbance-dependent birds in Eastern North America. Wildlife Society Bulletin. 29: 440–455.

James, F.C.; McCullouch, C.E.; Wiedenfeld, D.A. 1996. New approaches to the analysis of population trends in landbirds. Ecology. 77: 13–27.

Lanham, J.D.; Keyser, P.D.; Brose, P.H.; Van Lear, D.H. 2002. Oak regeneration using the shelterwood-burn technique: management options and implications for songbird conservation in the Southeastern United States. Forest Ecology and Management. 155: 143–152.

Litvaitis, J.A.; Wagner, D.L.; Tarr, M.D.; Snyder, E.J. 1999. Early successional forests and shrub-dominated habitats: land use artifact or critical community in the Northeastern United States. Northeast Wildlife. 54: 101–118.

Rappole, J.H.; McDonald, M.V. 1994. Cause and effect in population declines of migratory birds. The Auk. 11: 652–660.

Robbins, C.S.; Sauer, J.R.; Greenberg, R.S.; Droege, S. 1989. Population declines in North American birds that migrate to the neotropics. Proceedings of the National Academy of Science. 86: 7658–7662.

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

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.

Vanderwel, M.C.; Malcom, J.R.; Mills, S.C. 2007. A meta-analysis of bird responses to uniform partial harvesting across North America. Conservation Biology. 21: 1230–1240.

Van Lear, D.H.; Waldrop, T.A. 1989. History, uses, and effects of fire in the Southern Appalachians. Gen. Tech. Rep. SE-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 20 p.

Weakland, C.A.; Wood, P.B.; Ford, W.M. 2002. Responses of songbirds to diameter-limit cutting in the central Appalachians of West Virginia, USA. Forest Ecology and Management. 155: 115–129.

Wilson, C.W.; Masters, R.E.; Bukenhofer, G.A. 1995. Breeding bird response to pine-grassland community restoration for red-cockaded woodpeckers. Journal of Wildlife Management. 59: 56–67.

A BETTER WAY—UNEVEN-AGED MANAGEMENT OF SOUTHERN YELLOW PINE

Don M. Handley and Joshua C. Dickinson[1]

Abstract—Uneven-aged management of southern yellow pine offers family forest owners in the Coastal Plain and Piedmont of the Southeast an attractive economic alternative to the two most common forestry scenarios. First, the great majority of owners practice no management. Too often they call in a timber buyer or procurement forester who high grades the forest. Second are the owners who follow the widely promoted industrial model of even-aged plantations. In either scenario, the owner can expect one major income event in a lifetime, followed, if he chooses, by a significant investment in site preparation and replanting. The return from either of these once-in-a-lifetime events is generally significantly less than what could be earned over time under uneven-aged management. With the help of a trained forester, owners have historically earned over $100 per acre per year while maintaining full stocking of their forest.

INTRODUCTION

This advocacy paper is directed at two related audiences—family forest owners in the Coastal Plain and Piedmont of the Southeast and the willing foresters who are capable of serving them. Family forest owners control 70 percent of the forests in the region (Baker and others 1996). How these forests are managed is important. Will forestry be competitive with conversion to 5-acre ranchettes? The answer will be highly significant to both the regional economy and environment. Uneven-aged management offers the potential for higher and more continuous income from the forest than even-aged management of a plantation or simply holding onto unmanaged forest. The message for family forest owners is that ... there is a better way! We believe uneven-aged management of pine is the best option for most family forest owners.

Two Groups of Family Forest Owners

The first group of landowners, controlling the great majority of family forests, are the individuals who own, inherit, or buy a block of unmanaged forest land with varying proportions of pine (*Pinus* spp.) and low-value hardwoods. Into this group fall the first wave of the largest intergenerational transfer of family forest land in American history according to the Pinchot Institute (The Pinchot Letter 2005). Current owners, many in their 50s, are often not certain what their heirs are likely to do with the land. Few current owners practice active management and their offspring are only marginally knowledgeable about the value of the family's forest. Falling within this category also are more traditional southern landowners who see their forests as an untouched reserve, to be logged only in a dire emergency. Both tend to fall victim to the timber buyer with his "we buy timber" signs along rural roads, who offers what appears to be a whopping price for the timber, removes every merchantable stem, and leaves the forest trashed. For too many, income from the forest translates into a once-in-a-lifetime liquidation of the forest cover. Without past experience in active management, these owners may opt to sell their land rather than make the considerable investment in site preparation and replanting.

The second group of landowners has stands of planted pine. The owner practicing active management will carry out one to three thinnings as the trees grow to rotation age at 25 or more years. The owner may contract a consulting forester who will inventory the stand, get lump sum bids for the timber, and supervise the sale. Once the clearcut has been carried out, the forester may be called on to hire a crew to carry out site preparation and replanting. However, when the landowner contemplates the expense of replanting and waiting a generation before receiving appreciable income, he may call upon the same forester, as a licensed realtor, to sell the land.

An Inappropriate Model

Currently family forest owner's interests are not well served, either by the prevailing industrial forestry model or by the foresters trained in the application of that model. Since the 1960s continuing expansion of fast-growing plantations has assured forest industry a continued supply of cheap raw material for conversion into value-added products such as paper, oriented strand board, and low-quality lumber. A result has been the declining prestige of southern yellow pine as a quality building material. Experiment stations and industry in the Southeast have selected pines for maximum radial growth in plantations. Chipping saws recover one or two boards from larger pulpwood logs, with the remaining chips going into pulp. The two growth-rings-per-inch lumber entering the market in increasing volumes is despised by builders because of its tendency to warp.

Industrial forestry is in flux. Traditional forest industries like Georgia Pacific and International Paper have discovered that owning forests is not a prerequisite for meeting their raw material needs. As they divest, holdings have been bought by Timber Investment Management Organizations, i.e., Wachovia (Wells Fargo?) and John Hancock, and Real Estate Investment Trusts such as Plum Creek. This new class of owners is likely to continue the forestry practices of their predecessors, with real estate sales as an integral part of their business model.

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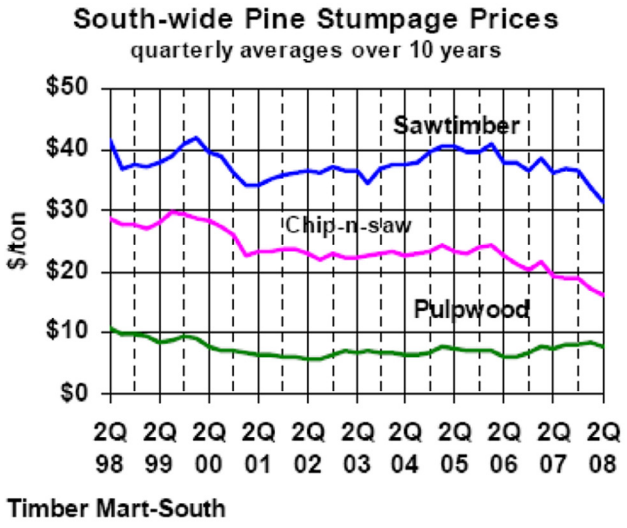

Figure 1—Southwide pine stumpage prices. Source: TimberMart-South, 2008, 13(2).

A continuing shift of pulpwood production offshore to countries with even higher growth rates and lower production costs such as Brazil and New Zealand assures that pulpwood remains cheap. Environmentalist pressure on industry to increase recycled content in paper will also tend to drive pulpwood prices lower, as will the shift from print to electronic media. Family forest owners that have been induced by extensionists and industry to convert their land to short-rotation plantations are stuck, they are assured bottom dollar for their pulpwood as supply continues to outstrip demand. As is clear in figure 1, producing pulpwood as a final product is a losing proposition for the family forest owner. Prices have not exceeded $10 per ton for a decade. Despite the no-win bind they find themselves in, family plantation owners are recognized by the forestry profession as progressive tree farmers.

There are good reasons why the great majority of family forest owners have not bought into the even-aged plantation model. Prominent is the realization that following the investment in site preparation and planting of seedlings one significant income event in a lifetime will occur when a plantation can be logged for sawtimber at rotation age. Some owners find the site disturbance associated with plantation establishment to be environmentally and aesthetically unacceptable. Others, particularly absentee inheritors of forest land, may be so disengaged that they give little consideration to any type of management. Family forest owners who follow the literature may be influenced by the significant bias against uneven-aged management and natural forest management in general (Bragg and others 2006, Cafferata and Kemperer 2000).

The environmental impact of plantation forestry is significant. Conversion to even-aged plantations negatively impacts an array of species, including a number that are threatened or endangered, the red-cockaded woodpecker (*Picoides borealis*) and flatwoods salamander (*Ambystoma cingulatum*) for example. Intensive site preparation results in habitat disruption and erosion of both soil and nutrients, negatively impacting aquatic ecosystems. Dense plantation monocultures of genetically similar trees are vulnerable to pest and disease outbreak. Aesthetically, even-age plantations are monotonous and uninteresting. The magnitude of the environmental threat is enormous. The U.S. Forest Service Southern Forest Resource Assessment predicts that the area in plantations is expected to increase from 32 to 54 million acres by 2040 and natural forest types to decrease from 149 to 122 million acres during the same period (Wear and Greis 2002).

Uneven-Aged Management—History

Uneven-aged management is not new, only largely forgotten—by foresters and forest owners. The practice of what evolved into uneven-aged management dates back to the mid-1920s in Arkansas when foresters L.K. Pomeroy and E.P. Connor founded the Ozark Badger Lumber Company. Their approach stood in sharp contrast to the “cut and run” logging of old-growth pine forests that had prevailed for decades. Pomeroy’s perspective was strongly influenced by a 1934 trip to visit forests in Germany where management had been practiced for centuries. Pomeroy noted, “Their attitude of guardianship of this [forest] wealth for future generations was a point entirely strange to me as an American lumberman.” (Pomeroy 1989). Pomeroy’s epiphany can be compared with that wrought by Gifford Pinchot and German, Carl Schenck, at the Biltmore Estate in North Carolina two decades earlier.

The Arkansas model also differed markedly from the “sustainable” forestry models advocated by leading forestry schools and the U.S. Forest Service. Good forestry at the time consisted of cutting all trees over a given diameter and leaving three to five seed trees per acre. This system was appropriate for large operations with many stands of different ages. However, for family forest operations envisioned by Pomeroy and Connor, this practice would leave the owner with long periods with no income from the forest. In the simplest terms, other foresters advocated cutting two-thirds of the stand and leaving up to five seed trees per acre; their approach to uneven-aged management involved leaving two-thirds of the stand and cutting up to five mature trees per acre.

R.R. Reynolds of the U.S. Forest Service established the Crossett Experimental Forest near Crossett, AR, in 1933 and directed its activities for the next 34 years (Reynolds 1980). In 1939, Reynolds established the “Good Farm Forestry Forty,” a well-stocked shortleaf/loblolly (*P. echinata*/*P. taeda*) stand, to demonstrate to farmers that good income can be generated under uneven-aged management, even from relatively small forest properties. This 40-acre parcel is still being managed and harvested today. The secret to the success of uneven-aged management for the family forest owner is the frequent sale of high-value, mature trees. This periodic thinning assures abundant replenishment of young seedlings and competition control in a multiaged forest, while maintaining near full stocking. Assuming that a competitive sale can be made on 50 acres of well-stocked timber, an owner with 300 acres could expect to have a sale every year, if desired.

Steady income coupled with the hydrological and wildlife benefits of maintaining a fully stocked forest ecosystem are among the benefits on uneven-aged management.

The Economics of Uneven-Aged Management

Don Handley grew up near Crossett, AR. Prior to college he did inventories and logging in forests under uneven-aged management for L.R. Pomeroy. His degree work in forestry at Arkansas A&M College at Monticello was closely linked to Reynolds' work at Crossett Experimental Forest. After graduating, Handley moved to South Carolina where he introduced uneven-aged management to family forest clients with a total of several thousand acres of forest. The examples addressed below are representative of the typical smaller forested ownership unit in the southeastern Coastal Plain.

The first example is from across the border from Handley's base in Florence, SC, in southeastern North Carolina. Handley helped this client convert a 45-acre, 20-year-old plantation over to uneven-aged management after the first thinning. Harvest from this property should rival the Crossett "Good Farm Forestry Forty" when it reaches full stocking (table 1).

Comparison of even- and uneven-aged management is complex, but critical if family forest owners are to have a valid basis for judging which management option to choose. The 85-acre property in Florence County, SC, used as an illustration here is broadly representative of the great majority of family forests in the southeastern Coastal Plain and lower Piedmont (tables 2 and 3). In 1988 the 85-acre unmanaged successional forest had a mixture of loblolly pine and low-value hardwoods with hardwoods dominating the understory and little or no pine regeneration. The beginning inventory was approximately 300 cords (800 tons) of pulpwood and 430,000 board feet (3,225 tons) of sawtimber.

Table 1—45-acre family forest

Year	Activity	Costs	Income
		---------- *dollars* ----------	
1988	First pulpwood thinning		10,800
1993	Timber sale		35,367
1997	Timber sale		49,148
2003	Timber sale		40,650
2008	Timber sale	–9,000	31,605
Total net income (10 years)			158,570
Residual value		94,000	

Table 2—85-acre family forest, even-aged management

Year	Activity	Volume	Costs	Income
			---------- *dollars* ----------	
1988	Sale	300 cords and 430,000 board feet	7,455	74,550
1990	Site prep and planting		14,900	
2010	First thinning		4,080	40,800
Totals			26,435	115,350
Net income			88,915 ($48 per acre per year)	
2010	Residual value of standing timber		81,600	

Note: All costs and income estimated.

Assumptions—1988 prices for pulpwood: $12 per cord ($4.50 per ton), sawtimber: $165 per 1,000 board feet ($22 per ton). Assume $20 per cord pulpwood price in 2008.

Table 3—85-acre family forest, uneven-aged management

Year	Activity	Volume	Costs	Income
			---------- *dollars* ----------	
1988	Timber cruise		510	
1988	Prescribed burn		1,275	
1989	Pole sale	772 poles (88,580 board feet)	2,354	23,543
1989	Clear hardwood		200	2,000
1998	Herbicide application for pine release		4,845	
2000	Timber sale	152 cords pulpwood 231,018 board feet timber	8,853	88,530
2005	Timber sale	138 cords pulpwood	5,525	55,251
Total to date			23,562	169,324
Net income to date			145,762 ($100 per acre per year)	
Residual value of stand				178,500

Note: All costs and income amounts are actual.

Assume $20 per cord pulpwood price in 2008.

Assumptions—1988 prices for pulpwood: $12 per cord ($4.50 per ton), sawtimber: $165 per 1,000 board feet ($22 per ton). Assume $20 per cord pulpwood price in 2008.

For comparison, table 2 represents the typical costs and returns to be expected from even-aged management had the owner chosen to practice even-aged management. Following common practice the forester who cruised the timber would have supervised a sale, and after the clearcut, overseen replanting. The first thinning would come in about 20 years. Data for even-aged management is extrapolated from local experience.

This comparison of even- and uneven-aged management is valid for the majority of private forest owners in the Southeast. Why? In practice, the unmanaged forest owner considering adopting either even- or uneven-aged management is starting with a mixed forest that has received little or no prior investment. The owner either (a) harvests marketable timber, clears, and plants seedlings; or (b) selectively harvests, removes competing hardwoods, and manages a naturally reproducing stand. This reality contrasts with studies that use either bare ground or rotation age plantations as the points of departure for comparing even- and uneven-aged management (Henderson 2008).

For the uneven-aged management scenario provided here in table 3, income per year is significantly higher than what would have prevailed under even-aged plantation management. Under uneven-aged management, income per sale can be expected to increase as the stand reaches its full volume potential. Annual return from a stand can be expected to well exceed $100 per acre per year with harvests at 5-year intervals. On larger properties with multiple stands, a timber sale every year could be anticipated. The residual value would also increase substantially as full stocking is reached.

Justification for Recruitment of Foresters With a Family Forest Commitment

The two vignettes presented here are illustrative of a widespread dysfunction in how timber buyers serve the family forest owner.

First example—A widow was concerned about the large mortgage on her home. With her modest income, it just wasn't possible to make much of a dent in the principal. Her sister came to the rescue by offering her the opportunity to sell the timber from their jointly owned 100 acres of unmanaged forest to help pay off her mortgage. With a copy of the deed and the letter from her sister in hand, she called a local timber dealer referred to her by a friend.

The dealer had long experience in procuring timber for the local mills. He visited the property the following week and cruised the timber. He offered a contract for clearcutting all merchantable timber on the 100 acres. This would net her $70,000, just a little more than she owed on her home. She called her accountant to ask what her tax liability would be. Her accountant suggested she get a second opinion

from an independent consulting forester and gave her the name of Handley Forestry Services in Florence, SC. Don Handley responded to her call. He cruised the 100 acres of unmanaged pine and low-value hardwoods. He suggested a different approach. Rather than clearcutting, the pines were selectively marked for harvest. The mature trees, >16 inches d.b.h., and the lower quality or crowded smaller trees were selected for harvest as sawtimber and pulpwood. The low-value hardwoods that were large enough were cut for pulpwood. The rest was chipped for fuel to be used by a local paper mill. Handley's fee for the cruise, marking the trees to be cut, and handling the sale was $9,360. The sawtimber, pulpwood, and chips brought $84,608, netting Mrs. Williams $75,248.

More important, the skidder left the forest floor clean and prepared to receive seed from the residual pines. This results in a new stand of pine seedlings in the understory and a well-spaced stand of healthy trees of different sizes overhead. The sisters were left with a healthy uneven-aged stand of pine with a prolific crop of new seedlings on the ground. They have dedicated the extra revenue to be used for improvements on the property. These improvements include a better access road, new gates, and herbicide application to control the hardwoods that will compete with the young pines. In 5 years they will be able to have another selective cut, estimated at $65,000. The best trees will be left to grow into a more valuable diameter class, making future harvests, every 5 to 7 years, worth even more.

Second example—Timber buyers employ "spotters" as they are called locally. They "spot" forested properties for potential sale. An elderly African-American couple was approached by someone they knew in the community who told them their timber was worth a lot of money and they should consider selling their timber to a timber buyer.

Later, the spotter told the couple that the company was going to offer $7,000 for their timber, but he knew it was worth more and recommended they take nothing <$8,000. The lady decided to call the State forestry office. Someone there suggested she contact Handley Forestry Services. Don Handley and his son cruised the timber on the couple's property and calculated the timber was worth $32,000, four times what the spotter had offered. Don recommended that the owners thin the stand, probably earning more than the buyer had offered, and practice uneven-aged management for an even greater return over the long term. In this case, the couple decided to take the full $32,000, investing some of the money in replanting.

CONCLUSION

It is clear from preceding tables and vignettes that the family forest owner would be better served by practicing uneven-aged management. The problem is that uneven-aged management requires continuing guidance from a knowledgeable forester. Few foresters have appropriate training because demand for their services has come through either procuring wood for industry or helping a relatively small number of private clients practice even-aged forest management oriented toward meeting industry's fiber demand.

Foresters should note how many years Handley Forest Services has worked with each client represented in the preceding tables. Uneven-aged management represents a potential lifetime engagement with the client. The consulting forester is not under constant pressure to find new clients willing to have their plantations clearcut or unmanaged forests high-graded in a once-in-a-lifetime fee-generating event. Even when the forester is engaged to oversee site preparation and planting of seedlings following harvest, a generation will pass before he might be called upon to direct a thinning operation.

We are launching a program to reach out to family forest owners and to foresters willing to learn how to serve them and show them a better way; uneven-aged management of southern yellow pine.

LITERATURE CITED

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.

Bragg, D.C.; Guldin, J.M.; Shelton, M.G. 2006. Stacking the log deck, or some fallacies about natural pine management. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 37–41.

Cafferata, M.J.S.; Kemperer, W.D. 2000. Economic comparisons between even-aged and uneven-aged loblolly pine silvicultural systems. Tech. Bull. 801. Research Triangle Park, NC: National Council for Air and Stream Improvement, Inc. 64 p.

Henderson, J.H. 2008. Uneven versus even-aged management of southern pines. Forest Landowner. July/August: 5–8.

Pomeroy, E.R. 1989. Two men and a forest. Drew County Historical Journal. 4(1): 47–70.

Reynolds, R.R. 1980. The Crossett story. Gen. Tech. Rep. SO-32. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 40 p.

The New Generation of Private Forest Landowners: Brace for Change. 2008. The Pinchot Letter. Winter. 10(2): 1–4.

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

HARDWOOD SILVICULTURE—INTERMEDIATE TREATMENT AND STAND DEVELOPMENT

Hardwood stand on a loessial bluff on Crowley's Ridge in the St. Francis National Forest near Marianna, Arkansas. (Photo by James M. Guldin)

INFLUENCES OF TREE, STAND, AND SITE CHARACTERISTICS ON THE PRODUCTION OF EPICORMIC BRANCHES IN SOUTHERN BOTTOMLAND HARDWOOD FORESTS

James S. Meadows, J.C.G. Goelz, and Daniel A. Skojac, Jr.[1]

Abstract—Epicormic branches are adventitious twigs that develop from dormant buds found along the main bole of hardwood trees. These buds may be released at any time during the life of the tree in response to various types of stimuli. Epicormic branches cause defects in the underlying wood and may cause significant reductions in both log grade and subsequent lumber value. Species, stress, and sunlight have been proposed as the three major factors affecting production of epicormic branches, but no definitive research has been conducted to evaluate this hypothesis. This paper reports preliminary evaluations of the influences of several tree, stand, and site characteristics on production of epicormic branches in undisturbed stands of southern bottomland hardwoods. Tree characteristics evaluated include species, diameter class, and crown class; stand characteristics evaluated include stand density and site index. Each characteristic was examined individually and in combination with other characteristics to determine the level of influence on formation of epicormic branches.

INTRODUCTION

Successful management of hardwood forests for sawtimber production depends on development and maintenance of high-quality logs. Log quality, generally expressed as log grade, greatly affects the monetary value of the sawtimber volume produced by the tree. The value of a hardwood log decreases rapidly in the downward progression from grade 1 to grade 3. Consequently, any event or circumstance that reduces log grade also significantly reduces the value of both the tree and the entire stand.

Epicormic branches are adventitious twigs found along the main bole of many hardwood trees. They develop from dormant buds that may be released at any time during the life of a tree in response to a variety of stimuli (Carpenter and others 1989). Because they produce knots in the underlying wood, epicormic branches, if present in sufficient numbers, may reduce log grade in standing trees. As a result, the presence of epicormic branches along the boles of hardwood trees often becomes a serious problem in management of hardwood forests for high-quality sawtimber production.

According to the Forest Service, U.S. Department of Agriculture, standard grading rules for hardwood factory logs (Rast and others 1973), large epicormic branches (>3/8 inch in diameter at the bark surface) are defects on logs of all sizes, grades, and species. In general, small epicormic branches (≥3/8 inch in diameter at the bark surface) are defects on all logs <14 inches in scaling diameter, but only every other one is counted as a defect on logs 14 inches or more in scaling diameter. Small epicormic branches are not counted as defects on black cherry (*Prunus serotina*) logs or on grade 3 logs of soft hardwood species, such as sweetgum (*Liquidambar styraciflua*) and eastern cottonwood (*Populus deltoides*), regardless of log diameter.

The grade and associated value of any hardwood log may be reduced significantly by the presence of a sufficient number of epicormic branches. For example, production of epicormic branches following a seedtree cut in South Carolina was substantial enough to reduce the grade of 44 percent of the cherrybark oak (*Quercus pagoda*) butt logs (Stubbs 1986). In a survey of bottomland oak stands in northeastern Louisiana, Hedlund (1964) found that the presence of epicormic branches on upper logs caused a one-grade reduction in nearly 40 percent of the logs and a two-grade reduction in 23 percent of the logs. Meadows and Burkhardt (2001) reported that production of epicormic branches in a thinned willow oak (*Q. phellos*) stand in Alabama was sufficient to cause a one-grade reduction in 45 percent of the butt logs and 46 percent of the upper logs as well as a two-grade reduction in 7 percent of the butt logs. Meadows and Burkhardt (2001) suggested that, as a general rule, as few as five epicormic branches somewhat evenly distributed on a 16-foot-long log are enough to cause a reduction in log grade.

The seriousness of the presence of epicormic branches on hardwood logs becomes even more apparent when those logs are sawn into lumber at the mill. Epicormic branches produce small knots, or defects, in the underlying wood. These defects may reduce the grade and subsequent value of the lumber produced from those logs. One of the factors used to grade hardwood lumber is the surface area of defect-free wood, called a clearcutting (Hanks and others 1980). Because hardwood lumber grade is affected more by the number and spatial distribution of defects rather than by the size of individual defects, the small knots produced by epicormic branches can affect the length and number of clearcuttings obtained from the lumber and therefore reduce its grade. Because defects caused by epicormic branches limit the size of clearcuttings and because the value of high-grade lumber may be several times greater than the value of low-grade lumber, the presence of epicormic branches on hardwood logs may have a detrimental effect on both lumber grade and its associated value. In one case study in Alabama, over 50 percent of the volume of willow oak lumber that would

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have been classified at one of the highest grades in the absence of epicormic branch defects was reduced to lower grades in the presence of epicormic branch defects. Based on red oak (*Q. rubra*) lumber prices prevailing at the time of the study, defects caused by epicormic branches resulted in a 13-percent loss of value in the lumber produced (Meadows and Burkhardt 2001).

Production of epicormic branches along the boles of hardwood trees is a poorly understood phenomenon that may be responsible for annual losses of millions of dollars in potential revenue. It once was thought that epicormic branches developed on hardwood trees solely as a result of sudden exposure to direct sunlight, especially after some type of partial harvest operation or other disturbance in the stand. However, mounting evidence indicates that tree health plays a major role in determining the propensity of a hardwood tree to produce epicormic branches (Brown and Kormanik 1970, Erdmann and others 1985, Meadows 1993).

To address this issue, Meadows (1995) proposed that production of epicormic branches is controlled by complex interactions among species, stress, and sunlight. Hardwood species vary significantly in their susceptibility to the production of epicormic branches. For example, most oaks are highly vulnerable, whereas green ash (*Fraxinus pennsylvanica*) generally is not. Stress experienced by individual trees may be caused by climatic events, site and stand conditions, suppression, and both stand-level and tree-level disturbances. High levels of stress may reduce tree health and may stimulate the production of epicormic branches. Within the range of susceptibility associated with any given species, tree health serves as the mechanism that controls production of epicormic branches when the tree experiences some type of stress. Sudden exposure of the bole to direct sunlight following some type of natural or anthropogenic disturbance may trigger the release of dormant buds that develop into epicormic branches. Under this hypothesis, trees of resistant species and healthy trees, even of susceptible species, are less likely to produce epicormic branches than are unhealthy trees.

Because there has been no definitive research to evaluate this hypothesis, we initiated a new research program designed to describe and model the influences of several tree, stand, and site characteristics on the production of epicormic branches on butt logs of hardwood trees of various species in both unthinned and thinned bottomland hardwood forests. This paper reports summaries of data collected solely from unthinned stands.

METHODS

Characteristics of Sample Stands

The portion of the overall research project reported in this paper consists of a comprehensive survey of a variety of undisturbed hardwood stands across the South. Sample stands were selected to represent a range of site and stand conditions, including site type, site quality, stand type, stand age, and species composition. Stands in which logging or other major natural or anthropogenic disturbances have occurred within the past 20 years were excluded from the survey.

Since 2005, we have sampled seven different bottomland hardwood stands in Mississippi (table 1). In general, stands characterized by the elm-ash-sugarberry species association are dominated by green ash, Nuttall oak (*Q. texana*), overcup oak (*Q. lyrata*), willow oak, and American elm (*Ulmus americana*). Green ash, sugarberry (*Celtis laevigata*), American elm, and water hickory (*Carya aquatica*) are the most abundant species in the lower canopies of those stands. In contrast, stands characterized by the red oak-sweetgum species association are dominated by willow oak, cherrybark oak, Nuttall oak, swamp chestnut oak (*Q. michauxii*), and sweetgum. Sweetgum, sugarberry, and swamp chestnut oak are the most abundant species in the lower canopies of those stands.

Sampling Design and Data Collected

Within each sample stand, we systematically established a grid of temporary, circular, 0.1-acre plots. Distance between plots along a transect line and distance between transect lines varied from one stand to another, primarily depending

Table 1—Characteristics of sample stands

Stand number	Stand age	County and State	Predominant soil series	Dominant species association
	years			
1	68	Washington, MS	Sharkey clay	Elm-ash-sugarberry
2	80	Oktibbeha, MS	Mathiston silt loam	Red oak-sweetgum
3	65	Oktibbeha, MS	Mathiston silt loam	Red oak-sweetgum
4	54	Sharkey, MS	Sharkey clay	Red oak-sweetgum
5	74	Washington, MS	Sharkey clay	Elm-ash-sugarberry
6	56	Washington, MS	Sharkey clay	Elm-ash-sugarberry
7	78	Washington, MS	Sharkey clay	Elm-ash-sugarberry

on the terrain and the size of the stand. Minimum distance between plots and between lines was 150 feet. Our goal was to sample at least 25 plots in each stand.

Sampling was limited to living hardwood trees ≥5.5 inches d.b.h. Data collected on every sample tree included species, d.b.h., crown class, hardwood tree class as defined by Meadows and Skojac (2008), and the number of epicormic branches on the 16-foot-long butt log. The number of large epicormic branches (>3/8 inch in diameter at the bark surface) and the number of small epicormic branches (≤3/8 inch in diameter at the bark surface) were tallied separately on each tree. Other tree variables, such as crown diameter and crown length, were considered for inclusion in the model, but ultimately were rejected. Crown variables are difficult to measure accurately and consistently in standing hardwood trees because the crowns of most hardwood trees are irregularly shaped, both horizontally and vertically.

Stand-level information, such as site type, forest cover type, estimated stand age, site index, stand density, and stand history was collected for each sample stand. Site index was estimated using the technique developed by Baker and Broadfoot (1979). Stand density was determined for each plot and was expressed as square feet of basal area per acre.

We established 503 temporary, 0.1-acre plots and collected data from 5,106 trees ≥5.5 inches d.b.h. across the 7 sample stands. We then discarded data from trees of species unsuitable for sawtimber production, such as American hornbeam (*Carpinus caroliniana*), winged elm (*U. alata*), boxelder (*Acer negundo*), red mulberry (*Morus rubra*), and eastern hophornbeam (*Ostrya virginiana*). We thus included data from 5,057 trees in our evaluations.

Preliminary Data Evaluation

Data collected from all sample stands were pooled and summarized in a variety of combinations to produce preliminary evaluations of five major characteristics that may influence production of epicormic branches on hardwood trees in undisturbed stands: (1) species, (2) site quality, (3) stand density, (4) tree size, and (5) crown class. The latter four characteristics may be indicators of the degree of stress experienced by individual trees. The influence of each characteristic on the number of existing epicormic branches on the butt log was evaluated separately and in combination with other characteristics. Unfortunately, site quality did not differ sufficiently across our sample stands to allow adequate evaluation. Consequently, the influence of site quality on production of epicormic branches is not addressed in this paper.

RESULTS AND DISCUSSION

Number of Sample Trees by Species

After combining data across the 7 sample stands, there were 12 species with more than 75 observations each—6 oak species and 6 non-oak species (table 2). Nuttall, willow, and overcup oaks were particularly numerous among the oaks,

Table 2—Number of sample trees and Meadows (1995) rating of susceptibility to production of epicormic branches, by species, across seven bottomland hardwood stands in Mississippi

Common name	Scientific name	Susceptibility rating	Trees
			number
Green ash	*Fraxinus pennsylvanica*	Low	716
Nuttall oak	*Quercus texana*	High	674
Willow oak	*Q. phellos*	High	662
Overcup oak	*Q. lyrata*	High	517
Sugarberry	*Celtis laevigata*	Low	513
Sweetgum	*Liquidambar styraciflua*	High	468
American elm	*Ulmus americana*	High	376
Water hickory	*Carya aquatica*	Not rated	273
Swamp chestnut oak	*Q. michauxii*	Medium	205
Eastern cottonwood	*Populus deltoides*	Low	129
Cherrybark oak	*Q. pagoda*	Medium	109
Water oak	*Q. nigra*	High	83
Other merchantable species			332
Total			5,057

while green ash, sugarberry, sweetgum, and American elm were the most numerous non-oak species. The data set also included 332 trees representing 16 different species that had fewer than 75 observations each. The more abundant of these species were common persimmon (*Diospyros virginiana*), black tupelo (*Nyssa sylvatica*), honeylocust (*Gleditsia triacanthos*), and Shumard oak (*Q. shumardii*).

Influence of Species on Production of Epicormic Branches

Species is an important characteristic that influences production of epicormic branches on hardwood trees in undisturbed stands. It is the only one of the five characteristics that is not related directly to the degree of stress experienced by a tree. Meadows (1995) hypothesized that each hardwood species can be associated with a general range of susceptibility to production of epicormic branches. For example, some species may be characterized by an inherently low level of susceptibility, whereas other species may be characterized by an inherently high level of susceptibility. Trees of any given species may exhibit a range of variability within the inherent level of susceptibility associated with that species. It is within this range of variability that stress exerts its influence on any particular tree.

Species varied considerably in their tendency to produce epicormic branches, even in undisturbed stand conditions (fig. 1). Among the 12 species evaluated here, epicormic branches were generally more numerous on oak species than on non-oak species. This general observation appears to be true not only for the total number of epicormic branches on the butt log, but also for the number of both large and small epicormic branches, as defined by Rast and others (1973). However, cherrybark oak had very few epicormic branches and appears to be an exception to this general observation. Based on published information (Burns and Honkala 1990, Putnam and others 1960) and on personal experience and observations, Meadows (1995) qualitatively rated most oaks, including Nuttall, willow, water (*Q. nigra*), and overcup oaks, as highly susceptible to the production of epicormic branches, but rated both cherrybark and swamp chestnut oaks as only moderately susceptible (table 2). To date, our results generally support these ratings, with the exception that we found swamp chestnut oak to be more than moderately susceptible, in contrast with Meadows (1995).

Among the six non-oak species shown in figure 1, green ash, water hickory, and eastern cottonwood had very few epicormic branches on the butt log. The average number of epicormic branches was moderately low on sugarberry and moderately high on sweetgum and American elm. Our results generally agree with the qualitative ratings published by Meadows (1995), who classified green ash, sugarberry, and eastern cottonwood as slightly susceptible to the production of epicormic branches, but classified sweetgum and elms as highly susceptible (table 2). Meadows (1995) did not classify water hickory, but rated a similar species, pecan (*Carya illinoinensis*), as slightly susceptible. Our results tentatively suggest that sugarberry may need to be reclassified as moderately susceptible.

Species also varied substantially in the relative proportions of large and small epicormic branches found on the butt log (fig. 1). Large epicormic branches accounted for roughly 50 percent of the total number of epicormic branches on Nuttall,

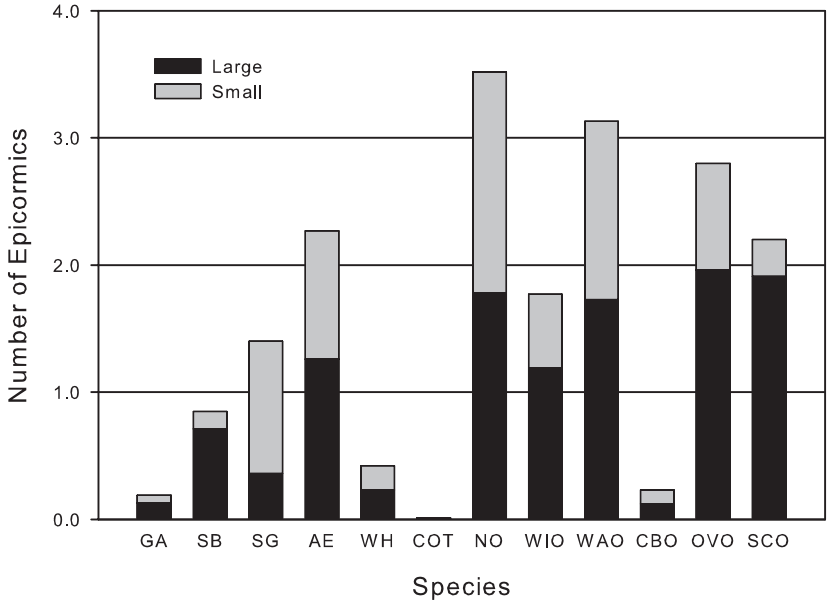

Figure 1—Mean number of large and small epicormic branches on the butt log, by species. Species include green ash (GA), sugarberry (SB), sweetgum (SG), American elm (AE), water hickory (WH), eastern cottonwood (COT), Nuttall oak (NO), willow oak (WIO), water oak (WAO), cherrybark oak (CBO), overcup oak (OVO), and swamp chestnut oak (SCO). Sample sizes are listed in table 2.

water, and cherrybark oaks, as well as on American elm and water hickory. The proportion of large epicormic branches was >50 percent on willow, overcup, and swamp chestnut oaks, as well as on green ash and sugarberry. In contrast, the proportion of large epicormic branches on the butt log of sweetgum was <50 percent.

The relative proportions of large and small epicormic branches observed among these 12 species may be indicative of the expected longevity of the epicormic branches produced on the butt log. If so, species with high relative proportions of large epicormic branches, such as some oaks, may tend to produce epicormic branches that are somewhat persistent. In these species, new epicormic branches produced on the bole tend to survive and persist year after year, even after they have grown large enough to be classified as large epicormic branches. In contrast, species with low relative proportions of large epicormic branches, such as sweetgum, may tend to produce epicormic branches that are more ephemeral. In these species, new epicormic branches produced on the bole tend to survive only a short time and generally die before they grow large enough to be classified as large epicormic branches. More research is needed to evaluate these proposed scenarios.

Our results support the hypothesis proposed by Meadows (1995) that species is an important characteristic affecting the susceptibility of hardwood trees to production of epicormic branches, at least in the 12 species evaluated in this paper. Data collected on the other 16 merchantable species in this study (those with fewer than 75 observations each) were insufficient to permit sound discussion.

Influence of Stand Density on Production of Epicormic Branches

Stand density is another important characteristic that we believe influences production of epicormic branches on hardwood trees in undisturbed stands. There is a general relationship between stand density and the degree of stress experienced by individual trees in a stand. Low to moderate levels of stand density generally do not cause undue stress in most trees in undisturbed stands. However, high stand density produces overcrowded conditions, which lead to intense competition among trees for limited resources. All trees in dense stands experience at least some degree of stress due to overcrowding. In closed stands, large trees have a competitive advantage over small trees, such that the degree of stress suffered as a result of high stand density is typically higher among small trees than among large trees. Meadows (1995) proposed that increased stress lowers tree health and leads to the production of epicormic branches along the boles of hardwood trees, even in undisturbed stands. To evaluate the concept that high stand density in undisturbed stands leads to increased competition, increased stress, reduced tree health, and a subsequent increase in the production of epicormic branches, we compared the average number of large epicormic branches on trees across a wide range of basal area classes, not only for trees of all merchantable species combined but also for trees of four representative species: green ash, sweetgum, Nuttall oak, and willow oak (fig. 2).

Stand basal area had little or no effect on the number of large epicormic branches on trees in undisturbed stands (fig. 2). When averaged across trees of all merchantable species, the number of large epicormic branches remained

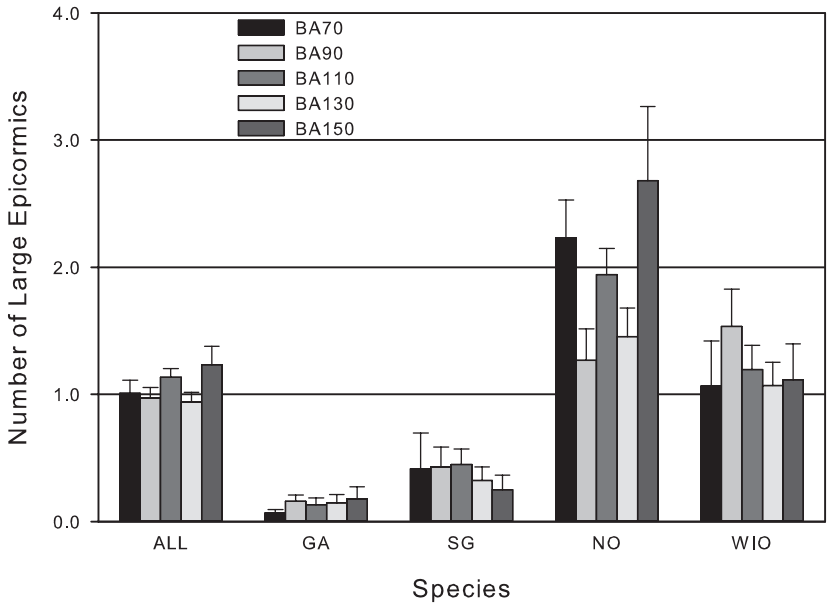

Figure 2—Mean (±SE) number of large epicormic branches on the butt log, by basal area class (square foot per acre), for all merchantable species combined (ALL) and for green ash (GA), sweetgum (SG), Nuttall oak (NO), and willow oak (WIO). Sample sizes are listed in table 2.

nearly constant across the range of basal area classes. We observed the same nearly level pattern in green ash, sweetgum, and, to a lesser extent, willow oak. Nuttall oak, on the other hand, produced more large epicormic branches under conditions of both low stand density and high stand density than it did under conditions of moderate stand density. The number of large epicormic branches on Nuttall oak not only fluctuated across the range of basal area classes, it also varied within each basal area class, as evidenced by relatively high standard errors. Even though it appears that stand density influences production of epicormic branches on Nuttall oak, the degree of variability both within and among basal area classes limits our ability to conclusively assess the role played by stand density in this species.

Results obtained so far in this study tend to disagree with the notion advanced by Meadows (1995) that high stand density imposes stress severe enough to promote the production of epicormic branches on hardwood trees in undisturbed stands, even though data collected on Nuttall oak appear to support the concept. More data clearly are needed to fully evaluate the possible influence that stand density may have on production of epicormic branches in various hardwood species.

Influence of Tree Size on Production of Epicormic Branches

Another important characteristic that may influence production of epicormic branches on hardwood trees in undisturbed stands is tree size. We selected d.b.h. to indicate tree size because it can be measured easily, quickly, and accurately. Other variables, such as crown size and tree height, also indicate tree size, but these traits are difficult and time consuming to measure accurately and reliably. In even-aged stands, tree size may be an indicator of the degree of stress experienced by a tree. In most cases, small-diameter trees are at a competitive disadvantage to large-diameter trees. They often are suppressed by larger trees and therefore have limited growing space and limited access to resources necessary for survival and growth. Meadows (1995) hypothesized that, because small-diameter trees suffer from suppression in even-aged stands, they experience more stress and are less healthy than large-diameter trees. If so, we expect to find more epicormic branches on small-diameter trees than on large-diameter trees, even in undisturbed stands. To assess this idea, we compared the average number of large epicormic branches on trees across four tree size classes: (1) poletimber, 5.5 to 11.9 inches d.b.h.; (2) small sawtimber, 12.0 to 17.9 inches d.b.h.; (3) medium sawtimber, 18.0 to 23.9 inches d.b.h.; and (4) large sawtimber, 24.0 inches d.b.h. and larger. Comparisons among these four tree size classes were made for trees of all merchantable species combined, and for trees of four representative species: green ash, sweetgum, Nuttall oak, and willow oak (fig. 3).

When averaged across all merchantable species, the number of large epicormic branches decreased uniformly with increasing tree size (fig. 3). A similar pattern was observed in Nuttall oak. In both sweetgum and willow oak, the number of large epicormic branches decreased sharply with increasing tree size. For both species, there were many more large epicormic branches on trees in the poletimber class than on trees in the three sawtimber classes. Small-diameter willow oak trees were especially prone to production of epicormic

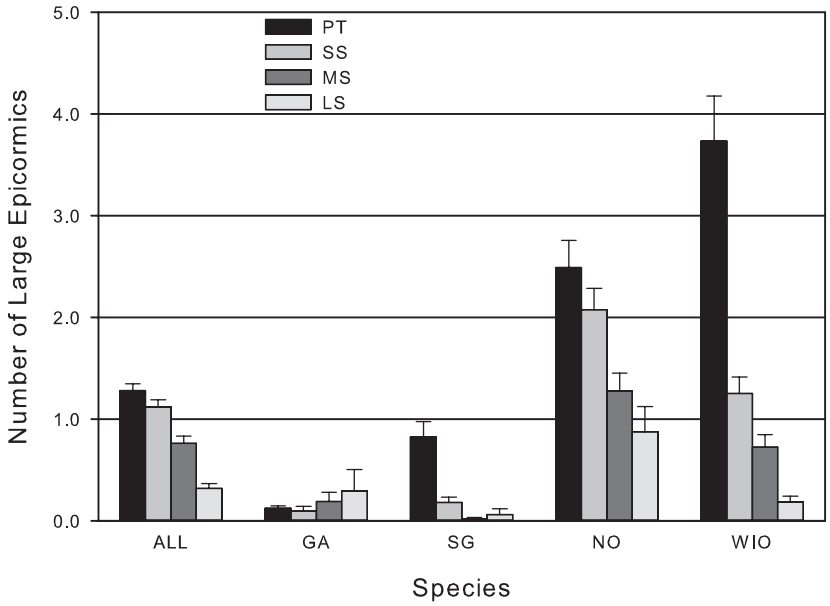

Figure 3—Mean (±SE) number of large epicormic branches on the butt log, by tree size class, for all merchantable species combined (ALL) and for green ash (GA), sweetgum (SG), Nuttall oak (NO), and willow oak (WIO). Tree size classes include poletimber (PT), small sawtimber (SS), medium sawtimber (MS), and large sawtimber (LS). Sample sizes are listed in table 2.

branches, probably an indication that these trees were highly stressed and unhealthy. In contrast, green ash had very few large epicormic branches regardless of tree size.

Our results strongly support the contention proposed by Meadows (1995) that, in undisturbed even-aged stands, small-diameter trees are generally less healthy than large-diameter trees and are therefore more susceptible to the production of epicormic branches. The high level of stress experienced by small-diameter trees is the primary factor that stimulates increased production of epicormic branches. Similarly, the low level of stress experienced by large-diameter trees is insufficient to promote production of epicormic branches.

Influence of Crown Class on Production of Epicormic Branches

Crown class is an extremely important characteristic that influences production of epicormic branches on hardwood trees in undisturbed stands. Crown class is widely recognized as a reliable indicator of the degree of stress experienced by trees in even-aged stands. In general, trees in the dominant and codominant crown classes have large, healthy crowns that promote vigorous tree growth. The level of stress experienced by dominant and codominant trees is generally low. In contrast, trees in the intermediate and overtopped crown classes typically have small, weak crowns that are able to support only minimal tree growth. The level of stress experienced by intermediate and overtopped trees is generally high. Meadows (1995) asserted that healthy trees are much less likely to produce epicormic branches than are unhealthy trees. If so, we expect to find more epicormic branches on trees in the intermediate and overtopped crown classes than on trees in the dominant and codominant crown classes. To evaluate this concept, we compared the average number of epicormic branches on trees across the four crown classes. Initial comparisons were made for both large and small epicormic branches on trees of all merchantable species combined (fig. 4).

When averaged across all merchantable species, the total number of epicormic branches and the average numbers of both large and small epicormic branches clearly increased across the spectrum of crown classes from dominant trees to overtopped trees (fig. 4). Both large and small epicormic branches were especially numerous on overtopped trees. The proportion of large epicormic branches, relative to the total number, remained fairly constant across crown classes and ranged from 52 percent for codominant trees to 63 to 64 percent for trees in the other three crown classes.

To investigate the influence of crown class on production of epicormic branches in more detail, we separated and summarized the data for six commercially important timber species: green ash, sweetgum, and Nuttall, willow, water, and cherrybark oaks (fig. 5). Because there were insufficient data within each species to allow an adequate evaluation across each of the four crown classes, we combined data into two classes: (1) dominant and codominant (D/CD), or upper crown-class trees; and (2) intermediate and overtopped (INT/OT), or lower crown-class trees. We also limited our evaluations to the number of large epicormic branches only. Similar to the trend observed when data were averaged across all merchantable species (fig. 4), the average number of large epicormic branches was substantially greater on lower crown-class trees than on upper crown-class trees of all species except green ash, which had few epicormic branches regardless of crown class (fig. 5). The largest differences

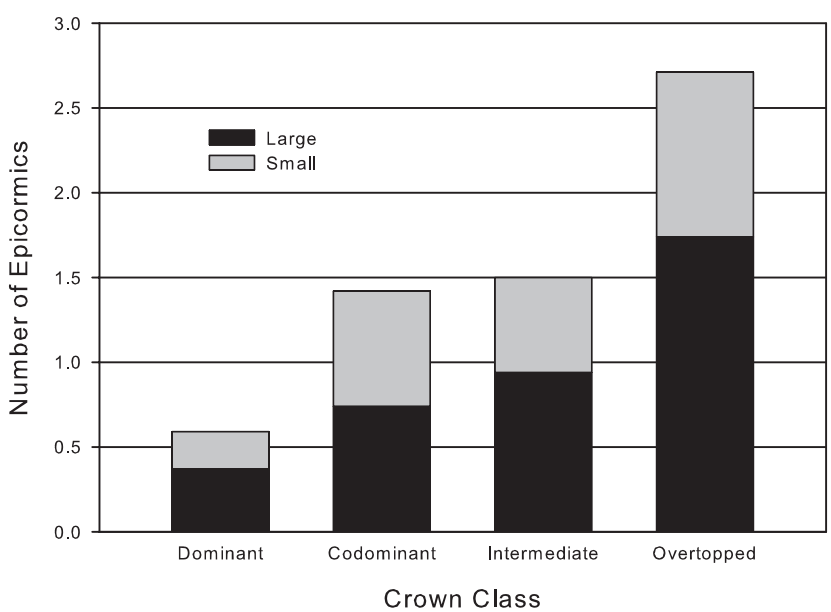

Figure 4—Mean number of large and small epicormic branches on the butt log, by crown class, for all merchantable species combined. Crown classes include dominant (n = 458), codominant (n = 1,651), intermediate (n = 1,606), and overtopped (n = 1,342).

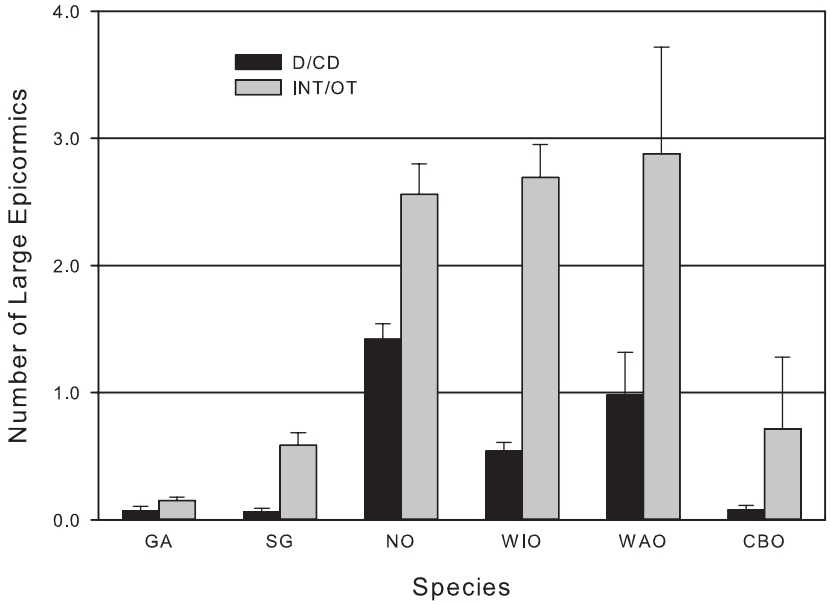

Figure 5—Mean (±SE) number of large epicormic branches on the butt log, by combined crown class, for six commercially important tree species: green ash (GA), sweetgum (SG), Nuttall oak (NO), willow oak (WIO), water oak (WAO), and cherrybark oak (CBO). Combined crown classes include dominant and codominant (D/CD) and intermediate and overtopped (INT/OT). Sample sizes are listed in table 2.

between lower crown-class trees and upper crown-class trees were found on willow and water oaks and, to a lesser degree, on Nuttall oak. An average difference of less than one epicormic branch was observed on both cherrybark oak and sweetgum.

The trends illustrated in figures 4 and 5 clearly demonstrate that the degree of stress experienced by an individual tree, as reflected by its crown class, plays a major role in the production of epicormic branches along the boles of trees of most hardwood species, with the exception of those species, such as green ash, that inherently produce few epicormic branches. As a general rule for most hardwood trees in undisturbed stands, as the level of stress experienced by a tree increases, the propensity of that tree to produce epicormic branches also increases. Our results strongly support the hypothesis advanced by Meadows (1995) that healthy trees under low levels of stress are much less likely to produce epicormic branches than are unhealthy trees under high levels of stress. Our results further demonstrate that crown class accurately and reliably reflects the level of stress experienced by an individual tree. Consequently, crown class appears to be a strong indicator of the propensity of an individual tree to produce epicormic branches in an undisturbed stand.

PRELIMINARY FINDINGS

Results presented in this paper are preliminary. We plan to collect data from perhaps as many as 20 additional sample stands over the next few years. Data from all undisturbed stands eventually will be combined with data from thinned hardwood stands to (1) revise the Meadows (1995) ratings of southern bottomland hardwood species for susceptibility to production of epicormic branches, and (2) develop models to predict epicormic branch production as a function of various tree, stand, and site characteristics in both unthinned and thinned stands. Separate models will be developed for each species and may be developed for different stand conditions and different site types. Our findings to date are:

1. Hardwood species vary considerably in their propensity to produce epicormic branches. Most oak species, except cherrybark oak, produce several epicormic branches, even in undisturbed stands, whereas green ash produces few epicormic branches.
2. Stand density appears to exert little influence on production of epicormic branches by most hardwood species in undisturbed stands. Preliminary results indicate that production of epicormic branches on Nuttall oak, however, may be somewhat sensitive to stand density.
3. In undisturbed, even-aged hardwood stands, production of epicormic branches generally decreases with increasing tree diameter. Small-diameter willow oaks are especially susceptible to the production of epicormic branches.
4. Crown class strongly influences production of epicormic branches in undisturbed hardwood stands. In general, upper crown-class trees produce few epicormic branches, whereas lower crown-class trees produce many epicormic branches. The influence of crown class is most pronounced in trees of highly susceptible

species, such as willow oak, and is least pronounced in trees of resistant species, such as green ash.

5. Our preliminary evaluations support an earlier hypothesis by Meadows (1995) that, in undisturbed hardwood stands, the number of epicormic branches present on the butt log of any given tree is primarily a function of species and the degree of stress experienced by that tree. Crown class appears to be a strong indicator of that degree of stress.

ACKNOWLEDGMENTS

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LITERATURE CITED

Baker, J.B.; Broadfoot, W.M. 1979. A practical field method of site evaluation for commercially important southern hardwoods. Gen. Tech. Rep. SO-26. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 51 p.

Brown, C.L.; Kormanik, P.P. 1970. The influence of stand disturbance on epicormic branching. In: Hansbrough, T., ed. Silviculture and management of southern hardwoods: 19th annual forestry symposium. Baton Rouge, LA: Louisiana State University Press: 103–112.

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

Carpenter, R.D.; Sonderman, D.L.; Rast, E.D.; Jones, M.J. 1989. Defects in hardwood timber. Agric. Handb. 678. Washington, DC: U.S. Department of Agriculture. 88 p.

Erdmann, G.G.; Peterson, R.M., Jr.; Oberg, R.R. 1985. Crown releasing of red maple poles to shorten high-quality saw log rotations. Canadian Journal of Forest Research. 15: 694–700.

Hanks, L.F.; Gammon, G.L.; Brisbin, R.L.; Rast, E.D. 1980. Hardwood log grades and lumber grade yields for factory lumber logs. Res. Pap. NE-468. Broomall, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 92 p.

Hedlund, A. 1964. Epicormic branching in north Louisiana Delta. Res. Note SO-8. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 3 p.

Meadows, J.S. 1993. Logging damage to residual trees following partial cutting in a green ash-sugarberry stand in the Mississippi Delta. 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: 248–260.

Meadows, J.S. 1995. Epicormic branches and lumber grade of bottomland oak. In: Lowery, G.; Meyer, D., eds. Advances in hardwood utilization: following profitability from the woods through rough dimension: Proceedings of the twenty-third annual hardwood symposium. [Memphis, TN]: National Hardwood Lumber Association: 19–25.

Meadows, J.S.; Burkhardt, E.C. 2001. Epicormic branches affect lumber grade and value in willow oak. Southern Journal of Applied Forestry. 25(3): 136–141.

Meadows, J.S.; Skojac, D.A., Jr. 2008. A new tree classification system for southern hardwoods. Southern Journal of Applied Forestry. 32(2): 69–79.

Putnam, J.A.; Furnival, G.M.; McKnight, J.S. 1960. Management and inventory of southern hardwoods. Agric. Handb. 181. Washington, DC: U.S. Department of Agriculture. 102 p.

Rast, E.D.; Sonderman, D.L.; Gammon, G.L. 1973. A guide to hardwood log grading. Revised. Gen. Tech. Rep. NE-1. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 31 p.

Stubbs, J. 1986. Hardwood epicormic branching—small knots but large losses. Southern Journal of Applied Forestry. 10(4): 217–220.

OVERSTORY TREE STATUS FOLLOWING THINNING AND BURNING TREATMENTS IN MIXED PINE-HARDWOOD STANDS ON THE WILLIAM B. BANKHEAD NATIONAL FOREST, ALABAMA

Callie Jo Schweitzer and Yong Wang[1]

Abstract—Prescribed burning and thinning are intermediate stand treatments whose consequences when applied in mixed pine-hardwood stands are unknown. The William B. Bankhead National Forest in northcentral Alabama has undertaken these two options to move unmanaged, 20- to 50-year-old loblolly pine (*Pinus taeda* L.) plantations towards upland hardwood-dominated stands. Our study of their management employs a randomized complete block design with a three by three factorial treatment arrangement and four replications of each treatment. Treatments are three residual basal areas (50 square feet per acre, 75 square feet per acre, and an untreated control) with three burn frequencies (frequent burns once every 3 to 5 years, infrequent burns once every 8 to 10 years, and an unburned control). To date, only one burn cycle has been completed (all burn units burned once). Stands were thinned from June through December, and burned in December through March. We measured the overstory response to these treatments 1 year after implementation. Pretreatment basal area ranged between 952 and 163 square feet per acre; harvesting reduced basal areas to 51 square feet per acre and 68 square feet per acre in the 50 and 75 retention treatments, respectively. The percentage of total basal area that was loblolly pine remained relatively unchanged posttreatment (47.4 percent pretreatment and 46.5 percent posttreatment). Oak basal area (*Quercus alba* L., *Q. prinus* L., *Q. falcata* Michx., *Q. rubra* L., *Q. velutina* Lam., and *Q. coccinea* Münchh.) increased slightly posttreatment, from 7 percent of total stand basal area to 10 percent. Harvested stands had a 30-percent reduction in percent canopy cover, and light penetration through the canopies ranged from 5 to 25 percent pretreatment to 29 to 60 percent posttreatment. The cool, slow moving burns had no discernable effect on the overstory trees.

INTRODUCTION

Decline and death of southern yellow pines (*Pinus* spp.) due to the southern pine beetle (*Dendroctonus frontalis* Zimmermann) has been detrimental to the health of portions of the William B. Bankhead National Forest (BNF), located in northcentral Alabama. On the BNF, the district ranger and staff have worked with a formal forest liaison board to gain interest and acceptance of the new Forest Health and Restoration Project (FHRP) (U.S. Department of Agriculture Forest Service 2003) to be implemented under traditional Forest Service authorities. Involved parties wished to have a scientific assessment of the effectiveness of the BNF's current management techniques, as detailed in their most recent Land and Resource Management Plan. A goal of this management was to initiate a series of processes that would move mixed hardwood-pine stands towards those dominated by upland hardwood species. Generally, a combination of various techniques (herbicide treatments, prescribed fire, thinning, and shelterwood regeneration harvests) applied at intervals (depending on vegetation response and site characteristics) may have the most profound effect on community structure and composition (Franklin and others 2003, Loewenstein and Davidson 2002, Lorimer 1992, Nowak and others 2002). However, the interactions of these techniques are largely unknown.

Prescribed burning and thinning are intermediate treatments that when applied in oak (*Quercus* spp.)-dominated stands can improve forest health, increase the abundance and size of oak regeneration, and control species composition. This large-scale study was designed to assess the effectiveness of silvicultural techniques used to restore and maintain upland oak-dominated ecosystems in the northern portion of the BNF. Our objectives for this paper are to summarize the first-year response of the overstory tree species, and to explore changes in canopy composition, structure, and light penetration induced by thinning and burning.

METHODS

Study Area

The BNF, established by proclamation in 1914, has a long history of repeated logging and of soil erosion caused by poor farming practices during the Depression era. The 180,000-acre BNF is in the Strongly Dissected Plateau subregion of the Southern Cumberland Plateau, within the Southern Appalachian Highlands (Smalley 1979). Base age 50 site indices for loblolly pine (*P. taeda* L.); red oaks (*Q. rubra* L., *Q. velutina* Lam., *Q. coccinea* Münchh., and *Q. falcata* Michx.); and white oaks (*Q. alba* L. and *Q. prinus* L.) are 75 feet, 65 feet, and 65 feet, respectively (Smalley 1979).

Study Design

The BNF study employed a randomized complete block design with a three by three factorial treatment arrangement in four blocks. The treatments were three residual basal area (BA) treatments (heavy thin leaving 50 square feet per acre, light thin leaving 75 square feet per acre, and an untreated control) with three prescribed burn frequencies (frequent burns every 3 to 5 years, infrequent burns every 8 to 10 years, and an unburned control; table 1). Each treatment was replicated 4 times, for a total of 36 treatment units. Treatments are representative of management practices described in the BNF's FHRP for restoring oak forests and woodlands (U.S. Department of Agriculture Forest Service 2003).

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Table 1—Thinning and burning treatment implementation schedule (month-yr), by block, for study stands on the William B. Bankhead National Forest, Alabama

Treatment	Harvest goal	Harvest date				Burn return frequency	Burn date			
		Block 1	Block 2	Block 3	Block 4		Block 1	Block 2	Block 3	Block 4
number	*feet²/acre*					*years*				
1	0					0				
2	0					10	Feb-06	Dec-06	Jan-07	Mar-08
3	0					3	Feb-06	Jan-07	Mar-07	Jan-08
4	50	Aug-05	Jun-06	Jul-06	Jun-07	0				
5	75	Aug-05	Jun-06	Jul-06	Nov-07	0				
6	50	Aug-05	Aug-06	Dec-06	Sep-07	3	Jan-06	Jan-07	Jan-07	Feb-08
7	75	Aug-05	Dec-06	Dec-06	Sep-07	3	Jan-06	Jan-07	Mar-07	Feb-08
8	50	Sep-05	Dec-06	Aug-06	Oct-07	10	Mar-06	Dec-06	Jan-07	Mar-08
9	75	Sep-05	Dec-06	Jul-06	Oct-07	10	Mar-06	Dec-06	Jan-07	Mar-08

Criteria for stand selection were based on species composition, stand size, and stand age. Treatment units for the study were located on upland sites currently supporting 20- to 50-year-old loblolly pine plantations with a pronounced hardwood component in the understory. Treatment units were at least 22 acres in size with BAs ranging from 95 to 163 square feet per acre.

Commercial thinning was conducted by marking from below smaller trees or trees that appeared diseased or damaged. 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 (December through March) using backing fires and strip headfires to ensure that only surface fire occurred. See table 1 for treatment implementation schedule.

Field Techniques

We established five 0.2-acre vegetation measurement plots in each treatment unit (stand) and measured plots prior to and one growing season after treatment implementation. All plot centers were permanently marked with rebar, flagging, and GPS coordinates. We permanently tagged all trees >5.5 inches diameter at breast height (d.b.h.) with aluminum tags. Tree distance and azimuth to plot center were recorded. We measured and recorded tree species; d.b.h. (diameter tape, to the nearest 0.1 inch); canopy cover (hand-held spherical densiometer, five measurements at each plot, one 10 feet from plot center in each cardinal direction and one at plot center); and damage (enumerated number of epicormic branches, recorded number of wounds on the lower bole and measured length and height of each wound with a ruler to the nearest inch). Photosynthetically active radiation (PAR) was measured using two synchronized ceptometers (AccuPAR LP-80, Decagon Devices, Inc., Pullman, WA). One ceptometer was placed in full sunlight and the second ceptometer was used to record light in each stand along predesignated transects. Additional canopy characteristics were assessed using hemispherical photographs taken at plot centers and analyzed using HemiView Version 2.1 (Delta-T Devices, Cambridge, United Kingdom). A global site factor (GSF) was obtained from analysis of hemispherical photographs taken at each vegetation plot center. All data were gathered pretreatment and posttreatment.

We sampled fuel loading using line transects and employed electronic recording devices and temperature sensitive paints to quantify fire behavior during burns (Clark and Schweitzer this proceedings). We revisited plots once near the end of the growing season to document recruitment and tree growth.

Statistical Analysis

We used analysis of variance (ANOVA) to test differences and interactions at the stand level. All analyses were performed in SAS (2000). ANOVA was performed using time of treatment implementation as block, harvest intensity, and prescribed burn interval as treatments, and the mean of vegetation plot measurements by stand as replicates. If no interaction existed, treatment effects were assessed separately using Tukey's studentized range test on main effect means using a $P<0.05$ level of significance.

RESULTS

Overstory Composition and Structure

We measured 10,448 trees with diameters that ranged from 5.6 to 24.3 inches. Twenty-three different species were identified in these plots. There were three pine species,

dominated by loblolly, with a smaller portion of Virginia (*P. virginiana* Mill.) and shortleaf (*P. echinata* Mill.). Other species included upland oaks (chestnut oak, white oak, northern red oak, scarlet oak, black oak, and southern red oak); yellow-poplar (*Liriodendron tulipifera* L.); red maple (*Acer rubrum* L.); and black cherry (*Prunus serotina* Ehrh.). We found no significant differences for BA ($P = 0.3116$) and stems per acre (SPA) ($P = 0.5801$) among the nine treatments prior to treatment implementation. BA in the study stands ranged from 95 to 163 square feet per acre (standard deviation 16, $n = 36$), and SPA were 191 to 379 stems per acre (std 48, $n = 36$) (see tables 2 and 3). Pretreatment stand BA was dominated by loblolly pine, which accounted for 87 percent of the BA and 85 percent of the SPA. Data are presented by block to assist research partners who are engaged in other aspects of this large study. The percent of total BA and SPA for upland oaks was 7 and 8, for yellow-poplar 6 and 5, and for both red maple and black cherry 2 and 3, respectively.

The prescribed burning had no effect on overstory composition and structure (species diversity as a percentage of total BA) ($P = 0.6697$), and there were no burn by thinning interactions ($P = 0.7040$). Only one fire has been implemented to date. Fires burned 70 percent of our plots, had a mean maximum temperature of 220 °F, spread at a rate of 10 feet per minute, and were not intense (heat index, the sum of temperature above 90 °F averaged 21,000 °F).[2]

The thinning treatments resulted in three significantly different BA and SPA groups. Unthinned stands had 132 square feet per acre of BA and 286 SPA, the light thinned had 68 square feet per acre and 113 SPA, and the heavy thinned stands had 51 square feet per acre and 84 SPA. Although the light thinned stands resulted in 7 square feet per acre less BA than was the goal, the overall objective of creating three distinct residual BA groups was achieved. The majority of the reduction in BA came from the removal of pine; in the light thinning, pine BA was reduced from 118 to 85 square feet per acre, and in the heavy thinning from 113 to 40 square feet per acre. Some hardwood BA was also affected by the thinning treatments. For the upland oaks, light thinning reduced total BA from 8 to 7 square feet per acre and heavy thinning from 9 to 7 square feet per acre. Yellow-poplar BA was similarly reduced, from 8 to 4 square feet per acre in light thinned stands and from 8 to 2 square feet per acre in heavy thinned stands.

These young stands contained predominately smaller diameter trees, and thinning targeted those trees in the 6- to 12-inch diameter classes. There were few tallied trees of any species with a d.b.h. >18 inches for the 24 stands that were thinned. For both the heavy and light thinned treatments, pine SPA in the 6-inch d.b.h. class was reduced 90 percent and 6-inch oak by 38 percent. There was a 78-percent reduction in 8-inch pine and a 65-percent reduction in 10-inch pines (fig. 1). Oak in the 8-inch class declined 17 percent and 10-inch oak was reduced by 42 percent (fig. 2).

Damage

In the thinned stands, 12 percent of the residual trees had damage attributable to logging activities. There was no

Table 2—Basal area for trees >5.5 inches d.b.h., pre- and posttreatment averages by block, for study stands on the William B. Bankhead National Forest, Alabama

	Pretreatment					Posttreatment				
Treatment	Block 1	Block 2	Block 3	Block 4	Block average	Block 1	Block 2	Block 3	Block 4	Block average
	square feet per acre									
1	124	128	128	148	132	128	135	132	152	137
2	109	113	131	139	123	108	125	133	145	128
3	144	95	123	125	122	151	106	132	132	130
4	108	131	148	140	132	48	60	50	56	53
5	116	137	125	153	133	55	73	67	73	67
6	111	128	124	147	128	51	51	41	55	50
7	102	134	141	147	131	60	58	63	75	64
8	125	150	146	107	132	55	51	49	42	49
9	112	163	137	146	139	58	62	88	78	72

[2] Stacy Clark. 2009. Unpublished data. Research Forester, U.S. Forest Service, Southern Research Station, P.O. Box 1568, Normal, AL 35762.

Table 3—Stems per acre of trees >5.5 inches d.b.h., pre- and posttreatment averages by block, for study stands on the William B. Bankhead National Forest, Alabama

	Pretreatment					Posttreatment				
Treatment	Block 1	Block 2	Block 3	Block 4	Block average	Block 1	Block 2	Block 3	Block 4	Block average
	stems per acre									
1	299	274	237	251	265	299	274	235	247	264
2	241	268	266	347	281	228	268	259	344	275
3	325	280	372	307	321	323	280	372	305	320
4	191	313	360	322	297	66	81	91	103	85
5	218	237	253	379	272	88	93	104	137	106
6	234	280	261	335	278	88	80	70	95	83
7	218	329	297	328	293	104	92	114	119	107
8	303	306	328	262	300	103	72	84	82	85
9	328	365	237	295	306	133	112	129	126	125

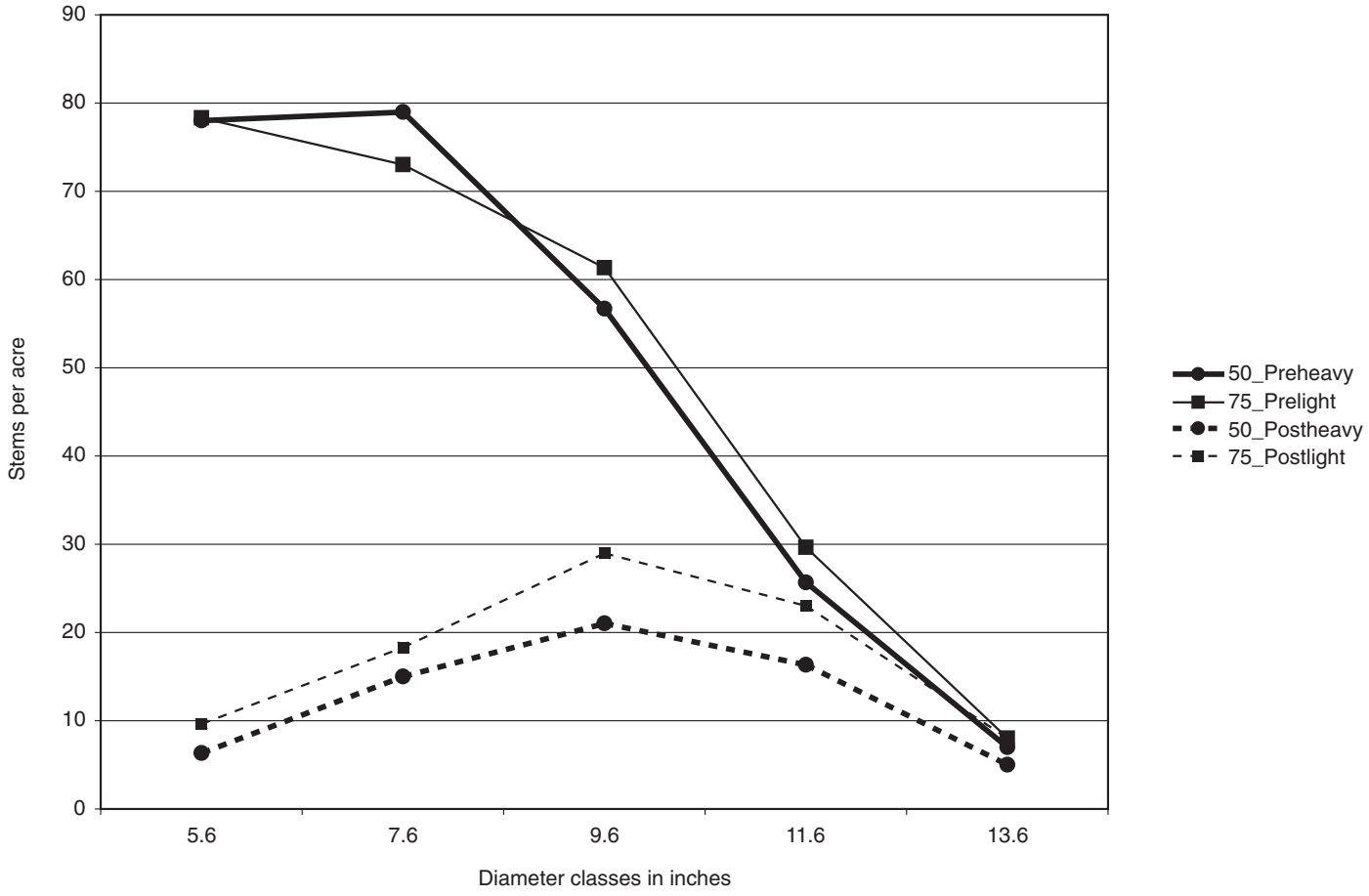

Figure 1—Pre- and posttreatment pine stems per acre for light (75 square feet per acre retention goal) and heavy (50 square feet per acre) thinning treatments on study stands in the William B. Bankhead National Forest, Alabama.

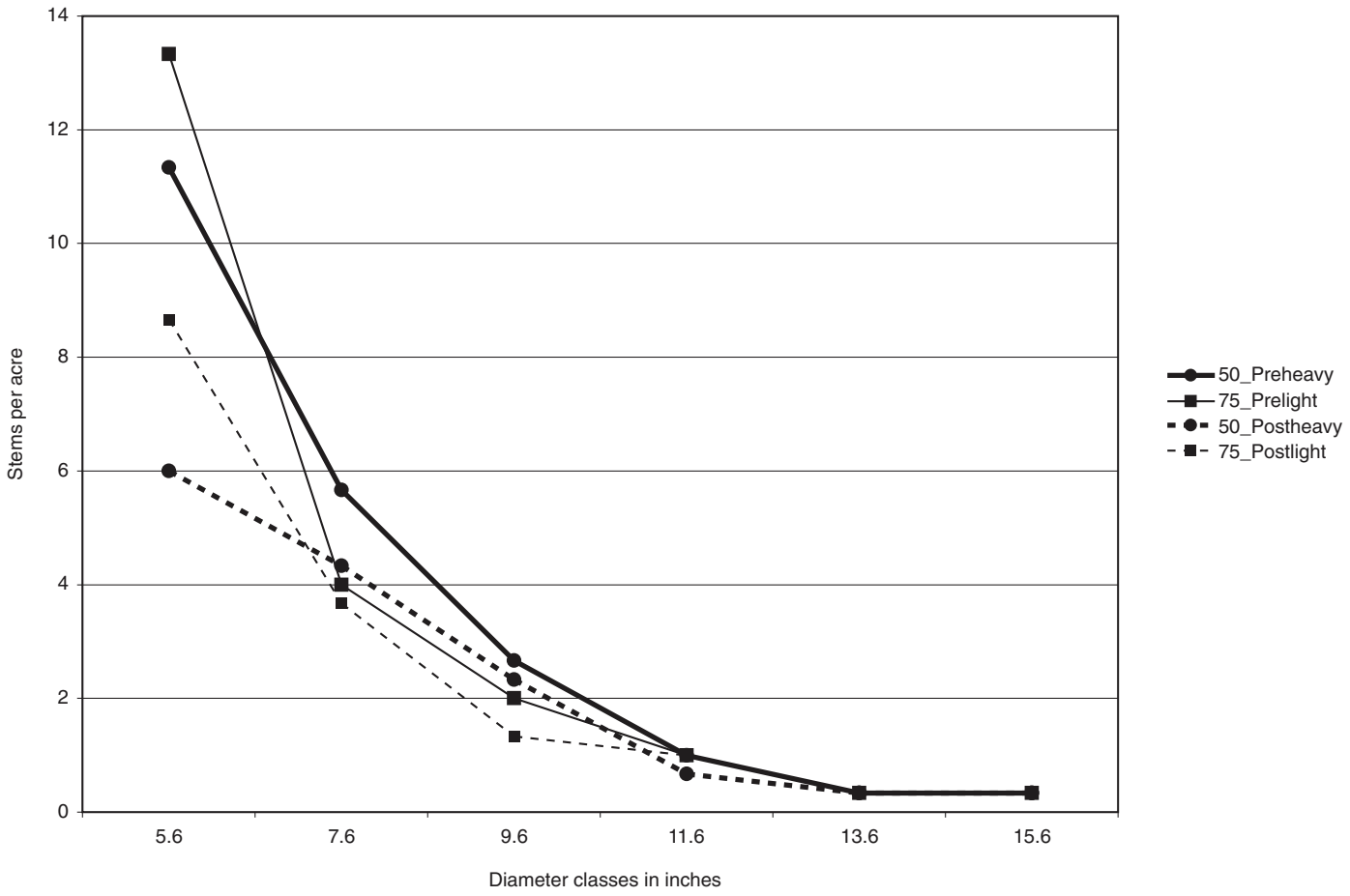

Figure 2—Pre- and posttreatment oak stems per acre for light (75 square feet per acre retention goal) and heavy (50 square feet per acre) thinning treatments on study stands in the William B. Bankhead National Forest, Alabama.

difference in the frequency of damage between the light and heavy thinned stands, with the light thinned stands having an average of 12 wounded residual SPA and the heavy having 10 wounded residual SPA ($P = 0.1316$). Epicormic branching was assessed for the residual hardwood species only. There was no significant difference among the three harvest treatments for the number of residual hardwood SPA that displayed epicormic branching ($P = 0.6995$). In the light thinned treatment, 24 percent of the residuals had some epicormic branching, which averaged 21 branches per tree. In the heavy thinned treatment, 41 percent of hardwood residuals epicormically branched, averaging 18 branches per tree. On average, 57 percent of hardwood stumps sprouted after 1 growing season; sprouting stumps averaged 11 sprouts with an average height of 3 feet.

Canopy Cover

We estimated the change in the light environment in the understory using three methods. Densiometer-derived canopy cover estimates, averaging 93 percent, did not differ among the stands prior to harvest ($P = 0.3427$). Postharvest cover for the control stands (93 percent) was significantly greater than the light (68 percent) and heavy thinned (66 percent) stands (Tukey $P = 0.0016$). The amount of PAR penetrating these canopies showed a similar trend. Ceptometer data, representing the amount of full sun penetrating the canopy relative to readings obtained in the open, showed that PAR was similar for the light (43 percent of full sun) and heavy thinned stands (52 percent of full sun) posttreatment, and the treated stands differed from the control (9 percent of full sun) (Tukey $P < 0.0001$). The GSF also corresponds with the percentage of total PAR reaching a site relative to a site in the open. There were no GSF differences among treatments prior to harvest ($P = 0.0862$). Postharvest GSF was greater for the light (33 percent) and heavy (38 percent) thinned stands compared to the control (23 percent) (Tukey $P = 0.0039$).

DISCUSSION

Thinning was used in these stands to accelerate natural succession and move the stands towards a species composition dominated by upland hardwoods. Hardwood encroachment into these unmanaged pine plantations commenced following stand initiation 20 to 50 years ago. Additional management objectives included improving

productivity and forest health. Thinning can also increase stand structural diversity and plant and wildlife diversity (Maas-Hebner and others 2005). Congruent studies being carried out by faculty and graduate students at Alabama A&M University will be used to document how these habitat changes impact wildlife.

Intermediate stand treatments such as thinning, and as in this study, prescribed fire, focus their effects on the residual stand. That said, every entry into a hardwood stand should also consider the impact on reproduction. For this paper, we only report on the residual overstory composition and structure; we are documenting the midstory and reproduction response and will report on those elsewhere.

One growing season postfire may not have allowed sufficient time to document any effects on the survival of the residual overstory trees. Waldrop and others (2008) found that hardwood trees had a delayed response to fire, while Arthur and others (1998) found that overstory trees were not damaged by fire. Chiang and others (2008) reported significant fire-related tree mortality in treatments that had experienced prescribed fire; other prescribed fire studies have reported no effect on overstory trees due to the fire itself (Blake and Schuette 2000, Blankenship and Arthur 2006, Franklin and others 2003). Prescribed fire at such low intensities was not expected to have any discernable impact on overstory tree composition or structure in our study. Our fires consumed unconsolidated leaf litter and fine woody fuels (1- and 10-hour fuel) only. Fires in this study were not intense enough to cause scarring on trees 5.6 inches d.b.h. or greater. Other studies have shown scarring due to fire, highly correlated with fire intensity and tree size (Guyette and Stambaugh 2004, Hutchinson and others 2008, McEwan and others 2007, Smith and Sutherland 1999).

Canopy-reducing disturbances influence all tiers of forest structure. Some have shown that large reductions in canopy cover followed by prescribed fire facilitated regeneration of desirable *Quercus* species (Brose and others 1999, Ellsworth and McComb 2003). Changes in cover and light are transient and will alter vegetation response, including seedling recruitment (Chiang and others 2005, Iverson and others 2004). In this study, we did document a change in light penetrating the canopy. Harvesting disturbance which facilitated this increase in light will also stimulate the growth of preexisting saplings, and their growth may narrow the window for reproduction response (Domeke and others 2007). Changes in the size structure of these managed stands may limit the recruitment of light-demanding species as well.

CONCLUSION

Thinning combined with prescribed burning is being examined in mixed pine-hardwood stands in the Southern Cumberland Plateau. The process of moving these stands towards upland-hardwood dominated composition has commenced. The initial response documented in this project was the creation of three different residual BAs. Changes in overstory composition, canopy cover, and subsequent below canopy light levels resulted primarily from the thinning, as the low intensity burns had no discernable impact on the overstory stratum. These treatments will certainly have an impact on other stratum dynamics, such as midstory sprouting and seedling recruitment. The consequences of repeated fires on the dynamics of these stands will be explored.

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LITERATURE CITED

Arthur, M.A.; Paratley, R.D.; Blankenship, B.A. 1998. Single and repeated fires affect survival and regeneration of woody and herbaceous species in an oak-pine forest. Journal of the Torrey Botanical Society. 125(3): 225–236.

Blake, J.G.; Schuette, B. 2000. Restoration of an oak forest in east-central Missouri: Early effects of prescribed burning on woody vegetation. Forest Ecology and Management. 139: 109–126.

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.; 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.

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.

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 productivity of overstory trees in an oak-hickory ecosystem in southern Ohio. Forest Ecology and Management. 255: 1584–1594.

Domke, G.M.; Caspersen, J.P.; Jones, T.A. 2007. Light attenuation following selection harvesting in northern hardwood forests. Forest Ecology and Management. 239: 182–190.

Ellsworth, J.W.; McComb, B. 2003. Potential effects of passenger pigeon flocks on the structure and composition of presettlement forests of Eastern North America. Conservation Biology. 17: 1548–1558.

Franklin, S.B.; Robertson, P.A.; Fralish, J.S. 2003. Prescribed burning effects on upland *Quercus* forest structure and function. Forest Ecology and Management. 184: 315–335.

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.

Hutchinson, T.F.; Long, R.P.; Forest, R.D.; Sutherhland, 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.; Prasad, A.M.; Hutchinson, T.F. [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.

Loewenstein, E.F.; Davidson, K.R. 2002. Ecological restoration through silviculture—a savanna management demonstration area, Sinkin Experimental Forest, Missouri. In: Outcalt, K.W.; Outcalt, P.A.; Tucker, R.B., eds. 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: 490–494.

Lorimer, C.G. 1992. Causes of the oak regeneration problem. In: Loftis, D.; McGee, C.E., eds. Oak regeneration: serious problems, practical recommendations: Symposium proceedings. Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 14–39.

Maas-Hebner, K.G.; Emmingham, W.H.; Larson, D.J.; Chan, S.S. 2005. Establishment and growth of native hardwood and conifer seedlings underplanted in thinned Douglas fir stands. Forest Ecology and Management. 208: 331–345.

McEwan, R.W.; Hutchinson, T.F.; Ford, R.D.; McCarthy, B.C. 2007. An experimental evaluation of fire history reconstruction using dendrochronology in white oak (*Quercus alba*). Canadian Journal of Forest Research. 37: 806–816.

Nowak, S.; Kershaw, G.P.; Kershaw, L.J. 2002. Plant diversity and cover after wildfire on anthropogenically disturbed and undisturbed sites in subarctic upland *Picea mariana* forest. Arctic. 55: 269–280.

SAS Institute Inc. 2000. SAS software. Version 8. Cary, NC: SAS Institute Inc.

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.

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

U.S. Department of Agriculture 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. 353 p.

Waldrop, T.A.; Yaussy, D.A.; Phillips, R.J. [and others]. 2008. Fuel reduction treatment affect stand structure of hardwood forests in western North Carolina and southern Ohio, USA. Forest Ecology and Management. 225: 3117–3129.

BOXELDER (*ACER NEGUNDO* L.) STAND DEVELOPMENT—CAN IT SERVE AS A TRAINER SPECIES?

Brian Roy Lockhart and Ray A. Souter[1]

Abstract—Boxelder (*Acer negundo* L.) is a shade-tolerant species commonly found in west Gulf Coastal Plain floodplains. It is a desirable species for wildlife habitat, but has long been considered a "weed" for timber management, especially when regenerating forests to more desirable species. Results from an archived dataset of stem analysis from a variety of bottomland hardwood species showed the successional pathway of boxelder following the pioneer species eastern cottonwood [*Populus deltoides* (Bartr.) ex Marsh.] on point bars along former Mississippi River channels. We were not able to show trainer effects of boxelder from these two-aged stands. A conceptual model of tree species to plant with red oaks (*Quercus* spp.) in bottomland hardwood afforestation, along with personal observations of boxelder, were used to develop hypotheses for future boxelder stand development research to determine if boxelder could serve as a trainer species. These hypotheses are based on development in even-aged stands.

INTRODUCTION

Boxelder (*Acer negundo* L.) is one of the most widespread of the maples, occurring throughout the Eastern and Central United States and Central Canada (Overton 1990). It is a dioecious, shade-tolerant tree usually found on alluvial soils associated with fronts near rivers (Putnam and others 1960). Further, it is a short-lived (about 60 years) (Green 1934), medium-sized tree reaching 24 to 48 inches d.b.h. and 50 to 75 feet tall (Overton 1990). In the Lower Mississippi Alluvial Valley (LMAV), boxelder usually follows pioneer tree species colonizing new land (point bars) in floodplains (Putnam and others 1960), though it can function as a pioneer tree species in the invasion of old fields (Hosner and Minckler 1960). Boxelder has limited timber value, having been described as a "weed tree" (Putnam and others 1960) and "a worthless and undesirable competitor of good species in forests" (Maeglin and Ohmann 1973). It is, however, a favorable species for wildlife habitat (Overton 1990).

Trainer trees, sometimes called "nurse trees," "fillers," or "companion species," are trees that aid in the development of desired crop trees, but do not have the potential to outgrow crop trees in the rotation (Maine Forest Service 2006). Trainer trees are essentially trees competing with desired trees early in stand development. Trainer trees may be taller during the early stages of stem exclusion, but eventually crop trees stratify above trainer trees to form a majority of the overstory canopy as the stand matures. McKinnon and others (1935) indicate that trainer trees play an important role in improving the quality of crop tree boles by restricting growth of lower branches and hastening pruning. These stem development effects also increase merchantable heights of crop trees. Trainer trees, once relegated to lower canopy positions, increase vertical and horizontal stand structure, and provide additional niches for wildlife. Nicholas and Brown (2002) provide specific steps in the use of trainer species such as radiata pine (*Pinus radiata* D. Don), eucalypts (*Eucalyptus* spp.), poplar (*Populus* spp.), and willow (*Salix* spp.) in growing blackwood (*Acacia melanoxylon* R. Br.) for quality tree boles in New Zealand.

While an old concept (McKinnon and others 1935), the idea of using trainer trees is slowly gaining acceptance in southern bottomland hardwood management (Oliver and others 1990). For example, research has shown that sweetgum (*Liquidambar styraciflua* L.) can provide training effects to develop quality boles in red oaks (*Quercus* spp.) (Clatterbuck and Hodges 1988, Oswalt 2008). This knowledge has been applied to mixed-species plantings of sweetgum and cherrybark oak (*Q. pagoda* Raf.) with similar results (Lockhart and others 2006). Unfortunately, we have little knowledge of other bottomland hardwood species as potential trainers. Lockhart and others (2008) developed a conceptual model of potential tree species that could serve as trainer trees in mixed-species plantings with bottomland red oak species. This model was based on silvical characteristics of the tree species and personal observations, but little direct evidence of mixed-species stand development patterns. Therefore, the objective of this study is to determine boxelder stand development patterns using an archived dataset that includes stem analysis data from a variety of bottomland hardwood species. Our hypothesis is that boxelder, in even-aged stands, will exhibit rapid early height growth compared to neighboring tree species, but will eventually be overtopped, thereby providing trainer tree effects for other species.

METHODS

A hardwood growth and yield dataset developed between 1975 and 1977 is archived at the Southern Hardwoods Laboratory in Stoneville, MS. Dr. Bryce Schlaegel published a series of individual tree species volume and weight tables from this data (Schlaegel 1981, 1984a, 1984b, 1984c, 1984d; Schlaegel and Wilson 1983). Twenty-five stands were located in the LMAV and the adjacent Brown Loam Bluffs in westcentral Mississippi. A circular 0.2-acre plot was located in each stand. Four additional 0.2-acre plots were randomly located within a 5-acre circular area of the center of the first plot such that each plot fell within one of four quadrants of the first plot without overlapping any of the other plots. All trees >4.5 inches d.b.h. were tallied for species; d.b.h. (inches);

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crown class (dominant, codominant, intermediate, and suppressed); and distance and azimuth from plot center.

Trees for destructive sampling were selected after trees on all five plots in a given stand were measured. About 15 trees per stand were selected for destructive sampling with no fewer than 13 trees per stand. Tables of limiting distances by tree diameters for different basal area factors (BAF) were used to select sample trees. An initial BAF 35 was used to determine the number of trees to be sampled from the five plots. If fewer than 13 trees were selected, then a BAF 30 was used and tree selection redone. A BAF 25 was used to gather the minimum number of trees if efforts with the BAF 30 were unsuccessful. Trees were selected as part of a bottomland hardwood growth and yield study, and not for stand development research objectives.

Each selected tree was mechanically felled and marked at 5-foot intervals to the top of the tree. Discs, 1- to 1.5-inch thick, were cut at each mark beginning at the base of the tree. Discs were labeled as to tree number and disc number and sealed in polyethylene bags. Discs for each tree were then placed in a burlap bag and labeled as to location and tree number, then taken to the laboratory for further analysis. Stem analysis followed standard techniques (Oliver 1978, 1982). Age was determined for each disc, then subtracted from the stump disc age to determine tree age at each 5-foot length interval. This data was plotted to determine individual tree length development and compared with the development of other trees sampled from the plot. With stem analysis data, tree length is used instead of tree height to account for differences in actual tree height measurements using stand equipment, such as a clinometer, and the summation of 5-foot intervals from stem analysis. Tree length above the last harvested disc, but <5 feet, is not used in stem analysis. Further, differences >5 feet between tree height and length are probably the result of a crooked main stem near the top of the tree or measurement error.

Distances between boxelders and the other selected trees were calculated using the law of cosines (Selby 1969):

$$a^2 = b^2 + c^2 - 2bc(\text{cosine } A) \qquad (1)$$

where
- a = distance between two trees
- b = distance from plot center to boxelder
- c = distance from plot center to other selected trees
- A = angle between the two trees calculated as the difference between the two azimuths

Crown radii were calculated using d.b.h. and equations developed by Bechtold (2003) for eastern cottonwood (*P. deltoides* Bartr. ex. Marsh.) and Lockhart and others (2005) for boxelder.

In reviewing the dataset, four plots were found in two stands that contained a destructively sampled boxelder and a second tree species. In all cases, the second species was eastern cottonwood. Three plots were located on Indian Point near the Mississippi River in Desha County, AR (33°42' N, 91°7' W). Soil is the Sharkey-Commerce-Coushatta association, and is considered frequently flooded [Sharkey clay (very fine, smectitic, thermic Chromic Eqiapuerts), Commerce silt loam (fine-silty, mixed, superactive, nonacid, thermic Fluvaquentic Endoaquepts), and Coushatta silt loam (fine-silty, mixed, superactive, thermic Fluventic Eutrudepts)]. The stand (stand 16) was dominated by eastern cottonwood that developed on a bar along a former channel of the Mississippi River. The stand initiated soon after a manmade channel diversion, called Caulk Cut-off, was completed in 1937. The stand contained 130 trees per acre (64 percent boxelder), 60 square feet of basal area per acre (67 percent eastern cottonwood), and an average stand diameter of 8.2 inches (14.2 inches for eastern cottonwood) (table 1). Trees destructively sampled for stem analysis were boxelder and eastern cottonwood.

The second stand (stand 19) was also located on Indian Point, but in Bolivar County, MS, about one-quarter mile north of the first stand. Soil is broadly mapped as alluvial soils with a loamy sand surface and sandy subsurface texture. The stand was a mixture of eastern cottonwood, black willow (*S. nigra* Marsh.), and riverfront hardwood species that also initiated soon after the Caulk Cut-off was completed. The stand contained 146 trees per acre [36 and 28 percent American sycamore (*Platanus occidentalis* L.) and eastern cottonwood, respectively]; 146 square feet of basal area per acre (74 percent eastern cottonwood); and an average stand diameter of 11.5 inches (21.5 and 18.5 inches for eastern cottonwood and black willow, respectively) (table 1). One boxelder and two eastern cottonwoods were sampled for stem analysis.

RESULTS

The plot age structure in both stands showed two distinctive age classes (figs. 1, 2, 3, and 4). Eastern cottonwood represented the older age class, while boxelder represented the younger age class.

Plot 16-2 contained three trees utilized for stem analysis: one boxelder and two eastern cottonwoods (table 2). The codominant eastern cottonwood was 14 years older than the boxelder. Further, it was 281 percent larger in d.b.h. and 127 percent taller in height than the boxelder. Average annual height growth was 3.4 and 3.3 feet for eastern cottonwood 1 and eastern cottonwood 2, respectively. Average annual height growth for the boxelder was 2.7 feet. The boxelder, which was underneath both eastern cottonwoods at the time of sampling (table 2), maintained a consistent increase in length, but not as rapid in length development as the eastern cottonwoods (fig. 1).

Plot 16-3 contained three boxelders and one eastern cottonwood from stem analysis (table 3). The eastern cottonwood was 31 years old, while the boxelders were 22 years old. Further, the eastern cottonwood was 214 percent larger in d.b.h. and 97 percent taller than the average of the

Table 1—Tree species composition, number per acre, basal area per acre, and average d.b.h. for stands containing boxelder used in stem analysis

	Stand 16			Stand 19		
Species	*n*	Basal area	Average d.b.h.	*n*	Basal area	Average d.b.h.
	trees per acre	*square feet per acre*	*inches*	*trees per acre*	*square feet per acre*	*inches*
American elm	—[a]	—	—	2.0 (<0.1)	0.4 (<0.1)	6.2 (0.6)
Black willow	1.0[b]	0.3	7.6	4.0 (1.4)	7.2 (1.9)	18.5 (2.4)
Boxelder	83.0 (4.8)	17.4 (1.0)	6.1 (0.4)	17.0 (5.0)	5.2 (2.1)	7.1 (1.0)
Cottonwood	36.0 (3.6)	40.2 (3.1)	14.2 (2.2)	41.0 (4.2)	107.1 (8.4)	21.5 (1.5)
Green ash	—	—	—	3.0 (0.7)	1.2 (0.3)	8.7 (0.3)
Sugarberry	6.0 (1.0)	1.4 (0.3)	6.0 (0.7)	27.0 (2.5)	7.9 (1.4)	6.7 (1.3)
Sycamore	4.0 (0.6)	0.7 (0.1)	5.6 (0.5)	52.0 (1.8)	16.4 (1.1)	7.4 (0.7)
Stand[c]	130 (18.4)	59.9 (12.3)	8.2 (0.5)	146.0 (19.5)	145.5 (30.5)	11.5 (0.9)

Numbers in parentheses represent one standard deviation.

[a] No trees for this species were present in the sampling for this stand.

[b] Only one tree was measured for this species resulting in no standard deviation calculation.

[c] Stand values are based on plot averages and not the addition of individual species trees per acre, basal area per acre, or average d.b.h.

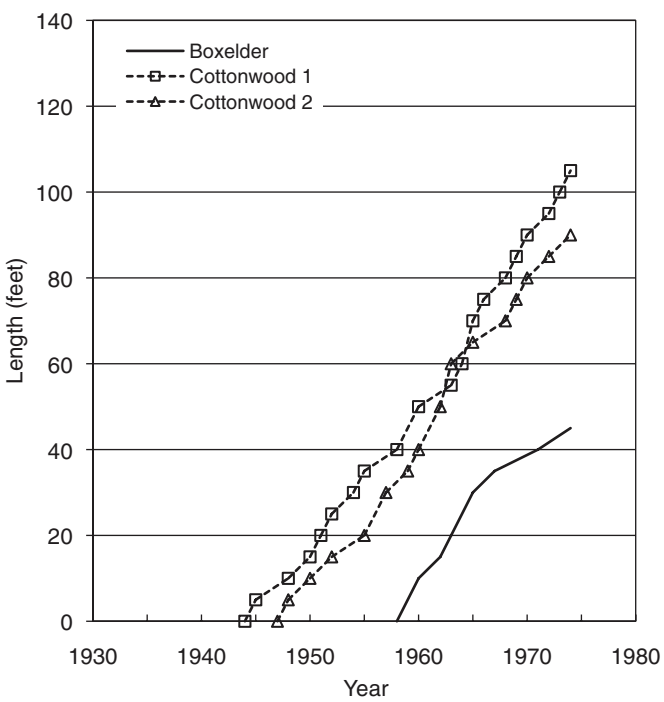

Figure 1—Boxelder and eastern cottonwood length development measured in a destructively sampled stem analysis plot (plot 16-2) on Indian Point, Desha County, AR, in 1977.

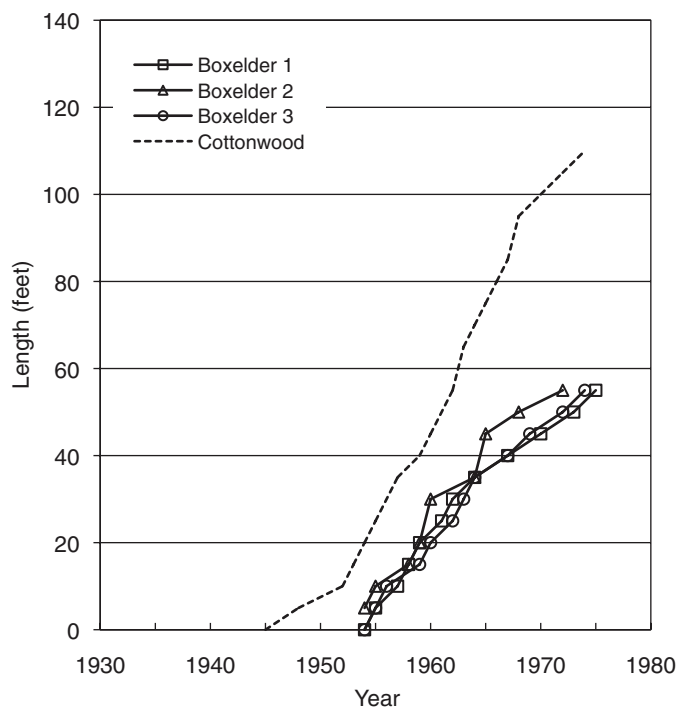

Figure 2—Boxelder and eastern cottonwood length development measured in a destructively sampled stem analysis plot (plot 16-3) on Indian Point, Desha County, AR, in 1977.

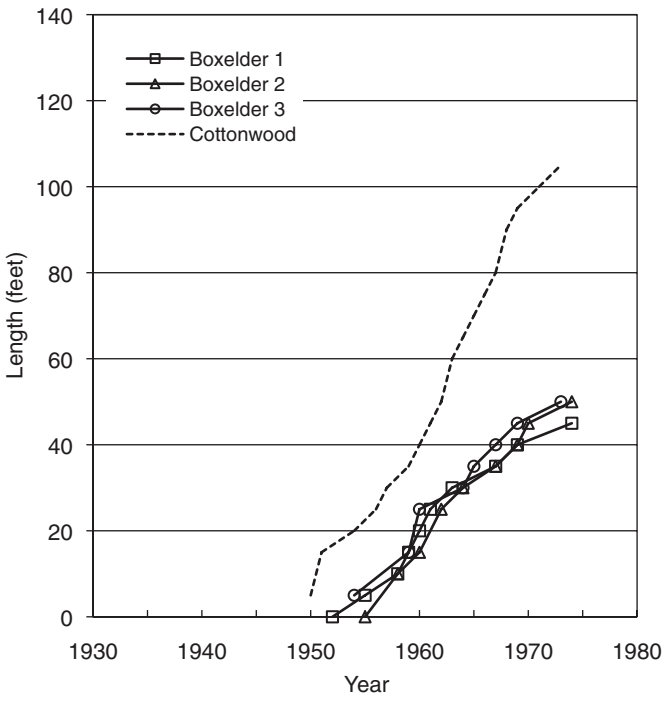

Figure 3—Boxelder and eastern cottonwood length development measured in a destructively sampled stem analysis plot (plot 16-4) on Indian Point, Desha County, AR, in 1977.

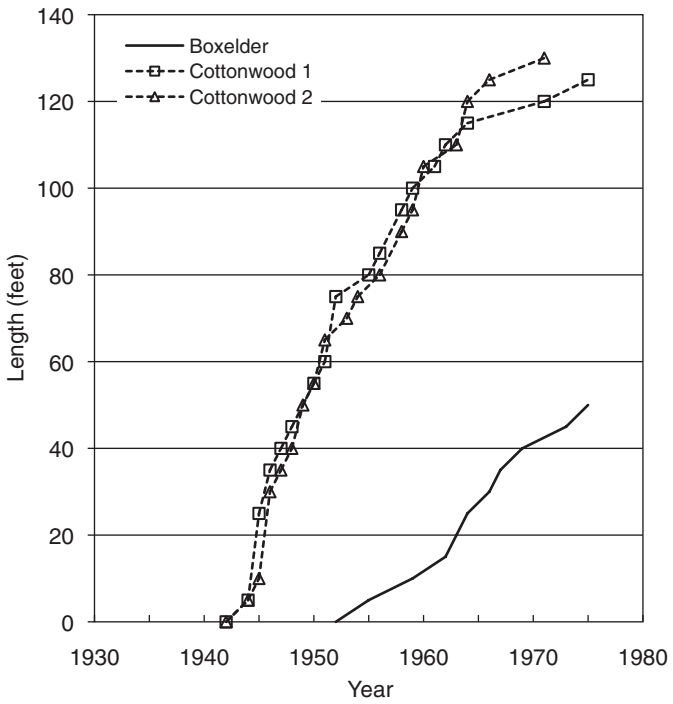

Figure 4—Boxelder and eastern cottonwood length development measured in a destructively sampled stem analysis plot (plot 19-5) on Indian Point, Bolivar County, MS, in 1977.

Table 2—Tree characteristics on a destructively sampled stem analysis plot (plot 16-2) on Indian Point, Desha County, AR, in 1977

Species	Age	D.b.h.	Height	Crown class	From plot center		Distance from boxelder	Crown radius
					Azimuth	Distance		
	years	*inches*	*feet*		*degrees*	*feet*	---------- *feet*	----------
Boxelder	18	5.8	48	Suppressed	67	6.0		6.3
Cottonwood 1	32	22.1	109	Codominant	155	15.6	16.5	34.6
Cottonwood 2	29	10.0	96	Intermediate	285	8.8	12.7	17.5

Table 3—Tree characteristics on a destructively sampled stem analysis plot (plot 16-3) on Indian Point, Desha County, AR, in 1977

Species	Age	D.b.h.	Height	Crown class	From plot center		Distance from cottonwood	Crown radius
					Azimuth	Distance		
	years	*inches*	*feet*		*degrees*	*feet*	---------- *feet*	----------
Boxelder 1	22	7.3	56	Intermediate	16	6.3	11.6	7.1
Boxelder 2	22	6.1	59	Intermediate	167	5.7	23.3	6.5
Boxelder 3	22	6.2	59	Intermediate	250	4.6	20.6	6.5
Cottonwood	31	20.4	114	Codominant	10	17.9		32.2

boxelders. As with the previous plot, the boxelders maintained a consistent increase in length, but not as rapid as the eastern cottonwood (fig. 2).

Plot 16-4 also contained three boxelders and one eastern cottonwood. The eastern cottonwood was only 2 to 5 years older than the boxelders, but was 174 to 302 percent larger in d.b.h. and 107 to 127 percent taller in height compared to the boxelders (table 4). Annual height growth was rapid for the eastern cottonwood (4.2 feet per year) compared to the boxelders (2.0 to 2.5 feet per year). As with the previous plots, the boxelders maintained a consistent increase in length, but not as rapid as the eastern cottonwood (fig. 3).

Plot 19.5 contained one boxelder and two codominant eastern cottonwoods. The eastern cottonwoods were the oldest and largest of the four plots at 34 years, 22.1 inches d.b.h, and 131 feet tall (table 5). They were slowing in length growth by the late 1960s, while the boxelder maintained slow, but consistent increases in length (fig. 4).

DISCUSSION

The dataset used in this paper was developed for a bottomland hardwood growth-and-yield study. Trees selected for destructive sampling were not selected to test hypotheses of bottomland hardwood stand development. The cottonwood trees may or may not have been in competition with the boxelder. Further, trees that appear to be competing with boxelder at the time of sampling may not have been competing with boxelder earlier in stand development. Regardless, this dataset, along with a conceptual model of tree species to plant with red oaks in bottomland hardwood afforestation and personal observations of boxelder, does present interesting questions for further study involving boxelder development and its potential role as a trainer species.

The finding of large, codominant eastern cottonwood in each of the stem analysis plots was not surprising since the plots were located along former channels of the Mississippi River in the batture lands—lands unprotected from river flooding. Soils were typical of new land formed along major river channels, where eastern cottonwood is considered a pioneer species (Greulich and others 2007). *Populus* species can tolerate sedimentation on these lands by developing new root primordia along the tree bole (Smith and Wareing 1972). As annual sedimentation decreases with increasing elevation from previous deposition, other tree species become established underneath the eastern cottonwood. This establishment pattern occurred in each plot with boxelder, although boxelder has also been observed to produce roots along its bole if covered by additional sedimentation (personal observation by the senior author). Our results, through stem age and length development analysis, confirm previous reports on the successional pathway of boxelder in riverfront

Table 4—Tree characteristics on a destructively sampled stem analysis plot (plot 16-4) on Indian Point, Desha County, AR, in 1977

Species	Age	D.b.h.	Height	Crown class	From plot center		Distance from cottonwood	Crown radius
					Azimuth	Distance		
	years	*inches*	*feet*		*degrees*	*feet*	---------- *feet*	----------
Boxelder 1	24	4.9	48	Suppressed	39	5.6	23.5	5.9
Boxelder 2	21	7.2	53	Suppressed	164	7.9	28.0	7.0
Boxelder 3	22	6.4	53	Intermediate	224	5.8	21.2	6.6
Cottonwood	26	19.7	109	Codominant	299	21.9		31.2

Table 5—Tree characteristics on a destructively sampled stem analysis plot (plot 19-5) on Indian Point, Bolivar County, MS, in 1977

Species	Age	D.b.h.	Height	Crown class	From plot center		Distance from boxelder	Crown radius
					Azimuth	Distance		
	years	*inches*	*feet*		*degrees*	*feet*	---------- *feet*	----------
Boxelder	24	6.6	52	Suppressed	359	7.1		6.7
Cottonwood 1	34	20.8	131	Codominant	140	24.2	30.1	32.7
Cottonwood 2	34	23.4	130	Codominant	191	18.4	25.4	36.4

hardwood stands (Maeglin and Ohmann 1973). Following eastern cottonwood establishment and development on newly formed land, species including boxelder, American sycamore, American elm (*Ulmus americana* L.), sugarberry (*Celtis laevigata* Willd.), and green ash (*Fraxinus pennsylvanica* Marsh.) become established (Hodges 1997).

We can neither prove or disprove our hypothesis that boxelder exhibited rapid early height growth compared to neighboring tree species in even-aged stands and was eventually overtopped, thereby providing training tree effects for these species. In each plot, eastern cottonwood became established before boxelder, resulting in two-aged stands. The rapid height growth of the older eastern cottonwood was too great for boxelder to ever become a member of the overstory canopy or to provide training effects on eastern cottonwood boles. We do present two hypotheses, based on the literature and personal observations, for future research consideration regarding boxelder stand development in mixed-species bottomland hardwood forests.

Hypothesis no. 1: Boxelder can serve as a trainer species for red oaks during development in even-aged stands.

A conceptual model of species to plant in intimate mixtures with red oaks in bottomland afforestation suggests that *Acer* species [boxelder, red maple (*A. rubrum* L.), and silver maple (*A. saccharinum* L.)] may be useful to "train" red oaks to develop better quality boles (Lockhart and others 2008). Tree characteristics that favor these species as trainers include fast (but not too rapid) early height growth, small relative twig diameter and durability, and an indeterminate shoot growth pattern. Following a period of slow early height growth, red oaks should be able to stratify above these species, primarily from crown abrasion following periodic, heavy wind events.

Boxelder's best growth occurs on sandy loam soils along river fronts. These sites, especially in the LMAV, are not commonly oak sites due to a basic or neutral soil pH. But, boxelder can grow on a variety of soils, ranging from heavy clays to pure sands, and is commonly associated with oaks such as Nuttall (*Q. nuttallii* Palmer), water (*Q. nigra* L.), willow, and overcup (*Q. lyrata* Walter) (Overton 1990). Therefore, natural mixtures of boxelder and bottomland oaks are not uncommon.

The senior author has observed annually for the past 9 years the development of an oak afforestation project on the Three Rivers Wildlife Management Area in Concordia Parish, LA. This site, adjacent to the Mississippi River mainline levee on the protected side of the levee, was a former agricultural field and then a pasture. It was planted in 1997 with Nuttall, water, swamp chestnut (*Q. michauxii* Nutt.), and white (*Q. alba* L.) oaks. The area was heavily invaded by other tree species early in stand development. A section of the stand was dominated early by boxelder and swamp dogwood (*Cornus drummondii* C.A. Mey.). The boxelder established from seed windblown from adjacent forests. The swamp dogwood reproduction was sprout origin resulting from periodic mowing of the site prior to planting trees. These species were 1.5 to 2 times the height of the red and white oaks during the early years of development. In the past 2 years, the planted oaks began emerging above these "competing" species and will soon become the primary overstory species. Boxelder and swamp dogwood will be relegated to lower canopy positions, but they are currently forcing the oaks to focus on height growth to overtop them instead of crown expansion. Oak crown expansion will occur after stratification above the boxelder and swamp, followed by increased diameter growth on a taller, cleaner bole.

Further circumstantial evidence that boxelder may serve as a potential trainer for *Quercus* spp. involves red maple in New England. Oliver (1978) showed a pattern of stand development in which black birch (*Betula lenta* L.) and red maple were initially taller than northern red oak. By age 40, the red oak stratified above the birch and maple, with the benefit of being trained by these species. As an *Acer* species, boxelder may provide a similar trainer role as red maple in the development of oaks on bottomland hardwood sites.

Hypothesis no. 2: Boxelder's time as a trainer species will be less than the time sweetgum serves as a trainer species for bottomland red oaks.

Clatterbuck and Hodges (1988) found that cherrybark oak would stratify above sweetgum in about 21 to 25 years in natural, old-field stands along minor stream floodplains in central Mississippi. A similar pattern of stand development occurred with sweetgum and bottomland red oaks (cherrybark, water, and willow) following clearcut harvesting, except it took the red oaks about 30 years to stratify above the sweetgum (Johnson and Krinard 1988). Further, Lockhart and others (2006) found cherrybark oak stratified above sweetgum between 17 and 21 years after planting in intimate mixtures. Interspecific competition between these species forced bottomland red oaks to grow in height to attain canopy position, resulting in high-quality boles. But our observations indicate potential shorter merchantable lengths on red oaks competing with boxelder because the oaks stratified above the boxelder earlier than the 21 to 30 years it takes red oaks to overtop sweetgum. Like sweetgum, boxelder can have rapid early height growth, but, unlike sweetgum, it apparently does not have the ability to sustain rapid early height growth for many years. Boxelder is a shade-tolerant, short-lived, small stature species while sweetgum is a shade-intolerant, long-lived, large species.

Research recommendations for boxelder involve the need for more knowledge about the interspecific interactions of boxelder and other bottomland hardwood species during early stand development. Can boxelder serve as a trainer species for oaks? Can boxelder serve as a trainer species for green ash? We believe it can, but further research is needed to confirm our observations. The use of trainer species to develop better quality boles and crowns of desired species is a simple, but often neglected, concept in mixed-

species hardwood management. As we learn more about how bottomland hardwood stands develop, we can use this information to direct silviculture practices to better meet landowner objectives.

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LITERATURE CITED

Bechtold, W.A. 2003. Crown-diameter prediction models for 87 species of stand-grown trees in the Eastern United States. Southern Journal of Applied Forestry. 27: 269–278.

Clatterbuck, W.K.; Hodges, J.D. 1988. Development of cherrybark oak and sweetgum in mixed, even-aged bottomland stands in central Mississippi, USA. Canadian Journal of Forest Research. 18: 12–18.

Green, G.R. 1934. Trees of North America—the broadleaves. Ann Arbor, MI: Edwards Brothers, Inc. 344 p. Vol. II.

Greulich, S.; Franklin, S.; Wasklewicz, T.; Grubaugh, J. 2007. Hydrogeomorphology and forest composition on Sunrise Towhead Island in the Lower Mississippi River. Southeastern Naturalist. 6: 217–234.

Hodges, J.D. 1997. Development and ecology of bottomland hardwood sites. Forest Ecology and Management. 90: 117–125.

Hosner, J.F.; Minckler, L.S. 1960. Hardwood reproduction in the river bottoms of southern Illinois. Forest Science. 6: 67–77.

Johnson, R.L.; Krinard, R.M. 1988. Growth and development of two sweetgum-red oak stands from origin through 29 years. Southern Journal of Applied Forestry. 12: 73–78.

Lockhart, B.R.; Gardiner, E.; Leininger, T.; Stanturf, J. 2008. A stand development approach to oak afforestation in the Lower Mississippi Alluvial Valley. Southern Journal of Applied Forestry. 32: 120–129.

Lockhart, B.R.; Hodges, J.D.; Ezell, A.W.; Clatterbuck, W.K. 2006. Using natural stand development patterns in artificial mixtures: a case study with cherrybark oak and sweetgum in east-central Mississippi, USA. Forest Ecology and Management. 222: 202–210.

Lockhart, B.R.; Weih, R.C., Jr.; Smith, K.M. 2005. Crown radius and diameter at breast height relationships for six bottomland hardwood species. Journal of the Arkansas Academy of Science. 59: 110–115.

Maeglin, R.R.; Ohmann, L.F. 1973. Boxelder (*Acer negundo*): a review and commentary. Bulletin of the Torrey Botanical Club. 100: 357–363.

Maine Forest Service. 2006. Weeding and thinning young forest stands. Inf. Sheet 1. Augusta, ME. 2 p.

McKinnon, F.S.; Hyde, G.R.; Cline, A.C. 1935. Cut-over old field pine lands in central New England: a regional study of the composition and stocking of the ensuing volunteer stands. Harvard For. Bull. 18. Petersham, MA: Harvard Forest. 80 p.

Nicholas, I.; Brown, I. 2002. Blackwood: a handbook for growers and end users. For. Res. Bull. 225. Rotorua, New Zealand: New Zealand Forest Research Institute. 95 p.

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

Oliver, C.D. 1982. Stand development—its uses and methods of study. In: Means, J.E., ed. Forest succession and stand development research in the Northwest. Corvallis, OR: Oregon State University, Forest Research Laboratory: 100–112.

Oliver, C.D.; Clatterbuck, W.K.; Burkhardt, E.C. 1990. Spacing and stratification patterns of cherrybark oak and American sycamore in mixed, even-aged stands in the Southeastern United States. Forest Ecology and Management. 31: 67–79.

Oswalt, C.M. 2008. Development of a simulation system for testing hardwood plantation silvculture. Knoxville, TN: University of Tennessee. 147 p. Ph.D. dissertation.

Overton, R.P. 1990. Boxelder. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America: hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service: 41–45. Vol. 2.

Putnam, J.A.; Furnival, G.M.; McKnight, J.S. 1960. Management and inventory of southern hardwoods. Agric. Handb. 181. New Orleans: U.S. Department of Agriculture Forest Service. 102 p.

Schlaegel, B.E. 1981. Willow oak volume and weight tables. Res. Pap. SO-173. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E. 1984a. Green ash volume and weight tables. Res. Pap. SO-206. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E. 1984b. Overcup oak volume and weight tables. Res. Pap. SO-207. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E. 1984c. Sugarberry volume and weight tables. Res. Pap. SO-205. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E. 1984d. Sweetgum volume and weight tables. Res. Pap. SO-204. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E.; Willson, R.B. 1983. Nuttall oak volume and weight tables. Res. Pap. SO-186. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Selby, S.M., ed. 1969. Standard mathematical tables. 17th ed. Cleveland, OH: The Chemical Rubber Co. 724 p.

Smith, N.G.; Wareing, P.F. 1972. The distribution of latent root primordia in stems of *Populus* x *robusta* and factors affecting the emergence of preformed roots from cuttings. Forestry. 45: 197–209.

LIGHT, CANOPY CLOSURE, AND OVERSTORY RETENTION IN UPLAND OZARK FORESTS

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Abstract—Foresters, wildlife biologists, and naturalists manipulate forest composition and structure for numerous reasons including forest regeneration, timber production, wildlife habitat, conservation of native biodiversity, and ecosystem restoration. Light conditions in the understory of forests and woodlands are often key in meeting the management objectives. In this study, predictive models were developed relating mean diurnal understory photosynthetically active radiation (PAR) (400 to 700 nm) as percentages of above canopy (understory PAR) to measures of canopy closure, stocking, basal area, and density. Relationships between percent crown closure and density in Missouri's upland oak (*Quercus* spp.) and oak-pine (*Pinus* spp.) stands are not well defined, nor are the relationships between each measure and light conditions near the forest floor. Our objectives for this paper were: (1) to analyze the relationships between understory PAR and stand metrics and (2) to analyze the relationships between canopy closure and stand metrics. Understory PAR may be predicted by models utilizing stocking, basal area, or stand density, but understory PAR was primarily controlled by canopy closure. Canopy closure was related to stocking and density.

INTRODUCTION

One of the primary reasons to alter stand density and structure is to increase the amount of light that reaches the understory (Blizzard and others 2007, Larsen and others 1999, Minckler and others 1973). Although foresters are interested in altering the amount of light in stands, they seldom measure light levels during routine forest inventories. Instead, they collect information that can be used to quantify stand density including the number of trees per unit area, quadratic mean diameter, basal area, stocking, and less commonly, canopy closure. Therefore, many studies utilize stand metrics to predict seedling and sprout growth and recruitment (Dey and others 2008, Jensen and Kabrick 2008, Larsen and others 1997).

Light is rarely measured by foresters, but physiological studies develop growth guidelines based upon photosynthetically active radiation (PAR) (Johnson and others 2002, Pallardy and Kozlowski 2007). Models relating the percent of above canopy PAR that reaches the understory would bridge between stand metrics and physiology studies. With understory PAR—stand metrics models, foresters would be able to predict understory PAR based upon the level of overstory retention.

We conducted this study to examine the relationships between light, canopy closure, and stand density in oak-pine (*Quercus* spp.-*Pinus* spp.) stands having a range of residual stocking levels in the Ozark Highlands of southeastern Missouri. Our objectives for this paper were: (1) to analyze the relationships between understory PAR, canopy closure, and stand metrics, and (2) to analyze the relationships between canopy closure and stand metrics. We hypothesized that these measures of stand density may be used individually to estimate (1) understory diurnal PAR as a percentage of above canopy PAR and (2) canopy closure under mature oak-pine canopies following clearcut, seed tree, light shelterwood, or heavy shelterwood treatments.

METHODS

The study was conducted on the Sinkin Experimental Forest located on the Salem Ranger District of the Mark Twain National Forest. The Sinkin Experimental Forest is located in the Current River Oak-Pine Woodland/Forest Hills landtype association in the Ozark Highlands (Nigh and Schroeder 2002). Overstory species included white oak (*Q. alba* L.), post oak (*Q. stellata* Wangenh.), black oak (*Q. velutina* Lam.), scarlet oak (*Q. coccinea* Münchh.), shortleaf pine (*P. echinata* Mill.), mockernut hickory [*Carya tomentosa* (Lam.) Nutt.], and black hickory (*C. texana* Buckley). Midstory and understory species included blackgum (*Nyssa sylvatica* Marsh.), persimmon (*Diospyros virginiana* L.), and dogwood (*Cornus florida* L.). Soils were mapped as Coulstone-Clarksville, 3 to 35 percent slope (loamy-skeletal, siliceous, semiactive, mesic Typic Paleudults) and Nixa (loamy-skeletal, siliceous, active, mesic Glossic Fragiudults)-Clarksville, 1 to 3 percent slope. Plots were located on broad ridges and upper backslopes in parent materials derived from dolomite of the Gasconade Formation.

The study was designed to simulate clearcut, seed tree, light shelterwood, and heavy shelterwood cuttings. Forty-eight 1-acre experimental units were thinned (12 each of 0, 20, 40, and 60 percent stocking) and residual stocking was checked randomly within each plot using a 10 basal area factor prism. Thinning occurred in 2005 to 2007. The ranges of acceptable basal areas were <10, 20 to 30, 50 to 60, and 60 to 80 square feet per acre, respectively. Oak and shortleaf pine stocking levels were calculated individually and summed to determine total stocking (Gin[g]rich 1967, Rogers 1983). Preferred residual trees were well spaced, with ≥11 inches diameter at breast height (d.b.h.) and healthy crowns. Some 6- to 10-inch

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d.b.h. trees were retained to meet stocking objectives. Order of preference for retention was: (1) shortleaf pine and white oak, (2) post oak, (3) scarlet oak, and (4) black oak. The midstory and understory were cut near the ground using chainsaws leaving a few dogwoods.

In the center of each 1-acre experimental unit, we established a circular 0.2-acre plot for collecting stand density, canopy closure, and PAR data. We sampled the quantity of PAR (400 to 700 nm) using one Onset S-LAI-M003 PAR in the center of each plot for over 48 hours (Onset Computer Corp., Bourne, MA). We also placed a sensor in a field to record above canopy PAR. The sensors were connected to Onset HOBO® H21-002 Micro Station dataloggers set to record microE per second for 1 second every 60 seconds and to average every hour. Hourly estimates were summed for each day and understory PAR was calculated as a percentage of above canopy PAR. Only the day with the highest readings was used for each group of 16 plots: group 3—June 22, group 1—August 5, group 2—August 15, 2008. Each group contained a mixture of stocking levels, but the plots were analyzed as completely randomized.

Canopy closure readings were taken by the same observer on all plots with a convex spherical densiometer model A (Robert E. Lemmon, Forest Densiometers, Bartlesville, OK) (Lemmon 1956). The recorded reading was the average of four readings taken facing the cardinal directions, with the spherical densiometer at plot center. Hemispherical color canopy photographs were taken above the center stake on cloudy days using a Canon EOS Rebel® 35 mm film body and a Sigma® 8mm F3.5 EX DG circular fisheye lens set on infinite focus. A polarizing gel was placed behind the lens running northsouth (top to bottom) in the space provided by the manufacturer. The camera was placed over plot center, leveled on a tripod with the top of the camera pointed north, and with the top of the lens at 39 inches from the ground. The film was developed and photographs were digitized from film. Projection calibration data were faxed from Sigma® and entered into the Gap Light Analyzer software (GLA) (Frazer and others 1997, 1999). The geographic coordinates and corresponding date that the light was measured on were entered into the GLA for each plot. Canopy closure was calculated in the GLA for each plot and then compared as a percentage of the values calculated for photograph taken in the open field to account for the ring of trees seen in the photographs of the clearcuts and the open field.

Linear and nonlinear models, for estimating understory PAR as a percentage of above canopy PAR and for estimating canopy closure measured by a spherical densiometer, were compared by predictor variables. Nonlinear models were selected that best fit the general relationship of the independent-dependent relationships. Logic tests were then applied. For instance, quadratic models were disqualified because they indicated an increase in understory light or a decrease in canopy closure in dense stands. The remaining model forms were tested for each independent-dependent variable relationship. Regression was performed using PROC MIXED and PROC NLMIXED (SAS 9.1, SAS Institute Inc., Cary, NC). *F*-values were used to test for model significance. Akaike's information criterion (AIC) was used to select between models that fit the general relationship of the dependent-independent relationships (Burnham and Anderson 2003). AIC was also used to select between linear and nonlinear models for each predictor rather than R-squared.

RESULTS

Percent of Above Canopy Diurnal Photosynthetically Active Radiation in the Understory

Percent of above canopy diurnal PAR measured in the understory of oak and oak-pine clearcut, seed tree, and light and heavy shelterwood stands was better predicted by spherical densiometer canopy closure based upon the lower AIC value (table 1). Understory PAR was inversely proportional to both measures of canopy closure (figs. 1A and 1B). The rate of change remained steady as canopy closure increased, rather than leveling off or declining at an increasing rate. In contrast, the percentage of above canopy diurnal PAR that reached the understory rapidly declined then leveled off as overstory retention increased based upon stocking, basal area, or density (figs. 2A, 2B, and 2C).

Table 1—Models relating percent of above canopy photosynthetically active radiation in understory to canopy closure and stand metrics

Models		*F*-value	AIC[a]	R^2
X = Canopy closure (spherical densiometer, %)	95.4407 – 0.8189 X	450.86[b]	345.3	0.91
X = Canopy closure (photograph, %)	106.2700 – 1.1713 X	202.28	376.4	0.81
X = Stocking (%)	71.2541 exp (–0.03362 X) + 22.4371	144.32	364.9	0.87
X = Basal area (square feet per acre)	68.2547 exp (–0.02839 X) + 25.2099	124.23	371.0	0.85
X = Density (trees per acre)	65.9156 exp (–0.03174 X) + 27.6884	147.87	363.9	0.86

[a] AIC = lowest Akaike's information criterion value was used to determine if the linear or exponential model was the better fit given the model complexity.

[b] *F*-values for all models were significant ($P < 0.00001$).

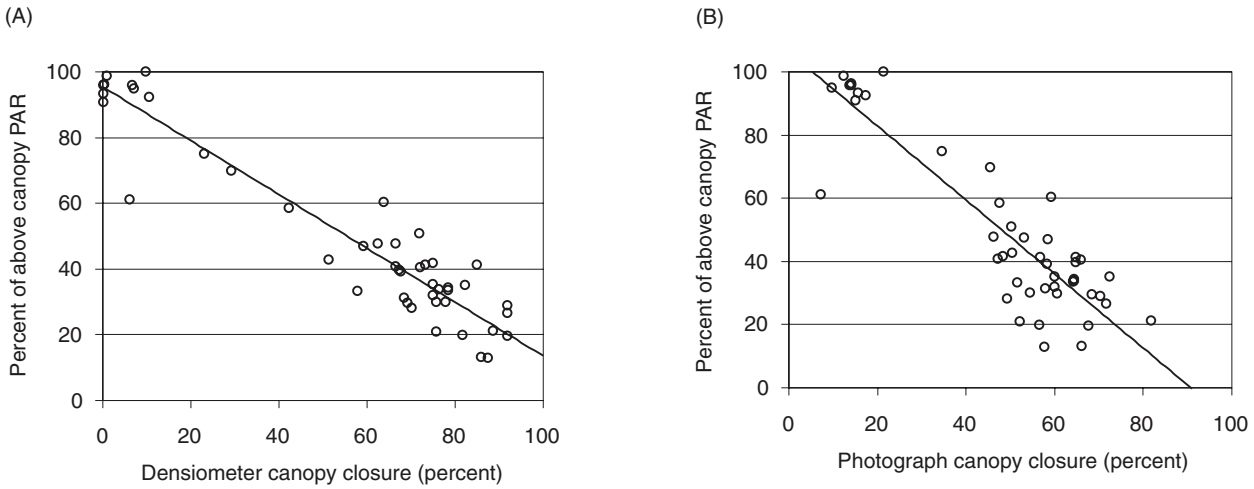

Figure 1—Percent of above canopy diurnal photosynthetically active radiation (PAR) (400 to 700 nm) that reach the understory by: (A) spherical densiometer canopy closure and (B) calibrated hemispherical photograph canopy closure. The percent of PAR that reached the understory was inversely related to canopy closure.

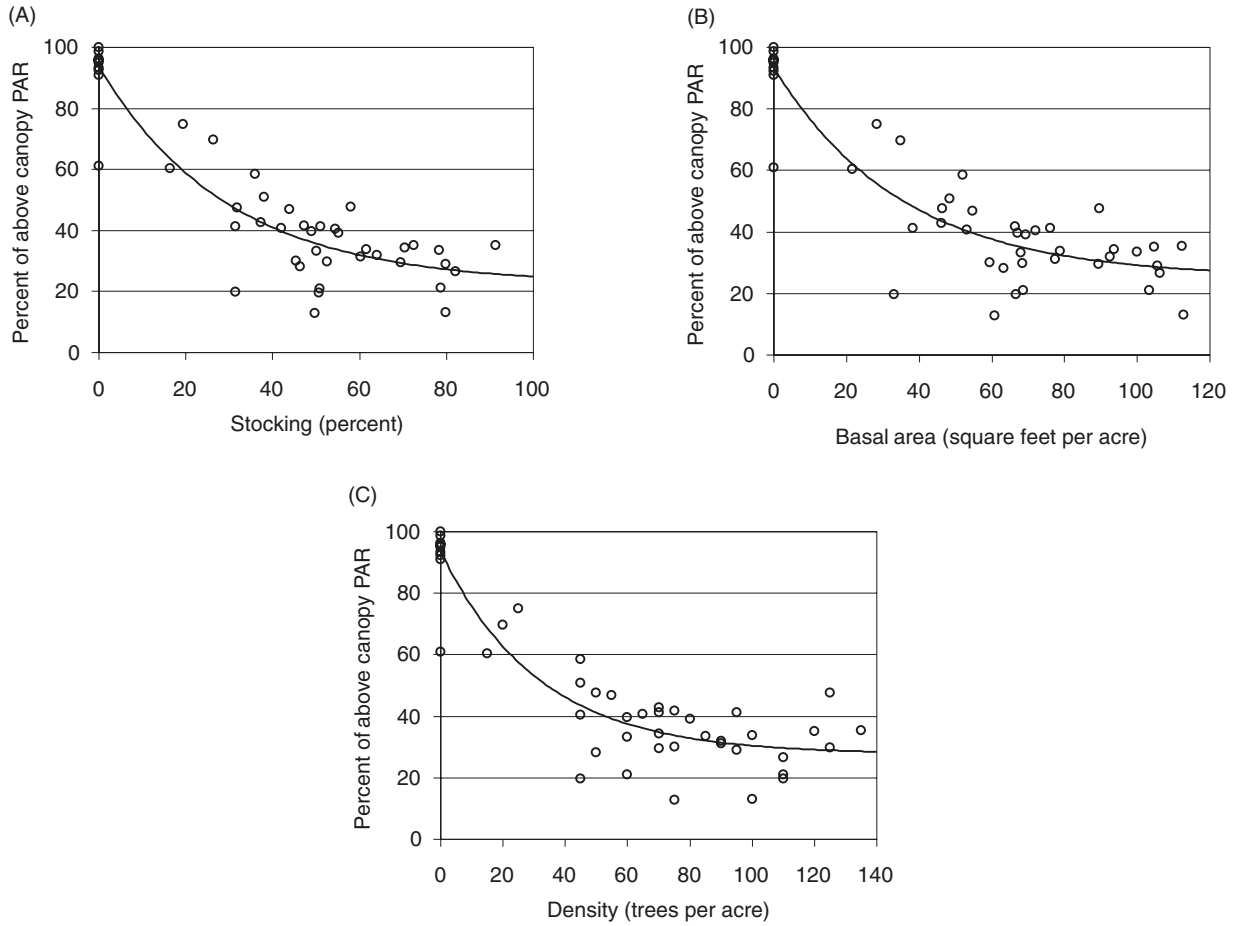

Figure 2—Percent of above canopy diurnal photosynthetically active radiation (PAR) (400 to 700 nm) that reached the understory by (A) stocking, (B) basal area, and (C) density. The exponential model reflects a rapid decrease followed by a leveling off as overstory retention increases.

Canopy Closure

Canopy closure measured by a spherical densiometer in oak and oak-pine clearcut, seed tree, and light and heavy shelterwood stands increased rapidly and leveled off as overstory retention increased (figs. 3A, 3B, and 3C). Stocking and density were the more reliable of the stand metric canopy closure indicators based upon AIC values, but not as good as directly measuring with a hemispherical photograph (table 2). The relationship of hemispherical photography canopy closure to spherical densiometer canopy closure was not 1 to 1. Spherical densiometer canopy closure had a wider range than the canopy closure calculated from hemispherical photographs, reflecting roughly 20 percent lower maximum readings and roughly 5 percent higher minimum readings even after calibrating with the photograph taken in a 5-acre field (fig. 3D).

DISCUSSION

Light under a forest canopy is affected by both species and structure of the stand (Brown and Parker 1994, Guo and Shelton 1998, Minckler and others 1973, Poulson and Platt 1989). For example, pines generally have a lower crown density than do oaks (Oliver and Larson 1996). Stands with multiple canopy layers generally have a more complex vertical arrangement of trees than do stands that are thinned from below (Lhotka and Loewenstein 2006, 2008; Motsinger and others 2010). Stands with canopy gaps will have a greater range of understory PAR levels across the stand. Consequently, relationships between measures of canopy closure or stand density and light levels need to be established for specific forest types and forest structures to meet the needs of managers. This project focuses on modeling understory PAR following clearcut, seed tree, and light and heavy shelterwood treatments of pine-oak and oak-pine stands.

Measures of stand density have been used to predict growth of reproduction due to their ease of monitoring before, during, and after stand alteration (Brookshire and Shifley 1997, Gwaze 2004, Shifley and Brookshire 2000, Shifley

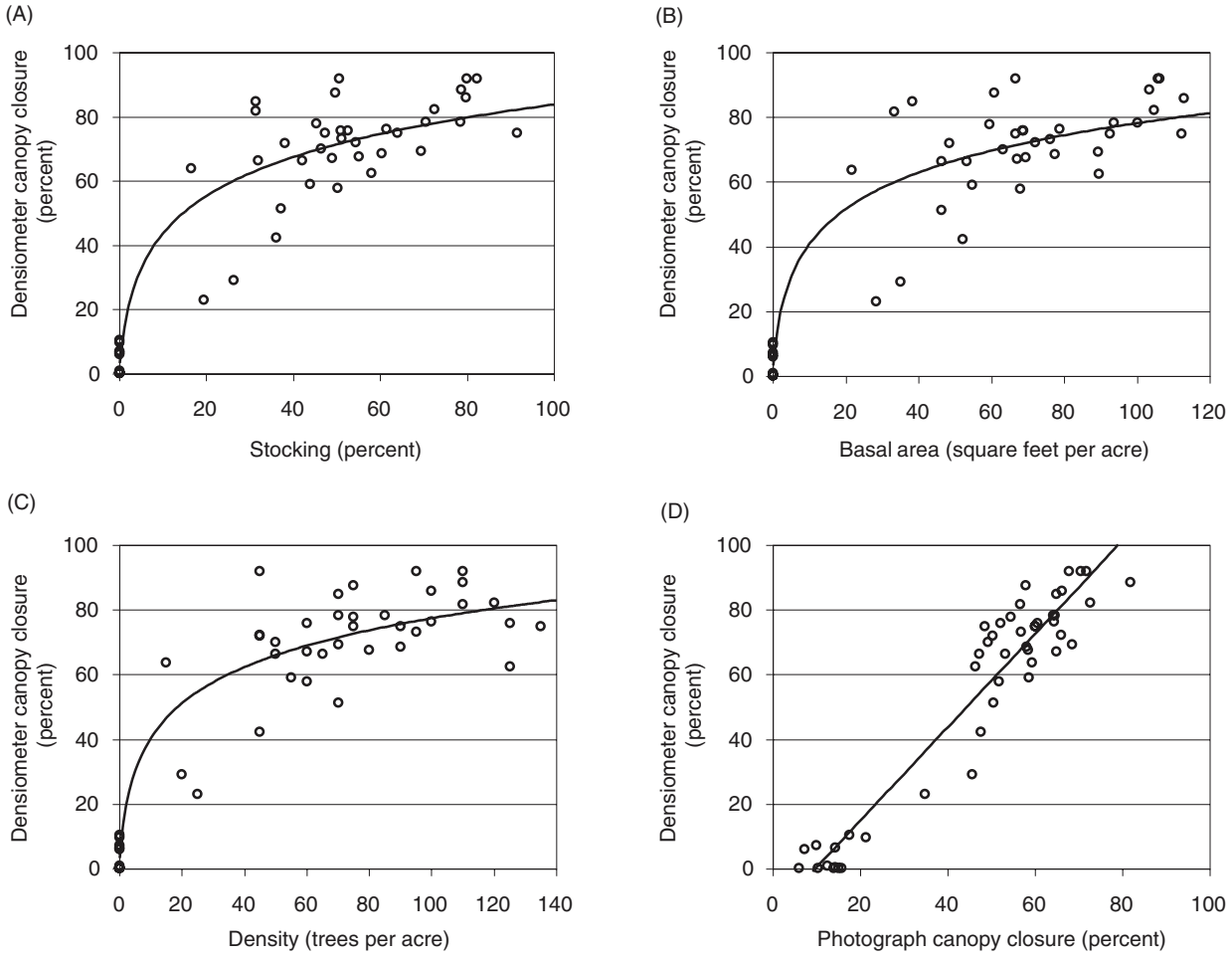

Figure 3—Canopy closure measured with a convex spherical densiometer by: (A) stocking, (B) basal area, (C) density, and (D) calibrated hemispherical photograph canopy closure. The logarithmic model reflects a rapid increase in canopy closure followed by a slower increase as overstory retention increases. Observed canopy closure increased linearly as canopy closure estimated from calibrated canopy photographs increased, although spherical densiometer canopy closure has a wider range of observed values than did canopy closure estimated from photographs even after calibration of the photographs.

Table 2—Models relating canopy closure measured by a spherical densiometer to stocking, basal area, density, and canopy closure calculated from a photograph

Models		*F*-value	AIC[a]	R^2
X = Stocking (%)	41.8335 log10 (1.2077 + X)	738.99[b]	370.0	0.91
X = Basal area (square feet per acre)	39.0443 log10 (1.2270 + X)	677.72	374.0	0.97
X = Density (trees per acre)	38.6471 log10 (1.2300 + X)	755.81	368.9	0.97
X = Canopy closure (photograph, %)	–13.7544 + 1.4415 X	476.22	356.1	0.97

[a] AIC = lowest Akaike's information criterion value was used to determine if the linear or exponential model was the better fit given the model complexity.

[b] *F*-values for all models were significant ($P < 0.00001$).

and Kabrick 2002). A primary goal of stand alteration is to decrease competition for growth resources. PAR in the understory is a key growth resource increased by stand management practices. Establishment and growth of reproduction has been linked to stand metrics, and light requirements are known for some species. Models presented here provide a link between stand metrics and relative light in the understory PAR of stands following clearcut, seed tree, and light and heavy shelterwood treatments.

Canopy closure measured with Lemmon's convex spherical densiometer was the better individual predictor of understory PAR because it is the more direct measure of sky obstruction. Sky obstruction is the primary cause of decreased sunlight (Jennings and others 1999, Oliver and Larson 1996). Canopy closure also integrates the height of the surrounding overstory as well as horizontal area of the canopy opening (Jennings and others 1999). Stocking, basal area, and density are measures of overstory structure, but do not take into account the height of the overstory.

Hemispherical photographs may be used to track changes in canopy closure over time for continuous forest inventory plots, but are less practical than other methods for estimating PAR due to the data entry and data processing required in the office (Lhotka and Loewenstein 2006). Understory PAR was better predicted by canopy closure measured with Lemmon's convex spherical densiometer than by hemispherical canopy photographs as analyzed here. Further techniques may be applied to improve the usefulness of hemispherical canopy photographs (Lhotka and Loewenstein 2006).

CONCLUSIONS

Single-variable stand metric models were successful in modeling the percent of above canopy diurnal PAR that reached the understory given the constraint implicit in the fact that seed tree and shelterwood regeneration methods attempt to leave uniformly spaced mature trees with healthy crowns. Mean percent of above canopy diurnal PAR that reaches the understory is controlled primarily by canopy closure. Canopy closure measured with a spherical densiometer was a better indicator of diurnal understory PAR than canopy closure from hemispherical photographs. As spherical densiometer canopy closure was a better predictor of understory PAR, it would be reasonable to consider additional photograph analysis techniques that mimic spherical densiometer estimates. Stocking and density in these stands were good indicators of spherical densiometer canopy closure.

Incorporating canopy closure measured with a convex spherical densiometer with stocking, basal area, or density may increase the reliability of the estimate of understory PAR. Stands with greater variability or denser midstory and understories may require more complex models (Guo and Shelton 1998; Lhotka and Loewenstein 2006, 2008; Motsinger and others 2010). Additional variables may include height to live crown and total height of the stand and of each canopy layer.

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LITERATURE CITED

Blizzard, E.M.; Larsen, D.R.; Dey, D.C. [and others]. 2007. The state of mixed shortleaf pine-upland oak management in Missouri. In: Kabrick, J.M.; Dey, D.C.; Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: Proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 153–157.

Brookshire, B.; Shifley, S.R., eds. 1997. Proceedings of the Missouri Ozark Forest Ecosystem Project symposium: an experimental approach to landscape research. Gen. Tech. Rep. NC-193. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Experiment Station. 378 p.

Brown, M.J.; Parker, G.G. 1994. Canopy light transmittance in a chronosequence of mixed-species deciduous forests. Canadian Journal of Forest Research. 24: 1694–1703.

Burnham, K.P.; Anderson, D.R. 2003. Model selection and multimodel inference: a practical information-theoretic approach. New York: Springer. 488 p.

Dey, D.C.; Jensen, R.G.; Wallendorf, M.J. 2008. Single-tree harvesting reduces survival and growth of oak stump sprouts in the Missouri Ozark Highlands. In: Jacobs, D.F.; Michler, C.H., eds. Proceedings, 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.

Frazer, G.W.; Canham, C.D.; Lertzman, K.P. 1999. Gap light analyzer (GLA). Version 2.0. Imaging software to extract canopy structure and gap light transmission indices from true-colour fisheye photographs, users manual and program documentation. Burnaby, British Columbia: Simon Fraser University; Millbrook, NY: The Institute of Ecosystem Studies. http://www.ecostudies.org/gla/. [Date accessed: June 4, 2007].

Frazer, G.W.; Trofymow, J.A.; Lertzman, K.P. 1997. A method for estimating canopy openness, effective leaf area index; photosynthetically active photon flux density using hemispherical photography and computerized image analysis techniques. Info. Rep. BC-X-373. Victoria, British Columbia: Canadian Forest Service, Forest Ecosystem Processes Network, Pacific Forestry Centre. 73 p.

Gin[g]rich, S.F. 1967. Measuring and evaluating stocking and stand density in upland hardwood forests in the Central States. Forest Science. 13(1): 38–53.

Guo, Y.; Shelton, M.G. 1998. Canopy light transmittance in natural stands on upland sites in Arkansas. 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: 618–622.

Gwaze, D., ed. 2004. Proceedings of the 2004 Missouri Ozark forest ecosystem project annual meeting. Jefferson City, MO: Missouri Department of Conservation. 119 p.

Jennings, S.B.; Brown, N.D.; Sheil, D. 1999. Assessing forest canopies and understorey illumination: canopy closure, canopy cover and other measures. Forestry. 72(1): 59–73.

Jensen, R.G.; Kabrick, J.M. 2008. Comparing single-tree selection, group selection; clearcutting for regenerating oaks and pines in the Missouri Ozarks. In: Jacobs, D.F.; Michler, C.H., eds. Proceedings, 16th central hardwood forest conference. Gen. Tech. Rep. NRS-P-24. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 38–49.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2002. The ecology and silviculture of oaks. Wallingford Oxon, United Kingdom: CABI Publishing. 528 p.

Larsen, D.R.; Loewenstein, E.F.; Johnson, P.S. 1999. Sustaining recruitment of oak reproduction in uneven-aged stands in the Ozark Highlands. Gen. Tech. Rep. NC-203. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Research Station. 11 p.

Larsen, D.R.; Metzger, M.A.; Johnson, P.S. 1997. Oak regeneration and overstory density in the Missouri Ozarks. Canadian Journal of Forest Research. 27: 869–875.

Lemmon, P.E. 1956. A spherical densiometer for estimating forest overstory density. Forest Science. 2(1): 314–320.

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.

Lhotka, J.M.; Loewenstein, E.F. 2008. Influence of canopy structure on the survival and growth of underplanted seedlings. New Forests. 35(1): 85–104.

Marquis, D.A. 1965. Controlling light in small clearcuttings. RP-NE-39. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 16 p.

Minckler, L.S.; Woerheide, J.D.; Schlesinger, R.C. 1973. Light, soil moisture, and tree reproduction in hardwood forest openings. RP-NC-89. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station: 6.

Motsinger, J.R.; Kabrick, J.M.; Dey, D.C. 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. 39: 195–213.

Nigh, T.A.; Schroeder, W.A. 2002. Atlas of Missouri ecoregions. Jefferson City, MO: Missouri Department of Conservation. 212 p.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. New York: John Wiley. 520 p.

Pallardy, S.G.; Kozlowski, T.T. 2007. Physiology of woody plants. 3rd ed. Boston: Elsevier. 454 p.

Poulson, T.J.; Platt, W.J. 1989. Gap light regimes influence canopy tree diversity. Ecology. 70(3): 553–555.

Rogers, R. 1983. Guides for thinning shortleaf pine. In: Jones, E.P., ed. Proceedings of the 2nd biennial southern silviculture research conference. Gen. Tech. Rep. SE-24. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Experiment Station: 217–255.

Shifley, S.; Kabrick, J. 2002. Proceedings of the second Missouri Ozark forest ecosystems project symposium: post-treatment results of the landscape experiment. Gen. Tech. Rep. NC-227. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 227 p.

Shifley, S.R.; Brookshire, B.L., eds. 2000. Missouri Ozark forest ecosystem project: site, history, soils landforms, woody herbaceous vegetation, down wood and inventory methods for the landscape experiment. Gen. Tech. Rep. NC-208. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Research Station. 314 p.

ASSESSING ANTHROPOGENIC AND NATURAL DISTURBANCES: FOREST RESPONSE TO SIMILARLY AGED CLEARCUT AND TORNADO DISTURBANCES IN AN EAST TENNESSEE OAK-HICKORY FOREST

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Abstract—In February of 1993, an F3 tornado caused a large-scale disturbance in an east Tennessee oak-hickory (*Quercus* spp.-*Carya* spp.) forest. Vegetation response to anthropogenic and natural disturbances was compared by examining two tornado-disturbed areas and five adjacent 1-acre silvicultural clearcut areas unaffected by the tornado disturbance. Nested overstory, midstory, and understory plots (0.1-, 0.02-, and 0.001-acre plots, respectively) were measured to determine species composition, species diversity, stocking, and structure, as well as coarse woody debris (CWD) volume, density, and percent cover. Results, 14 years postdisturbance, indicate that similarities exist in species composition and diversity between the clearcut and tornado areas, though differences in density do exist. Structural differences also occur between the two disturbance types. The presence of residual overstory and midstory trees in the tornado-disturbed areas caused the diameter distribution to have an irregular distribution compared to the typical even-aged distribution of the clearcut. CWD volume, density, and percent cover were significantly higher in the tornado-disturbed blocks.

INTRODUCTION

Prior to human settlement, forests were shaped by natural disturbances such as fire, insect infestations, and wind events. In mesic hardwood forests of the Eastern United States, where humidity limits fire frequency, wind is reported to be one of the main causes of large-scale natural disturbances (Canham and Loucks 1984). Large-scale wind disturbance is relatively irregular in this region but can have more of a long-term effect on stand composition than smaller, more frequent single-tree blowdowns (Clinton and Baker 2000).

On February 21, 1993, the University of Tennessee Forest Resources Research and Education Center (FRREC) in Oak Ridge, Anderson County, TN, was hit by an F3 (Fujita scale) tornado. Since 1950, there have been four recorded tornadoes in Anderson County (National Oceanic and Atmospheric Administration 2007). The aforementioned tornado was the most destructive tornado in the county during that timeframe. The main path of this tornado was roughly 10 miles (16.1 km) long, 0.45 miles (0.72 km) wide, and irregularly shaped (Newbold 1996). On the FRREC, 249 acres of forest were heavily damaged by the tornado and 103 acres had light to moderate damage, totaling roughly 352 acres of damage.

Due to the moderately infrequent and unpredictable nature of tornadoes and large-scale wind disturbances in the Ridge and Valley region, there is little work examining the successional pattern following such disturbance. The tornado disturbance site at the FRREC provides a unique opportunity to evaluate vegetation development following different disturbances. Because vegetation response to large-scale wind disturbance is unpredictable and no predictive outline has yet been created (Peterson and Pickett 1995), we evaluated the vegetation and coarse woody debris (CWD) response of tornado disturbance compared to a silvicultural clearcut (SCC) harvested 4 years prior to tornado disturbance to evaluate differences and similarities between natural and anthropogenic disturbances. The SCC and tornado-disturbed areas were adjacent to each other and had similar site conditions. We recognize that some variation is associated with the 4-year temporal difference between the two disturbances.

Objectives

Fourteen years after the tornado damage and roughly 18 years after the clearcut harvest, vegetation and CWD data were collected and used to meet the following objectives:

- Evaluate the impacts of tornado disturbance and SCC harvests on stand characteristics, including species composition, diversity, diameter distribution, and density
- Assess the impacts of tornado disturbance and SCC harvests on CWD density, CWD volume, and CWD biomass

The objectives listed were met by testing a null hypothesis that all treatments are not significantly different from each other for a given stand characteristic. The alternative hypothesis is that the treatments are significantly different for a given stand characteristic.

STUDY SITE

The University of Tennessee FRREC is located in Oak Ridge, TN, in Anderson County. The FRREC is a 2,260-acre (915-ha) tract that is part of the University of Tennessee Agricultural Experiment Station. The FRREC is found in the Ridge and Valley physiographic province, which is characterized by long, parallel, northeast-southwest running ridges that create narrow valleys between them (Moneymaker 1981).

Average annual precipitation for Oak Ridge is 55.1 inches (140.0 cm), 11.1 inches (28.2 cm) from snowfall. Average monthly temperatures range from 57.2 °F (14 °C) to 77.3 °F (25.2 °C) during spring and summer growing season months (April through September) (National Oceanic and Atmospheric Administration 2007). Soils within the research

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area are in the Fullerton series and are described as very deep, well-drained cherty silt loam soils. These soil types have slopes ranging from 5 to 45 percent (Natural Resources Conservation Service 2007). The average slope in the study area is 22.5 percent (research data).

In February 1993, an F3 tornado impacted the FRREC with maximum wind speeds of 158 to 206 miles per hour (254 to 332 kph) (City-Data 2007). The tornado was limited to the Chestnut Ridge portion of the FRREC and caused extensive levels of windthrow on roughly 350 acres (142 ha). As part of Karen Adreadis's (1995) and Chris Newbold's (1996) thesis projects, three treatments were implemented to evaluate small mammal and avian communities, respectively, and their usage of postdisturbance habitats. These three treatment areas were laid out in the spring of 1994. The tornado-disturbance-only treatment (no subsequent salvage harvest) was the natural disturbance used in this study to compare to anthropogenic disturbance (fig. 1).

A clearcut treatment was the anthropogenic disturbance used in this study. The clearcut was part of a site preparation study conducted in 1989 (Andrews 1995) to determine the most efficient way to regenerate a mixed pine-hardwood stand. The four treatments were SCC, SCC followed by burning, SCC followed by herbicide application then burning, and commercial clearcut. Treatments were broken into 1-acre (0.4-ha) blocks, replicated five times. Following site preparation, all treatments were planted on a 20- by 20-foot (6.1- by 6.1-m) spacing with 50 eastern white pine (*Pinus strobus*) seedlings on half of the block and 50 loblolly pine (*P. taeda*) seedlings on the other half, totaling 100 trees per acre (247 trees/ha).

Figure 1—Treatment areas: Tornado (black) and Clearcut (crosshatch) at the University of Tennessee Forest Resources Research and Education Center in Oak Ridge, TN. (Other areas with hatched pattern were salvaged following tornado disturbance and are not part of this study.)

For the purpose of this research, only SCC blocks were assessed. Seedling survivorship for white pine and loblolly pine were 55 and 70 percent, respectively, 1 year after planting. Since then, the component of planted pine is negligible. The only remaining individuals are along replicate block edges, outside of vegetation and CWD plots of this study.

No predisturbance inventory was conducted on the tornado-disturbance study area. However, Newbold (1994) cited the predisturbance stand as a two-aged oak-hickory (*Quercus* spp.-*Carya* spp.) forest. The two-age classes consisted of a 100- to 120-year-old cohort of mostly oak species and a 60- to 80-year-old cohort that consisted of a significant amount of yellow-poplar (*Liriodendron tulipifera*). Timber cruises and preharvest inventories conducted in surrounding stands prior to tornado disturbance also indicate that the study area was an oak-hickory forest. More specifically, the stand was dominated by red oak species (mainly *Q. rubra*), white oak (*Q. alba*), and chestnut oak (*Q. prinus*). Inventories indicate that yellow-poplar, hickory, and miscellaneous hardwood and pine species also existed within the stand, but to a lesser extent than the oak species.

METHODS

Vegetation

Vegetation data were collected beginning in August 2007 within the tornado-disturbed area, and July 2008 in the clearcut area. Within a single treatment some variability exists between treatment areas due to different slope positions, aspects, and, within the tornado treatment, level of disturbance caused by the tornado. Although this variability exists, analysis has been conducted at the treatment level. Pseudoreplication occurs as each plot is statistically treated as a replicate considering low number of true replicates resulting in low statistical power. The assumption of true replication in the complete randomized design is admittedly not satisfied because natural disturbances such as tornadoes cannot be replicated.

Plot setup was as follows: 27 plots were established in the tornado-disturbed study areas, and 10 plots were established in the clearcut study area. Within the tornado treatment, plots were systematically laid out with a series of transect lines connecting plots, with a random starting point. Thirteen plots were established in the northern tornado study cell, and 14 plots were established in the southern study cell. Within the clearcut treatment, two vegetation plots were measured at each of the five 1-acre SCC blocks. The two plots were positioned 40 feet (12.2 m) north and south of the SCC block center stake along the division line between eastern white pine and loblolly pine plantings. If a planted pine was inside the plot or was determined to have an effect on the natural development of the vegetation within the plot, the plot was systematically moved 54 feet (16.5 m) parallel to the block boundary to the side of the block planted with eastern white pine. Since planted white pine survival was so low in the SCC, no plots were adversely affected. For both treatments, plots landing on a skid trail, road, old-log landing or treatment

area boundary were moved one chain (66 feet or 20.1 m) perpendicular to the transect line away from the obstruction.

At each plot, three nested, fixed-area plots were established to measure the overstory, midstory, and understory. Percent slope, aspect, and slope position were recorded at the center stake of each plot. Percent slope was assessed using a Suunto clinometer. Aspect was categorized as being one of the four cardinal directions or a midpoint between two of the four cardinal directions, i.e., N, NE, E, SE, etc. Slope position was classified as being on the upper, middle, or lower third of the slope. These data were used as explanatory variables for variability between plots within treatments.

Each vegetation plot was broken into five strata categories:

- Overstory vegetation
- Midstory vegetation
- Understory woody vegetation >4 feet (1.2 m) tall
- Understory woody vegetation <4 feet (1.2 m) tall (presence/absence only)
- Understory herbaceous vegetation (presence/absence only)

One-tenth-acre (0.04-ha) overstory plots were centered along the predetermined transect line. At each overstory plot, trees with a diameter at 4.5 feet (1.4 m) from the ground (d.b.h.) ≥4.5 inches (11.4 cm) were tallied, and species and d.b.h. were recorded for e ach tree. D.b.h. was measured by 1-inch diameter classes where the ith inch-class ranged from [i–1].5 to [i].4. Midstory plots were 0.02 acres (0.008 ha) with the same plot center as the overstory plot. At each midstory plot, trees from 1.5 to 4.4 inches (3.8 to 11.2 cm) d.b.h. were measured and recorded by species and diameter class. Two understory plots were measured at each overstory plot. The center of each of the two plots was 0.33 chains (6.7 m) perpendicular to and on either side of the transect line. The plots measured 0.001 acres (0.0004 ha) and every woody stem measuring <1.5 inches (3.8 cm) d.b.h. was recorded. Woody stems with a height ≥4 feet (1.2 m) were tallied by species. All woody plants <4 feet (1.2 m) as well as herbaceous plant species were recorded by species.

Coarse Woody Debris

In the spring of 2008, CWD was evaluated using the line intersect method, described by Waddell (2002). For every downed tree along a transect line, diameter at large and small end and length were measured. CWD measurement parameters included density, volume, and total biomass based on formulas derived from De Vries (1973) and presented by Waddell (2002) and Woodall and Monleon (2007). All of the following methods for CWD data collection are based on Waddell (2002).

CWD plots were placed at the same plot location as the vegetation plots for both treatments. Each plot contained three transect lines, 37.2 feet (11.3 m) long, from the center of the circular plot. Transect lines were oriented at 0, 135, and 225 degrees. Transect lines were traversed and a piece of CWD was measured if:

1. The central longitudinal axis of the piece intersected the transect
2. The diameter at the point of the intersections was at least 5 inches (12.7 cm)
3. The piece length was at least 3.3 feet (1 m)
4. The piece was not decayed to the point of having no structural integrity

In situations where large limbs from the main bole of the tree intersected the transect line or a tree forked and both forks crossed the transect line, they were treated as two separate pieces. In this instance, the larger diameter stem was considered the main bole of the tree, and smaller segment(s) were measured to the main bole as a separate piece.

For each CWD piece crossing the transect line, diameter at the large end and small end, and the total length were measured, and a decay class of 1 to 5 was assigned to each CWD piece (Waddell 2002) (table 1). Cubic-foot-volume for a single CWD piece was first calculated using the following equation:

$$\text{Volume of a log: } V_{ft} = [(\pi/8)(D_S^2 + D_L^2)l]/144 \qquad (1)$$

where
V_{ft} = the volume (cubic feet)
D_S = the small-end diameter (inches)
D_L = the large-end diameter (inches)
l = the piece length (feet)

Table 1—Decay-reduction constants for coarse woody debris pieces by species group in the tornado disturbance treatments at the University of Tennessee Forest Resources Research and Education Center in Oak Ridge, TN

	Species group	
Decay class	Softwood	Hardwood
1	1	1
2	0.84	0.78
3	0.71	0.45
4	0.45	0.42
5	0	0

Source: Waddell (2002, table 4).

Per-unit-area attributes were computed based on De Vries' (1973), Waddell's (2002), and Woodall and Monleon's (2007) formulas:

$$\text{Volume (ft}^3\text{/acre)} = (\pi/2L)(V_{ft}/I_i)f \quad (2)$$

$$\text{Density (logs/acre)} = (\pi/2L)(1/I_i)f \quad (3)$$

$$\text{Biomass (tons/acre)} = (1/2000)*[(\pi/2L)(V_{ft}/I_i)f]*BD*DC \quad (4)$$

where

f = the per-unit-area expansion factor [43,560 square feet per acre (10 000 m^2/ha)]

L = the transect length [37.2 feet (11.3 m)]

BD = the bulk density of hardwood [28.7 pounds per cubic foot (459.7 kg/m^3)] (Woodall and Monleon 2007, appendix 7.3)

DC = the decay-reduction constant (table 1)

Because species group was not recorded during data collection and most of the tree composition at the time of the storm was in the hardwood species, hardwood decay-reduction constants were used.

Data Analysis

Upon completion of data collection, vegetation and CWD data were analyzed for multiple stand and community characteristics. Species diversity (H') was calculated for woody vegetation >4 feet (1.2 m) tall, midstory, overstory, and combined midstory/overstory using the Shannon index (Shannon 1948).

Midstory and overstory density and basal area were calculated. Overstory and total basal area were reported for each treatment with associated stand errors to evaluate variability within treatments. Relative density and relative dominance (basal area) were calculated to determine species importance values (IV). Relative frequency was omitted from the standard IV calculation (Curtis and McIntosh 1951). If included, it would have caused bias based on the low abundance of many species in the overstory strata and spatial variability associated with the tornado disturbance. Therefore, the total IV summation index was calculated out of 200 instead of 300. Midstory and overstory IVs were calculated as separate strata, then together to determine species importance of all trees 2 inches (5.1 cm) and greater in diameter. Black oak (*Q. velutina*), scarlet oak (*Q. coccinea*), and northern red oak were grouped together as "red oaks"; chinquapin oak (*Q. muhlenbergii*), chestnut oak, and white oak were grouped as "white oaks"; and red elm (*Ulmus rubra*), winged elm (*U. alata*), and American elm (*U. americana*) were grouped as "elms."

Once IVs were calculated, species scoring <4.0 were dropped from the analysis. This value was selected because there was a natural break in the dataset, where species scoring <4.0 were usually observed on a single occasion within a given treatment. The five species with the highest IVs for each treatment were then used in diameter distribution curves and species density tables.

All statistical analyses were performed to test the null hypothesis, stating that there is no significant difference between treatments. To test this, a Mann-Whitney analysis (Mann and Whitney 1947) was computed where plot means were ranked and an analysis of variance was performed on the rank to determine statistical significance between treatments. Shannon (H') index scores, IVs, and CWD volume, density, and biomass were all tested in this manner by treatment.

RESULTS

Variability

Total basal area and the associated errors were similar between the two treatments, having high variability (table 2). Overstory basal area was much greater in the tornado treatment, but had a standard error that was 2.5 times greater than the clearcut treatment, illustrating the high variability in the tornado treatment.

Importance Values

Thirteen species/species groups had an IV >4.0 in either the tornado or clearcut areas. Of those 13 species only 3 species showed significant differences (α = 0.05) between tornado and clearcut treatments: blackgum (*Nyssa sylvatica*), redbud (*Cercis canadensis*), and sugar maple (*Acer saccharum*) (table 3). Yellow-poplar, red maple (*A. rubrum*), black cherry (*Prunus serotina*), sourwood (*Oxydendrum arboreum*), and the white oak group have the five highest IV for both tornado and clearcut treatments. However, red maple has the highest IV for tornado and yellow-poplar has the highest IV for clearcut treatments. Although large differences occur for IVs between species such as red oaks, smooth sumac (*Rhus glabra*), eastern white pine, and yellow-poplar there are no significant differences between the two treatments for those species.

Tree Density and Structure

Tree density curves for tornado and clearcut treatments had similar shape but different densities, especially in the larger diameter classes. In the 2- to 7-inch diameter classes, the clearcut treatment averaged a higher stem density. However, all diameter classes >7 inches (where trees were present)

Table 2—Overstory and total basal area calculations for tornado and clearcut treatments at the University of Tennessee Forest Resources Research and Education Center in Oak Ridge, TN

Mean basal area	Tornado	Clearcut
	------- *square feet per acre* -------	
Overstory	63.30	25.91
(Standard error)	7.20	2.85
Total	90.11	82.99
(Standard error)	6.05	5.69

Table 3—Importance values by species based on a maximum 200 possible value in the tornado disturbance treatments at the University of Tennessee Forest Resources Research and Education Center in Oak Ridge, TN

Species	Tornado	Clearcut	*p*-value
Black cherry	24.07	28.36	0.4392
Blackgum*	8.25	1.4	0.0073
Flowering dogwood	10.39	5.85	0.0883
Hickory spp.	5.81	4.87	0.8083
Redbud*	7.6	0	0.0072
Red oaks	9.36	3.83	0.7089
Red maple	38.13	42.05	0.8934
Smooth sumac	0.39	4.13	0.2592
Sourwood	14.46	23.89	0.0541
Sugar maple*	4.69	0.39	0.0411
White oaks	21.58	19.83	0.7248
Eastern white pine	4.7	0	0.0796
Yellow-poplar	36.65	55.35	0.2504

P-values are reported (* signifies $P \leq 0.05$).

had greater stem densities in the tornado treatment (figs. 2 and 3). No trees in the 10-inch class or greater were found in the clearcut treatment.

The diameter distributions of both tornado and clearcut treatments indicated that red maple is a dominant species in the 2-inch class (table 4). This pattern continues in the tornado treatment for red maple into the 3-inch class as well. Yellow-poplar, which has the highest overall density in the tornado area, has the greatest density from the 5- to 18-inch diameter classes, as well as the 25-, 28-, and 30-inch classes (table 4). No trees exist for red maple beyond the 13-inch class, for black cherry beyond the 11-inch class, and sourwood beyond the 10-inch class. White oak, however, has a similar distribution shape as yellow-poplar from the 11- to 30-inch diameter classes with a lesser density.

In the clearcut, red maple density quickly dropped and yellow-poplar had the greatest density in all other diameter classes (table 4). Sourwood was competitive in the 2- and 3-inch classes, but no trees existed beyond the 5-inch class. Black cherry had its highest density in the 4-inch class and had a slightly lesser density, but a similar distribution curve, as yellow-poplar in the 4- to 9-inch classes. White oak had the lowest overall density of the five selected species, but had the greatest density in the 7- to 9-inch classes (table 4).

Shannon Diversity

Tornado areas had higher species diversity than clearcut areas in the overstory and combined midstory/overstory strata while having lower diversity in the midstory and understory strata (table 5). Differences in diversity were only significant in the overstory vegetation strata between treatments.

Coarse Woody Debris

CWD volume, density, and biomass were all significantly different between treatments (table 6). CWD volume was over four times greater in the tornado treatment. Similarly, CWD density was 2.5 times higher and CWD biomass was over seven times greater in the tornado treatment compared to the clearcut treatment.

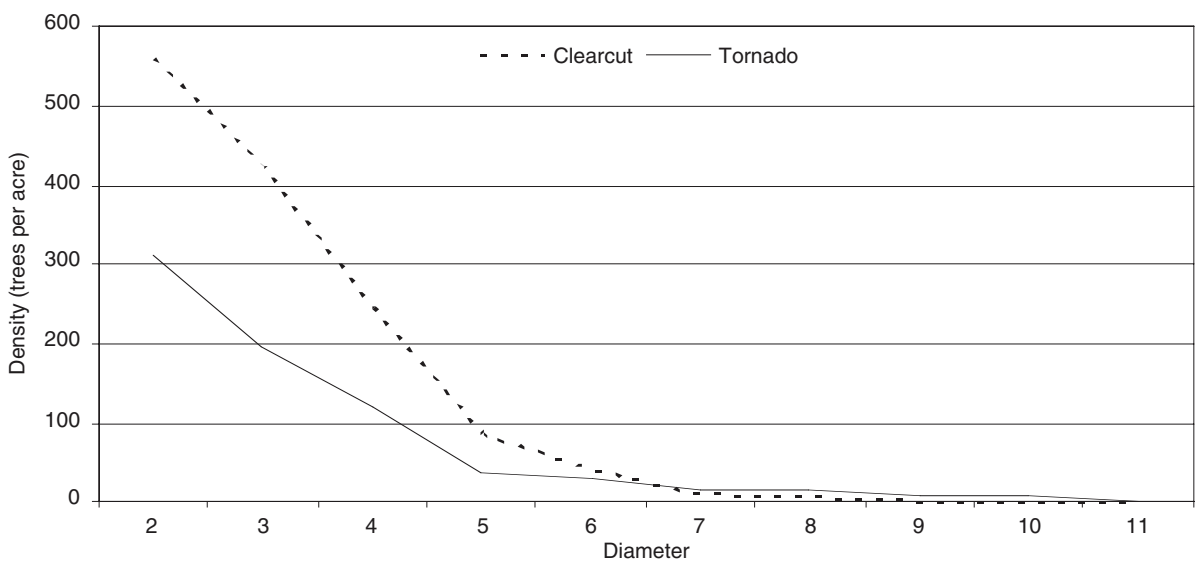

Figure 2—Tree density (trees per acre) for tornado and clearcut treatments for 2- to 11-inch trees at the University of Tennessee Forest Resources Research and Education Center in Oak Ridge, TN.

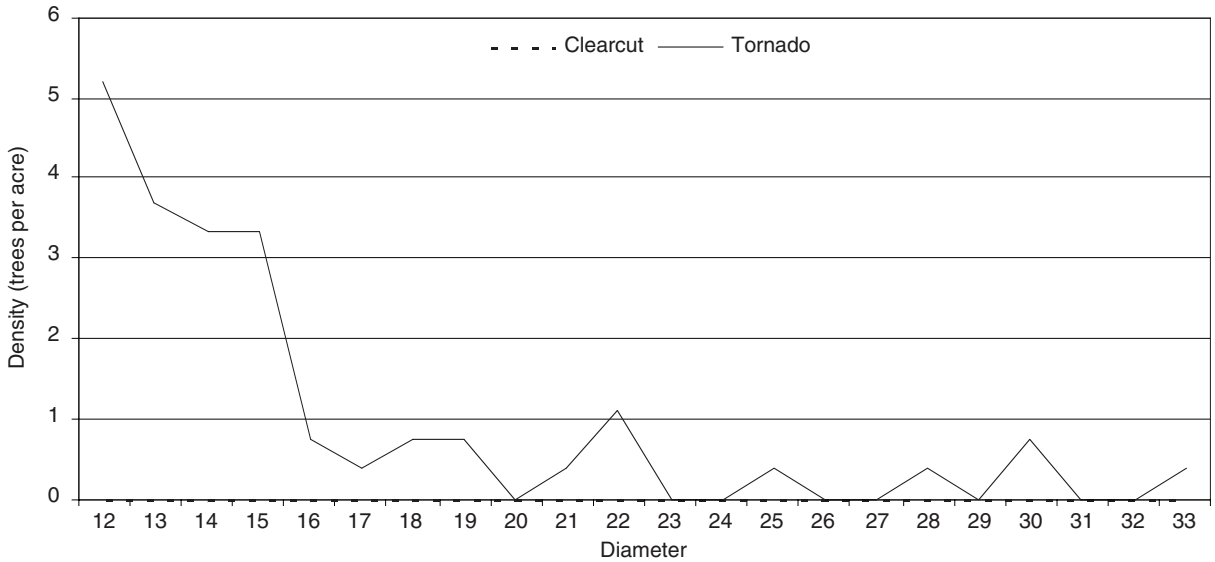

Figure 3—Tree density (trees per acre) for tornado and clearcut treatments for trees >12 inches at the University of Tennessee Forest Resources Research and Education Center in Oak Ridge, TN.

DISCUSSION

Species Composition

Thirteen species had an IV >4.0. The null hypothesis can only be rejected for three species: blackgum, redbud, and sugar maple. All three of these species are considered to be shade tolerant and had significantly greater importance in the tornado areas compared to the clearcut areas. Blackgum and sugar maple had an IV almost 6 and 12 times greater, respectively, in the tornado area. Redbud was not detected in the clearcut areas and therefore had no IV.

The comparison between tornado and clearcut disturbances expressed a relationship between the shade tolerance of the three significantly different species and the two different disturbance types. The clearcut was a stand-initiating disturbance. The tornado was considered to be an incomplete stand-scale disturbance, though one of relatively high intensity, creating large canopy gaps and few widely spaced residual trees. However, enough trees remained standing after the tornado disturbance to create light conditions significantly more suitable for shade tolerants like blackgum, redbud, and sugar maple (Burns and Honkala 1990).

Since the tornado created light conditions more suitable for shade-tolerant species, one would expect to see the inverse, where shade-intolerant species were significantly more important in the clearcut areas. The IV for many of the shade-intolerant species, e.g., yellow-poplar, black cherry, and white oaks (Burns and Honkala 1990), were much greater for both treatments than the shade-tolerant species. Their relatively high density and importance indicate that light conditions were suitable for intolerant species to grow in both treatments. Therefore one can deduce that absence of shade, in conjunction with higher stem densities in the clearcut made it more difficult for shade-tolerant species to compete. On the other hand, the few residual trees left in the tornado areas created more shade and lower stem densities, allowing shade-tolerant species to remain as part of the species composition. Miller and others (2006) support the theory that residual trees have an effect on the amount of light reaching the forest floor and thus the species composition of the new cohort.

Tree Density and Structure

No statistical analysis was conducted to test for treatment differences for diameter distribution and thus, density. Analysis could not be done for all diameter classes due to lack of residual overstory trees in the clearcut. Although the clearcut treatment was harvested almost 4 years before the tornado disturbance, total stem density is nearly two times greater in the clearcut areas. However, data in table 4 showed that proportions of yellow-poplar, red maple, black cherry, and white oaks were similar across both treatments.

The greatest differences in stem density occurred in smaller diameter classes (fig. 2), where the clearcut had roughly twice the density of the tornado areas. In the 7-inch diameter class, tornado densities became greater than clearcut densities and continue to be greater for the remaining diameter classes. Tornado disturbance has been noted to provide a greater range of diameter classes compared to clearcut disturbances (Price and others 1998). Similarly, no trees >9 inches (22.9 cm) were measured in the clearcut areas, while tornado areas had a greater range of diameters, with the largest tree having a 33-inch (83.8-cm) diameter. Both treatments have negative exponential curves, but the tornado areas have more trees

Table 4—Tree densities by diameter class for tornado and clearcut treatments at the University of Tennessee Forest Resources Research and Education Center in Oak Ridge, TN

	Tornado treatment					
D.b.h.	YEPO	REMA	BLCH	SOWO	WO	Total
	trees per acre					
2	28	96	50	20	13	313
3	31	50	31	17	6	196
4	22	17	24	7	2	119
5	38	8	6	4.8	9	38
6	29	6	3.3	2.2	7	29
7	14	3	1.1	3	3.7	14
8	14	4.4	0	2.2	2.6	14
9	8	1.5	0	0.7	2.6	8
10	8	0	0.7	0.7	2.6	8
11	3	0.4	0.4	0	1.1	3
12	5	0.4	0	0	3	5
13	3.7	0.7	0	0	0	4
14	3.3	0	0	0	1.9	3
15	3.3	0	0	0	0.7	3
16	0.7	0	0	0	0.4	1
17	0.4	0	0	0	0	0
18	0.7	0	0	0	0	1
19	0.7	0	0	0	0.7	1
20	0	0	0	0	0	0
21	0.4	0	0	0	0.4	0
22	1.1	0	0	0	0	1
23	0	0	0	0	0	0
24	0	0	0	0	0	0
25	0.4	0	0	0	0	0
26	0	0	0	0	0	0
27	0	0	0	0	0	0
28	0.4	0	0	0	0	0
29	0	0	0	0	0	0
30	0.7	0	0	0	0	1
Total density	216.3	187	117.4	58.1	56.3	763
Relative density (%)	28.40	24.50	15.40	7.60	7.40	
	Clearcut treatment					
D.b.h.	YEPO	REMA	BLCH	SOWO	WO	Total
	trees per acre					
2	135	255	20	65	55	560
3	135	65	35	90	25	425
4	70	55	65	30	20	245
5	29	11	24	1	15	87
6	14	8	11	0	5	41
7	3	1	0	0	4	10
8	1	0	2	0	2	7
9	1	0	1	0	0	2
Total density	388	395	158	186	126	1377
Relative density (%)	28.20	28.70	11.50	13.50	9.20	

YEPO = yellow-poplar; REMA = red maple; BLCH = black cherry; SOWO = sourwood; WO = white oak group.

Table 5—Shannon H' values by treatment reported for each vegetation strata in the tornado disturbance treatments at the University of Tennessee Forest Resources Research and Education Center in Oak Ridge, TN

Vegetation strata	Tornado	Clearcut	*P*-value
Understory >4 feet	0.663	0.788	0.6218
Midstory	1.337	1.475	0.1978
Overstory	1.505	1.323	0.0144
Midstory/overstory	1.785	1.646	0.4002

Table 6—Coarse woody debris volume, density, and biomass by treatment at the University of Tennessee Forest Resources Research and Education Center in Oak Ridge, TN

CWD attribute	Tornado	Clearcut	*P*-value
Volume (cubic feet per acre)	635.9	155.2	0.0057
Density (logs per acre)	108.1	42.4	0.0129
Biomass (tons per acre)	0.666	0.089	0.0023

intermittently spaced in the larger diameter classes. Although sparse, the trees in the larger diameter classes are from the residual stand. Since the residual stand was two-aged, the tornado areas are now in the complex stage of development. The complex stage is reached when gaps created at different points in time result in a multiaged stand (Oliver and Larson 1996). On the other hand, the clearcut areas are currently in the stem exclusion stage as the clearcut itself was a complete stand-initiating disturbance.

In both areas, red maple had similar diameter distribution, with high densities in the 2- and 3-inch classes. However, the number of red maple stems quickly descended to the densities of the other five important species with increasing size class. Red maple has a tendency to vigorously stump sprout. Many of these sprouts would likely decrease in the stand in the near future as intraspecific competition causes mortality of smaller stems (Burns and Honkala 1990). Considering only the new cohort of trees, yellow-poplar had the highest densities in the remaining diameter classes. White oaks existed in relatively low densities in all but the larger diameter classes for each treatment.

Species Diversity

Differences in H' diversity were not detected in any of the strata except in the overstory; therefore, the null hypothesis can be rejected for the overstory stratum. Here, the tornado areas had significantly greater diversity than the clearcut, due to a greater number of residual large diameter trees and more tree species.

Coarse Woody Debris

The tornado treatment had significantly greater CWD volume, density, and biomass; hence the null hypothesis can be rejected for all three attributes. In the tornado areas, there was more than four times the CWD volume, more than two times the CWD density, and more than seven times the CWD biomass. Much of the fine woody debris in the clearcut has decayed since the time of harvest and most of the large or CWD was removed at part of the harvest prescription. Pole stands (the category the clearcut treatment is most comparable to) tend to have the lowest CWD loads of all the even-aged developmental stages (McCarthy and Bailey 1994) due to the lack of large CWD pieces left after harvest (Price and others 1998).

The nature of each disturbance led us to expect that there would be more CWD loads in the tornado area which was supported by the data. Both CWD volume (155.2 and 157 cubic feet per acre, respectively) and CWD biomass (0.089 and 0.073 tons per acre) were similar between these two treatments. Although the harvests between the clearcut and salvage/slash were similar (removing to the 2-inch (5.1-cm) diameter class), small diameter trees were removed from the clearcut to allow for the planting of pines.

MANAGEMENT IMPLICATIONS

As the value of forest management becomes more recognized as an integral part of ecosystem management, there is a heightened interest in how silviculture can be used to emulate natural disturbances. Data from this study indicate that clearcuts and tornados are structurally two different disturbances. Although species importance and relative density are similar in both treatments, the tornado area is more structurally diverse. Overall, diameter distributions had greater ranges and CWD loading was greater in the tornado areas.

If silvicultural methods are used to imitate natural disturbance, a more irregular and erratic marking prescription is needed to ensure leaving residual trees that will have similar effects as the residual trees in the tornado areas. Diameter distributions and field observations for the tornado area show that trees from all crown classes must be left to imitate such conditions. Furthermore, to emulate the CWD loads from a tornado disturbance, some downed trees of all sizes would need to be left onsite.

From a timber management perspective, lower stem densities seen in the tornado areas will likely result in shorter, lower grade trees because early competition played less of a part in individual tree growth (Clatterbuck and Hodges 1987).

Although relative densities and IVs for individual species are similar, density-related competition occurs to a lesser extent in the tornado areas.

RESEARCH IMPLICATIONS

Some level of warning should be given along with the results of this research. The tornado areas had high levels of variability in them due to the nature of wind disturbances. Silvicultural clearcuts, however, tended to have less variability associated with them because such a disturbance is fairly constant across the stand.

As discussed in the "Methods" section, vegetation measurements were conducted in the same manner at each plot location for both treatments. However, the number of plots and plot layout was different for each treatment because of the small area of the SCC areas from the mixed pine-hardwood study (Andrews 1995). The low number of plots from the clearcut treatment compared to tornado area, in conjunction with the variability in the tornado treatment may have increased the chances making type II error. For instance, table 3 indicated a difference of almost 20 between treatments of yellow-poplar IVs. The theories behind stand dynamics would indicate that yellow-poplar should have more success in a complete stand-initiating disturbance such as a clearcut versus an incomplete disturbance such as a tornado, yet no significant differences were detected.

Conversely, the aforementioned difficulties in experimental design should strengthen the importance of detected differences. This is especially true in the vegetation stand characteristics, i.e., IVs and Shannon H', where both low number of clearcut plots and tornado-treatment variability could have hidden stand differences. For this reason, the dissimilarities discovered between these two treatments signify that differences are apparent for those stand characteristics.

LITERATURE CITED

Adreadis, D.K.W. 1995. The effects of natural and anthropogenic disturbance on small mammals in a forest ecosystem of eastern Tennessee. Knoxville, TN: University of Tennessee. 94 p. M.S. thesis.

Andrews, V. 1995. Effects of four pine-hardwood regeneration techniques on first year regeneration and wildlife habitat. Knoxville, TN: University of Tennessee. 165 p. M.S. thesis.

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

Canham, C.D.; Loucks, O.L. 1984. Catastrophic windthrow in the presettlement forests of Wisconsin. Ecology. 65(3): 803–809.

City-Data. 2007. Oak Ridge, TN City-Data. http://www.city-data.com/city/Oak-Ridge-Tennessee.html. [Date accessed: May 4].

Clatterbuck, W.K.; Hodges, J.D. 1987. 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: 12–18.

Clinton, B.D.; Baker, C.R. 2000. Catastrophic windthrow in the Southern Appalachians: characteristics of pits and mounds and initial vegetation responses. Forest Ecology and Management. 126: 51–60.

Curtis, J.T.; McIntosh, R.P. 1951. An upland forest continuum in the prairie-forest border region of Wisconsin. Ecology. 32(2): 476–496.

De Vries, P.G. 1973. A general theory on line intersect sampling with application to logging residue inventory. Wageningen, The Netherlands: Mededelingen Landbouwhogeschool. (73): 11.

Mann, H.B.; Whitney, D.R. 1947. On a test of whether one of two random variables is stochastically larger than the other. Annals of Mathematical Statistics. 18: 50–60.

McCarthy, B.C.; Bailey, R.R. 1994. Distribution and abundance of coarse woody debris in a managed forest landscape of the central Appalachians. Canadian Journal of Forest Research. 24: 1317–1329.

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(3): 241–251.

Moneymaker, R.H. 1981. Soil survey of Anderson County, Tennessee. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service; U.S. Government Printing Office. 165 p.

National Oceanic and Atmospheric Administration. 2007. National Weather Service forecast office, Morristown, TN. http://www.srh.noaa.gov/mrx/tornado/anderson.php. [Date accessed: December 17].

Natural Resources Conservation Service. 2007. Web soil survey, official soil series descriptions, Fullerton deries. http://www2.ftw.nrcs.usda.gov/osd/dat/F/FULLERTON.html. [Date accessed: December 17].

Newbold, C. 1996. The effects of tornado and clearcut disturbances on breeding birds in a Tennessee oak-hickory (*Quercus-Carya* spp.) forest. Knoxville, TN: University of Tennessee. 116 p. M.S. thesis.

Oliver C.D.; Larson, B.C. 1996. Forest stand dynamics. New York: Wiley. 520 p.

Peterson, C.J.; Pickett, S.T.A. 1995. Forest reorganization: a case study in an old-growth forest catastrophic blowdown. Ecology. 76(3): 763–774.

Price, K.; Brewer, L.; Poirier, N. [and others]. 1998. Windthrown or clearcut—what's the difference? Northwest Science. 72(2): 30–33.

Shannon, C.E. 1948. A mathematical theory of communication. Bell System Technical Journal. 27: 379–423, 623–656.

Waddell, K.L. 2002. Sampling coarse woody debris for multiple attributes in extensive resource inventories. Ecological Indicators. 1(3): 139–153.

Woodall, C.W.; Monleon, V.J. 2007. Sampling protocol, estimation, and analysis procedures for the downed woody materials indicator of the FIA program. Gen. Tech. Rep. NRS-22. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 68 p.

LONG-TERM STAND GROWTH AFTER HELICOPTER AND GROUND-BASED SKIDDING IN A TUPELO-CYPRESS WETLAND: 21-YEAR RESULTS

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Abstract—Three disturbance treatments were implemented on a tupelo-cypress forested wetland in southwestern Alabama on the Tensaw River in 1986: (1) clearcutting with helicopter log removal (HELI), (2) HELI followed by rubber-tired skidder traffic simulation (SKID), and (3) HELI followed by removal of all vegetation during the first two growing seasons via glyphosate herbicide application (GLYP). At year 2 the SKID treated areas were wetter and had dramatic negative changes in soil characteristics and hydrology. By year 7 these negative impacts of skidder traffic were ameliorated. Further, the SKID treatment resulted in increased regeneration of *Nyssa aquatica*. Water quality was not adversely impacted by any disturbance treatment and sediment accumulation was actually improved by both HELI and SKID treatments. At year 21 the SKID and HELI treatments are well stocked with over 4,000 stems/ha, but the GLYP plots have little woody plant regeneration. Despite large stand composition differences at year 7, the SKID and HELI plots are becoming more similar. This is largely due to a decreasing *Salix nigra* component. Aboveground tree biomass is also similar for the SKID and HELI treatments with no significant differences for any species.

INTRODUCTION

Forested wetlands, such as bottomland hardwoods, are valued by society because of their unique influence on ecological functions, such as hydrology, water quality, nutrient cycling, and wildlife habitat (Daniels and Gilliam 1996, Klapproth 1996, Sheridan and others 1999, Walbridge 1993, Welsch 1996). Timber harvesting in forested wetlands is viewed with concern because it has the potential to influence these ecological functions. After the passage of the Clean Water Act of 1972 most Southeastern States developed best management practices (BMP) to address these concerns. However, little long-term research has been conducted in forested wetlands that address the effects of harvesting techniques on ecological function or stand development and growth. The purpose of this paper is to summarize the 21-year stand growth results from an on-going study comparing helicopter and ground-based skidding after clearcut logging in a tupelo-cypress (*Nyssa* spp.-*Taxodium* spp.) wetland.

Site Description

The study site is located within the Mobile-Tensaw River Delta along the western bank of the Tensaw River approximately 4.5 km southwest of Stockton, AL. Sporadic baldcypress (*T. distichum*) harvesting occurred on the sites as early as the 1700s with at least two additional harvests in the 1860s and 1915. Evidence of past pullboat logging operations is evident with pullboat channels from 30 to 150 cm in depth every 20 to 50 m. Climate in the area is subtropical with a mean annual temperature of 20 °C, 250 frost-free days, and <3 weeks below freezing. Average annual precipitation is evenly distributed and at 1600 mm/yr (Riccio and others 1973). The Tensaw River is a freshwater river at the study site, but has semidiurnal tidal influence, which occasionally allows the pullboat channels to serve as a direct conduit to the river. During the summer months, when evapotranspiration is maximized, the water table ranges from 25 cm above the surface to 50 cm below the surface. The soils in the area are very poorly drained fluvial sediments classified as Levy silty clay loam. Prior to our harvest the site was a two-age stand with >80 percent of stems at 72 years of age and <20 percent comprised of older residual stems. Water tupelo (*N. aquatica*) comprised 85 percent of the stems, with 10 percent baldcypress and 4 percent Carolina ash (*Fraxinus caroliniana*).

METHODS

Three disturbance treatments were included in the research design with nondisturbed areas retained to serve as a reference (REF). The entire disturbance area was clear-felled with chainsaws down to a 5-cm diameter, resulting in a biological clearcut. Then a Bell 205 helicopter was used to fly all merchantable stems from the area. This constitutes the helicopter harvest treatment (HELI). A Franklin 105 skidder equipped with rubber tires was used to traffic previously harvested HELI areas in order to simulate typical ground-based harvesting. The skidder treatment (SKID) resulted in 52 percent of the area trafficked to an average depth of 30 cm. In order to separate coppice regeneration from total regeneration an additional disturbance treatment (GLYP) was implemented that included complete suppression of all vegetation regrowth after helicopter harvest using glyphosate for 2 years. Treatments were arranged as three squares of 3 by 3 Latin squares providing nine replications of each treatment with nine pseudoreplications of the REF treatment. We used the Tukey-Kramer multiple comparison procedure to test for treatment differences with an alpha level of 0.05. More detailed site descriptions, preharvest characterizations, and descriptions of past methods used can be obtained from papers developed from research conducted at this site (Aust and Lea 1991, 1992; Aust and others 1989, 1990, 1991, 1997, 1998, 2006; Gellerstedt and Aust 2004; Gellerstedt and others 2002; Jackson and others 2004; Mader and others 1989a, 1989b)

RESULTS AND DISCUSSION

Year 2

Initial results indicated that 2 years after harvesting, both SKID and HELI treatments had substantial effects on site hydrology, soil characteristics, water quality, nutrient cycling,

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and habitat. Soil properties and hydrology were generally negatively affected by the three disturbance treatments. Water table depth during the growing season was raised; mechanical resistance was increased; saturated hydraulic conductivity, soil oxygen levels, and soil reduction-oxidation potential were decreased when compared to the REF treatment (Aust and Lea 1992). The greatest impacts were on the severely disturbed SKID treatment, which became substantially wetter, favoring regeneration of the more flood tolerant *N. aquatica*. Water-quality indices were either not affected or improved with disturbance (Aust and others 1991). Total Kjeldahl nitrogen and total phosphorus were similar for all periods and treatments. Sediment accumulations were greater on the SKID and HELI treatments indicating an improvement in water quality leaving the site. This improvement in sediment trapping was due to increased surface roughness on the SKID and HELI treatments.

During the first 2 years following disturbance, herbaceous and woody vegetation of the HELI and SKID treatments responded vigorously. The HELI and SKID treatments had similar patterns of net primary production aboveground biomass, total height, and numbers of desirable tree species (Mader and others 1989a, 1989b). Stocking in the SKID treatment favored the more flood tolerant *N. aquatica* with more *F. caroliniana* in the HELI treatment (Aust and others 1997). The general conclusion at age 2 was that the HELI and SKID treatments were well stocked, but there would be negative consequences on stand growth in the SKID treatment because of the negative effects on soil and hydrology (Aust and others 1997).

Year 7

Due to the changes in soil physical properties and hydraulic changes we predicted that the SKID treatment would have a negative effect on long-term site productivity. However, by year 7 the opposite of our prediction was evident. Most of the measured soil physical properties and hydraulic conditions had returned to REF levels. The SKID treatment continued to have a greater number of stems of *N. aquatica* and had a greater aboveground overstory biomass than the HELI treatment (Aust and others 1997). This recovery was due to rapid and robust coppice regeneration, canopy closure, return of high transpiration rates, sediment inputs, high site fertility, and soil shrink swell properties. This recovery is unique to this forest type and hydrological setting and should not be expected in other systems.

Sediment

Sediment accumulations were greater than the REF in all three disturbance treatments (table 1). Initially, the GLYP treatment was accumulating less sediment than the REF due to the treatment leaving the ground completely devoid of vegetation. However, once herbaceous vegetation rebounded, the GLYP treatment had the highest levels of sediment accumulation and significant differences with the other treatments until the last sediment measurement at year 16. The SKID and HELI treatments had increased sediment accumulation for approximately 6 years because of increased surface roughness with accumulations returning to REF levels at more recent measurement periods (Aust and others 2006). The increases in sediment accumulation due to site disturbance is an ameliorative mechanism that has helped the disturbed plots to recover and a positive benefit to the water quality that flows from the area.

Year 21 Stand Characteristics

At year 21, stocking levels are high for both the SKID and HELI treatments (table 2) with a mean of more than 4,000 stems/ha. The SKID treatment continues to have significantly more of the commercially desirable *N. aquatica* compared to the HELI treatment with 668 more stems/ha. The higher number of stems/ha of *F. caroliniana* in the HELI treatment compared to the SKID treatment, that were evident during earlier years, have been reduced, and the difference is no longer statistically significant. Black willow (*Salix nigra*) stocking in the HELI and SKID treatments has decreased dramatically since year 7 from 1,153 to 115 stems/ha in the HELI treatment and from 1,153 to 198 stems/ha in the SKID treatment. This indicates that the black willow is being replaced by other species in these treatments. This is expected because of *S. nigra*'s position as an early succession, pioneer tree species. Black willow has increased from 33 to 279 stems/ha since year 7 in the GLYP treatment indicating less

Table 1—Mean annual sediment accumulation by treatment for 1987 to 2002

	Sediment accumulation					
Treatment	1987–1988	1989–1993	1994–1996	1997–1998	1999–2002	Total
	cm/year					*cm*
REF	1.1 ab	0.8 a	1.4 a	0.6 a	0.5 a	13.6 a
HELI	2.2 b	1.6 b	2.6 a	0.8 a	0.7 a	24.6 bc
SKID	1.4 ab	1.3 ab	2.2 a	0.8 a	0.6 a	19.9 ab
GLYP	0.8 a	2.1 b	3.4 c	1.4 b	1.0 b	29.1 c

Values with the same letters are not significantly different (alpha = 0.05).

REF = nondisturbed areas retained to serve as a reference; HELI = helicopter harvest treatment; SKID = skidder treatment; GLYP = disturbance treatment that included complete suppression of all vegetation regrowth after helicopter harvest using glyphosate for 2 years.

Source: Gellerstedt and Aust (2004), Warren (2001).

Table 2—Mean stand characteristics 21 years after disturbance

Biometric/treatment	*Fraxinus caroliniana*	*Fraxinus profunda*	*Nyssa aquatica*	*Salix nigra*	*Taxodium distichum*	Total
Density (stems/ha)						
HELI	1392 a	1178 a	1070 a	115 a	305 ab	4110 a
SKID	1186 a	890 ab	1738 b	198 a	478 a	4695 a
GLYP	222 b	16 b	273 c	279 a	74 b	575 b
Diameter (d.b.h. cm)						
HELI	6.3 a	6.5 a	12.0 a	15.1 a	6.2 a	9.0 a
SKID	5.9 a	6.2 a	12.9 a	20.4 a	7.0 a	10.3 a
GLYP	6.8 a	4.4 a	9.7 a	16.9 a	12.7 a	10.7 a
Total height (m)						
HELI	7.7 a	8.0 a	9.6 a	13.5 a	5.5 a	8.3 a
SKID	8.1 a	7.3 a	9.9 a	12.6 a	6.9 a	9.0 a
GLYP	7.8 a	6.8 a	8.1 a	7.9 a	8.6 a	7.0 a
Overstory biomass (t/ha)						
HELI	11.8 a	10.7 a	90 a	18.7 a	6.4 a	139.4 a
SKID	9.1 a	7 b a	112.1 a	34.6 a	8.5 a	173.2 a
GLYP	3.0 b	0.1 b	15.2 b	40.9 a	5.1 a	70.9 b

Means with the same letter are not significantly different (alpha = 0.05).

HELI = helicopter harvest treatment; SKID = skidder treatment; GLYP = disturbance treatment that included complete suppression of all vegetation regrowth after helicopter harvest using glyphosate for 2 years.

competition from other tree species and their ability to colonize these sites. *T. distichum* has steadily increased in all three disturbance treatments since year 7 with 478 stems/ha in the SKID treatment and 305 stems/ha in the HELI treatment (table 2). Height and diameter of all species are similar with no statistically significant differences between treatments at year 21.

Year 21 Biomass

Aboveground tree biomass continues to be greatest in the SKID treatment at year 21 with 173.2 Mg/ha (table 2). The HELI treatment has 139.4 Mg/ha, but the difference is no longer statistically significant. Much of the difference between the two is still because of greater biomass of *N. aquatica* in the SKID treatment. There continues to be greater aboveground biomass of *F. caroliniana* in the HELI treatment compared to SKID treatment, but again this difference is no longer statistically significant. The lack of significant differences indicates that the two treatments are becoming more similar to each other in terms of aboveground tree biomass.

The GLYP treatment has substantially less biomass at year 21 than the other disturbance treatments. This treatment is still in an early successional stage with herbaceous plants and grasses dominating the plots. Stocking for all species in the GLYP treatment has increased steadily over the course of the study, but it is still understocked with 575 stems/ha and approximately half of the biomass of the SKID and HELI treatments. The GLYP plots are recovering slowly, stressing the importance of coppice regeneration and herbaceous competition for rapid site recovery after extreme disturbance such as agricultural production.

CONCLUSIONS

The negative effects of harvesting and ground-based skidding on the soils and hydrology of the HELI and SKID treatments returned to REF conditions by year 7. This rapid recovery after disturbance can be explained by the prevalence of coppice regeneration, the inherent fertility of the site, frequent sediment inputs, and the shrink swell nature of the soil. Water quality was not adversely impacted and sediment accumulation was actually improved by both HELI and SKID treatments.

The SKID treatment initially made the site wetter which favored the flood tolerant *N. aquatica*. However, differences between the treatments are diminishing. Helicopter and ground-based skidding have resulted in good stocking at year 21 with similar species mixes, stocking levels, and aboveground tree biomass. The stands in these two treatments are becoming more similar, particularly as *S. nigra* decreases. Differences between

treatments may also be diminishing due to increases in stand variability. Variability in the stands is increasing because of natural stand disturbances such as storm damage, disease, and flooding. This study has demonstrated the importance of continuing research over longer timelines. If conclusions were based on results from early measurement periods in this study, inappropriate and premature management suggestions would have been accepted.

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LITERATURE CITED

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

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

Aust, W.M.; Lea, R. 1992. Comparative effects of aerial and ground logging on soil properties in a tupelo-cypress wetland. Forest Ecology and Management. 50: 57–73.

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

Aust, W.M.; Mader, S.F.; Lea, R. 1989. Abiotic changes of a tupelo-cypress swamp following helicopter and rubber-tired skidder harvest. In: Proceedings of the fifth southern silvicultural research conference. Gen. Tech. Rep. SO-74. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 545–551.

Aust, W.M.; Mader, S.F.; Mitchell, L.J.; Lea, R. 1990. An approach to the inventory of forested wetlands for timber harvest impact assessment. Forest Ecology and Management. 33(34): 215–225.

Aust, W.M.; Miwa, M.; Schoenholtz, S.H.; Fristoe, C.T. 1998. Growth and development of water tupelo-baldcypress following helicopter and skidder harvesting: 10 year results. In: Proceedings of the ninth southern silvicultural research conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 363–367.

Aust, W.M.; Schoenholtz, S.H.; Zaebst, T.W.; Szabo, B.A. 1997. Recovery status of a baldcypress-tupelo wetland seven years after harvesting. Forest Ecology and Management. 90: 161–169.

Daniels, R.B.; Gilliam, J.W. 1996. Sediment and chemical load reduction by grass and riparian buffers. Soil Science Society of America Journal. 60: 246–251.

Gellerstedt, P.; Aust, W.M. 2004. Timber harvesting effects after 16 years in a tupelo-cypress swamp. In: Proceedings of the 12th southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 524–527.

Gellerstedt, P.; Aust, W.M.; Warren, S. 2002. Fifteen year patterns of sediment removal by a riparian wetland as influenced by harvesting [CD-ROM]. In: Proceedings of the forestry best management practices research symposium. Atlanta: National Council for Air and Stream Improvement.

Jackson, C.R.; Sun, G.; Amatya, D. [and others]. 2004. Fifty years of forest hydrology in the Southeast. In: Ice, G.G.; Stednick, J.D., eds. A century of forest and wildland watershed lessons. Bethesda, MD: Society of American Foresters: 33–112. Chapter 3.

Klapproth, J. 1996. The benefits of riparian buffers. Forest Landowner. 55(6): 28–31.

Mader, S.F.; Aust, W.M.; Lea, R. 1989a. Changes in functional values of a forested wetland following timber harvesting practices. In: Hook, D.D.; Lea, R., eds. Proceedings of the symposium: The forested wetlands of the Southern United States. Gen. Tech. Rep. SE-50. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 149–154.

Mader, S.F.; Aust, W.M.; Lea, R. 1989b. Changes in net primary productivity and cellulose decomposition rates in a water tupelo-baldcypress swamp following timber harvest. In: Proceedings of the fifth southern silvicultural research conference. Gen. Tech. Rep. SO-74. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 539–543.

Riccio, J.F.; Hardin, J.D.; Lamb, G.M. 1973. Development of a hydrologic concept for the greater Mobile metropolitan-urban environment. Geol. surv. of Alabama Bull. 106. Tuscaloosa, AL. [Number of pages unknown].

Sheridan, J.M.; Lowrance, R.; Bosch, D.D. 1999. Management effects on runoff and sediment transport in riparian forest buffers. Transactions of the ASAE: 42(1): 55–64.

Walbridge, M.R. 1993. Functions and values of forested wetlands in the Southern United States. Journal of Forestry. 91(5): 15–19.

Welsch, D.J. 1996. Riparian forests buffers: function and design for protection and enhancement of water resources. NA-PR-07-91. [Broomall, PA]: U.S. Department of Agriculture Forest Service, Northeastern State and Private Forestry. [Number of pages unknown].

A PRELIMINARY TEST OF ESTIMATING FOREST SITE QUALITY USING SPECIES COMPOSITION IN A SOUTHERN APPALACHIAN WATERSHED

W. Henry McNab and David L. Loftis[1]

Abstract—Characteristic arborescent communities of mesophytic or xerophytic species have long been recognized as indicative of forest site quality in the Southern Appalachians, where soil moisture availability is the primary environmental variable affecting productivity. But, a workable quantitative system of site classification based on species composition is not available. We devised a prototype expert system by assigning a relative moisture weight to upland forest tree species according to their position of modal occurrence on a soil moisture gradient ranging from xeric to mesic. We classified forest sites by their position on the gradient, which was quantified by an index representing the average moisture weight for all species present. We determined the relationship of the moisture index with upland oak site index on permanent plots dominated by even-aged stands of either mixed oaks (*Quercus* spp.) or yellow-poplar (*Liriodendron tulipifera*). Regression analysis indicated the moisture index was significantly ($P < 0.001$) associated with observed site index and explained 62 percent of its variation. Validation of the model with an independent dataset resulted in a mean absolute error in oak site index of 6.9 feet. Results of this exploratory study suggest that estimation of site index based on species composition has potential for application in mixed upland hardwood stands of the Southern Appalachians.

INTRODUCTION

The productive capacity of forest stands strongly influences their response to silvicultural treatments (Smith 1962). Site index is the method most used to evaluate site quality in eastern upland hardwood stands (Carmean 1970) and Beck and Trousdell (1973) provide a thorough description of the method, particularly its underlying assumptions and limitations. Estimation of site index in mixed hardwoods is often problematic, however, because suitable sample trees are often lacking, particularly on sites of intermediate or lower quality where oaks (*Quercus* spp.) and hickories (*Carya* spp.) predominate (Carmean 1970). Estimated site index may be biased in many stands, therefore, resulting in erroneous classifications of productivity. Replacement of conventional site index estimation based on sample trees with an alternate method is highly desirable for ecosystems where oaks are important. One such method is a procedure reported by Whittaker (1956) for arraying stands on environmental gradients based on composition of the tree stratum.

Forest productivity in the Southern Appalachian Mountains is associated primarily with temperature and moisture gradients (Whittaker 1966) and somewhat with fertility. Whittaker (1966) reported that ". . . an index of site moisture conditions based on weighted averages of stand composition . . ." was highly correlated with forest production. He subdivided the topographic-soil moisture gradient within broad elevation zones into four soil moisture classes (mesic, submesic, subxeric, and xeric) and assigned a weight to each class. Each tree species was assigned to a soil moisture class based on its modal frequency of occurrence along the gradient. Whittaker used the weighted average of each species present >1-inch d.b.h. as an index of the soil moisture conditions for a site. The index, which was a means for quantifying the relative position of sites on the moisture gradient, was highly correlated with primary forest production for vegetative communities occupying environments ranging from xeric to mesic in the Great Smoky Mountains National Park (Whittaker 1966). The simplicity of such a site classification system is appealing for a number of reasons: it can be readily applied with data typically collected from sample plots in a systematic inventory of stand conditions, it is easily adapted to other ecosystems with their associated species, it can be extended to other environmental gradients of temperature and nutrients, the system has an ecological basis because it is not based on commercial timber species as is site index, and its underlying basis is easily conveyed to other audiences.

This report describes results of a method of forest site classification based on arborescent species composition of upland hardwood stands in the Southern Appalachians. Previous work on the method (McNab and others 2002, 2003) utilized landscape scale datasets to explore possibilities of using species composition for site classification, but our current study is the first assessment using field data appropriate for model development and accuracy testing. This exploratory study investigated the question: Is a measure of species composition correlated with site index? Because methodology for the proposed method of site classification has not been evaluated, the purpose of this investigation was to obtain information on the type of vegetation data to collect. Such information included the strata of vegetation to inventory, e.g., saplings, trees, and the diagnostic value of rare species and ubiquitous species. Other important questions dealing with appropriate species weight values and if an index of moisture regime is more strongly related to growth response of silvicultural treatment or other ecological responses than to site index, will be addressed in future studies. Therefore, the intent of this exploratory study was the initial assessment of a new technique to determine if further evaluation and development is warranted and not to report a new method of site classification for immediate application.

METHODS

Study Area

The study was conducted in Bent Creek Experimental Forest—a 5,500-acre watershed located about 10 miles southwest of Asheville, NC. This area is characterized by

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short, mild winters and long, warm summers. Elevation ranges from 2,200 to 4,000 feet. Annual precipitation ranges from about 45 to 55 inches, depending on elevation and is evenly distributed throughout the year. Geologic formations consist of gneisses and schists of Precambrian age that have weathered to form a complex, dissected land surface. Soils, which consist mainly of Ultisols on gentle slopes and lesser areas of Inceptisols on steeper terrain, are generally deep and support over 50 hardwood species. The arborescent overstory canopy on dry slopes and ridges typically consists of communities dominated by scarlet oak (*Q. coccinea* Muenchh.), white oak (*Q. alba* L.), chestnut oak (*Q. prinus* L.), and black oak (*Q. velutina* Lam.). Typical mesophytic species occurring on moist slopes and in coves include yellow-poplar (*Liriodendron tulipifera* L.), northern red oak (*Q. rubra* L.), white ash (*Fraxinus americana* L.), eastern hemlock (*Tsuga canadensis* L.), cucumbertree (*Magnolia accuminata* L.), and black locust (*Robinia pseudoacacia* L.). Midstory species include flowering dogwood (*Cornus florida* L.), blackgum (*Nyssa sylvatica* Marsh.), sourwood [*Oxydendrum arboreum* (L.) DC.]; shortleaf pine (*Pinus echinata* L.) or pitch pine (*P. rigida* Mill.) may occur on heavily disturbed dry sites. Several species of hickory occur on dry and moist sites. Red maple (*Acer rubrum* L.) is usually present on sites of all moisture regimes. American chestnut [*Castanea dentata* (Marsh.) Borkh.] was a major component of most stands, particularly on dry slopes, before it was lost as a canopy species resulting from an introduced disease in the 1920s. Frothingham (1931) suggested that mountain forests below 4,000 feet elevation may be classified in two broad groups: moist slope and cove, and dry slope and ridge.

Most stands in the Bent Creek Experimental Forest have been affected by past settlement. Extensive areas of gentle slopes were cleared for subsistence farming from 1800 until about 1900, when land abandonment resulted in establishment of pine-hardwood mixtures on dry sites and yellow-poplar on moist sites. Timber stands on areas of steeper slopes, which were not cultivated or cleared for pasture, were typically burned, grazed, and periodically harvested by high-grading. Following acquisition of the watershed by the U.S. Forest Service around 1916, burning ceased and timber stand improvement work was done in selected areas to reduce stocking of undesirable species such as red maple, sourwood, and dogwood. Natural fires resulting from lightning is not common.

Field Plots

Field plots for our study had been established originally for two other investigations, one dealing with site index of upland oaks on dry sites and the other to study growth and yield of yellow-poplar on moist sites. In the site index study, Doolittle (1957) established 114 0.2-acre plots in even-aged stands of upland oaks where site index averaged 62 feet (range 36 to 87). All arborescent vegetation on the plots in the regeneration, sapling, and tree size strata was inventoried by species in 1970. The growth-and-yield study consisted of 34 0.25-acre plots established in even-aged stands dominated by yellow-poplar (Beck and Della-Bianca 1970) where site index averaged 99 feet (75 to 116). We pooled field data from the two studies to form a dataset extending over the mid- to upper range of site qualities occurring in the watershed. The group of plots established for the growth-and-yield study are dominated by mesophytic species and are considered representative of highly productive stands on mesic and submesic sites. The group of plots established for the study of site index was dominated primarily by oaks and other xerophytic species on xeric and subxeric sites. Site index relationships developed by Schnur (1937) for upland oak stands in the central hardwood region were used as the standard measure of productivity. We converted yellow-poplar site index to oak site index using the relationships presented by Doolittle (1958). Across all 148 plots, oak site index averaged 65 feet and ranged from 36 to 96 feet.

Vegetation Inventory

Arborescent vegetation occurring on each sample plot was inventoried to determine the species present by size classes and an assessment of their abundance. The number of species present was determined by inventory of arborescent vegetation in three size classes: regeneration (stems 0.1 to 4.5 feet tall), sapling (0.01 inch to 4.5 inches d.b.h.), and tree (>4.5 inches d.b.h.). The number of individuals of a species occurring in a size class, or abundance, was of particular importance in our study. Although Whittaker (1956) determined the abundance of each species by counting density, we used the less subjective method of presence or absence for several reasons. First, other ecological studies have shown presence is an easily quantified objective measure of a species occurrence that provides analysis capability equal or superior to density counts as a measure of abundance (Strahler 1977). More important, however, a conventional measure of density, e.g., stems per acre, was not used because we have observed abundance of some tree species tends to be associated with the intensity of recent disturbance rather than response of the species to effects of environmental factors (Beck and Hooper 1986, McGee and Hooper 1975, see table 1). We used a modified measure of density to determine if species rarity was an important source of variation. To account for the occurrence of a species resulting from a random, stochastic event unrelated to the environment, a maximum of two individuals of each species were recorded in each size class, which provided a measure of rareness. A species was classified as rare if it was represented by only one individual, e.g., $n = 1$, in the three combined vegetation size classes, or common if it occurred more than once, e.g., $n > 1$. For example, if only one sapling of Virginia pine (*P. virginiana* Mill.) was inventoried on a sample plot ($n = 1$), that species was classified as rare for that sample. If, however, both a seedling and a sapling were found on a sample plot ($n > 1$), then Virginia pine was classified as a common species. Evaluation of appropriate methods for inventory of tree species was an important part of our study.

Moisture Regime Classes and Index Values

Following the rationale of Whittaker (1956) we subdivided the observed upland soil moisture gradient in the Bent Creek watershed into four classes ranging from xeric to

Table 1—Moisture regime class, moisture weight assigned, and description of typical upland sites along a moisture gradient in Bent Creek Experimental Forest

Class	Weight	Description
Xeric	1	Usually ridge sites exposed to excessive moisture loss through wind and solar radiation; precipitation is primary source of soil moisture; moisture deficits may be common during parts of each growing season; typical arborescent species are xerophytes such as Virginia pine, blackjack oak, and post oak.
Subxeric	2	Upper to middle side slopes that receive a small amount of soil moisture from the downslope movement of subsurface water; moisture deficits may occur annually during the middle to late growing season; typical tree species are black oak, chestnut oak, and sourwood.
Submesic	3	Middle to lower sideslopes that receive a moderate amount of soil moisture from downslope movement of water; moisture deficits occur occasionally in the late-growing season during years of lower than average rainfall; typical species are black locust, dogwood, and northern red oak.
Mesic	4	Coves or northerly lower slopes protected from wind; important source of soil moisture is downslope movement from higher landforms; moisture deficits seldom occur except during exceptional drought; typical species are mesophytes such as white ash.

mesic (table 1). We subjectively assigned each tree species occurring in the watershed to a moisture regime class based on its perceived location of modal occurrence along the gradient. The binomial distribution represents the probability of occurrence for a species at various positions along the moisture gradient (fig. 1). Under the assumption that species are distributed individually in relation to their physiological characteristics, each species should have a unique binomial distribution.

Each of the four moisture classes, from xeric to mesic, was assigned a weight value ranging from 1 to 4, respectively. The weight can be viewed as either the relative availability of soil moisture for that position on the gradient during the frost-free season or the relative moisture requirements of a species. We refined the moisture weights by assigning half values to some species. White oak, for example, was assigned a weight of 2.5 because we have observed that its modal occurrence tends to occur between subxeric and submesic moisture classes.

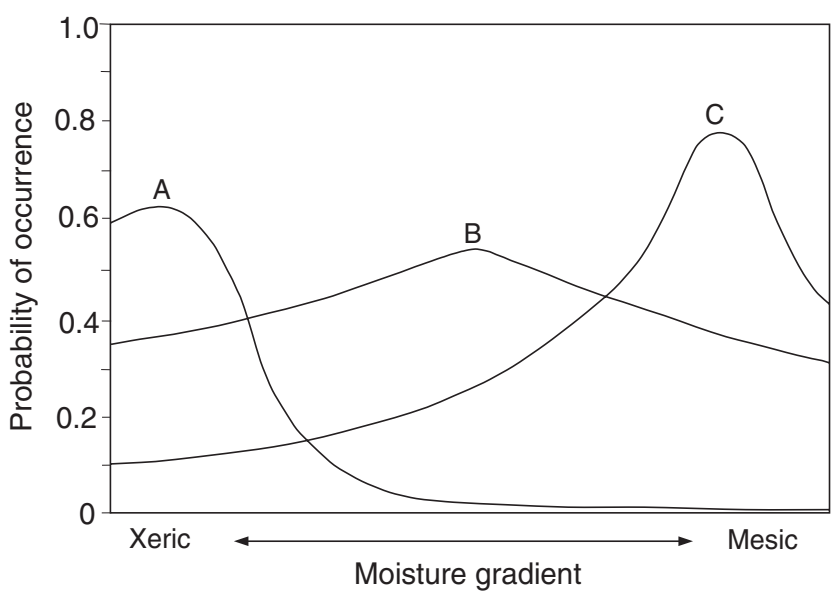

Figure 1—Examples of characteristic binomial probability distributions for tree species classified as (A) xerophytic, (C) mesophytic, or (B) ubiquitous.

Moisture Index

The cumulative information presented by all species occurring on a sample plot serves to locate its position on the moisture gradient. The position of each plot was quantified as the weighted average of the species in a value we termed moisture regime index (MRI):

$$MRI = \sum(species_i * MW_i + ... + species_j * MW_j)/N\ species \quad (1)$$

where

species = each tree species present on a sample site (value either zero or 1)

MW = moisture weight value for each species (values range 1 to 4)

N = number of species used in calculation of the index

MRI was calculated using 12 methods based on combinations of sample data from: three size strata for each species, e.g., tree, tree plus sapling, tree plus sapling plus regeneration; two levels of species abundance, e.g., rare ($n = 1$) and common ($n > 1$); and two levels of species ubiquity (intentional inclusion of a ubiquitous species or exclusion of the species). Red maple was selected as a trial ubiquitous species because our observations, and those of Whittaker (1956), suggest that it is a generalist species in this region and commonly occurs across all soil moisture classes, from xeric to mesic.

Data Analysis

We used a completely randomized design where sample plots had been established previously in stands meeting criteria for study of forest productivity. Correlation analysis was used to determine the association of site index with each of the 12 methods for calculation of the moisture index. A simple linear regression model was developed for site index as a function of the moisture index values for each field plot calculated by the method that produced the highest correlation. The statistical significance of the independent variable in the regression model was assessed at the probability level of $P = 0.05$.

Accuracy of the selected regression model for predicting site index from the moisture index was tested using holdout validation. The validation dataset was obtained by systematic exclusion of 10 percent ($n = 15$) of the total 148 plots from the analysis. The validation data consisted of 12 samples from the xerophytic sites and 3 from the mesophytic sites. The remaining data consisting of 133 observations formed the training data used for derivation of the relationship between site index and the moisture index.

RESULTS AND DISCUSSION

Thirty-one tree species were present on the 148 sample plots (table 2). Although dry-site and moist-site species occurred on plots of all moisture regimes, the frequencies of occurrence were consistent with the perceived soil moisture conditions, e.g., xerophytic oaks tended to dominate plots characterized by xeric and subxeric soil moisture regimes.

Little variation in strength of relationship occurred among the 12 correlations of site index and the MRI; coefficients of correlation ranged from 0.75 to 0.79 ($n = 133$, $P < 0.01$) (table 3). Correlations of site index and moisture index were highest ($r = 0.77$ to 0.79) for calculation of the index using the combined tree and sapling-size classes and lowest ($r = 0.75$) when the regeneration class of tree species was included. Including red maple as a ubiquitous species resulted in slightly higher correlations between site index and the moisture index ($r = 0.77$ to 0.79) compared to the exclusion of red maple ($r = 0.77$ to 0.78).

Regression analysis of site index as a function of the moisture index calculated by the method resulting in the highest correlation coefficient produced the relationship:

$$Oak\ SI\ (\text{ft}) = -18.3638 + 35.0359 * (MRI) \quad (2)$$

where

SI = site index (50 years) in feet for mixed species of oaks (Schnur 1937)

MRI = moisture regime index based on trees and saplings, red maple present, and species count $N > 1$.

As expected from the correlation analysis MRI was highly significant ($P < 0.0001$). This equation has a coefficient of determination of $r^2 = 0.62$ and mean square error of 8.11 feet. The pattern of residuals from the regression appeared to be uniformly distributed, suggesting a prediction equation with little bias (fig. 2).

We evaluated accuracy of the prediction equation using the validation data from 15 sample plots excluded from the analysis. Site index predicted by equation (2) was strongly correlated with observed values ($r = 0.90$) (fig. 3). The validation test produced site index estimates with a mean absolute error of 6.9 feet. Site index estimates for the validation plots, however, did not group around the diagonal line of perfect correlation, but formed a linear trend with deviations similar to the pattern displayed in figure 2. One explanation for the unusual pattern of estimated values is an artifact associated with the small size ($n = 15$) of the validation dataset. Examination of the validation data overlaid on the training data (fig. 2) indicates a pattern of higher than average site index for higher quality sites and lower than average site index for sites of lower quality. Another explanation for the unusual pattern is variation in site index associated with some samples that was not explained by the independent variable (MRI). Different validation results would likely have been obtained by drawing a second systematic sample, increasing sample size, or using another validation method, such as jackknifing or bootstrapping.

The species composition method we tested is similar to the indicator species approach that is widely used for ecological classification, where the presence of certain species indicates a site property such as moisture regime. The species composition approach, however, utilizes all

Table 2—Arborescent species present (>4.5 inch d.b.h.) on plots utilized in this study, weights assigned to each species relative to its modal position of occurrence on a local moisture gradient, and their frequencies of occurrence (percent of total plots sampled) on plots installed previously for two studies of xerophytic or mesophytic vegetation in Bent Creek Experimental Forest

Common name	Scientific name	Moisture weight	Species moisture class	
			Xerophytic[a]	Mesophytic[b]
			---------- *percent of plots* ----------	
Post oak	*Quercus stellata* Wangenh.	1.0	1	0
Virginia pine	*Pinus virginiana* Mill.	1.0	0	6
Scarlet oak	*Q. coccinea* Münchh.	1.5	75	0
Black oak	*Q. velutina* Lam.	2.0	60	32
Blackgum	*Nyssa sylvatica* Marsh.	2.0	32	21
Chestnut oak	*Q. prinus* L.	2.0	71	15
White pine	*P. strobus* L.	2.0	0	18
Persimmon	*Diospyros virginiana* L.	2.0	1	0
Sassafras	*Sassafras albidum* (Nutt.) Nees	2.0	2	15
Shortleaf pine	*P. echinata* Mill.	2.0	32	9
Sourwood	*Oxydendrum arboreum* (L.) DC.	2.0	86	56
Southern red oak	*Q. falcata* Michx.	2.0	2	0
Hickory spp.	*Carya* spp. Nutt.	2.3	0	62
American holly	*Ilex opaca* Aiton	2.5	0	3
White oak	*Q. alba* L.	2.5	80	32
Red maple	*Acer rubrum* L.	2.5	74	82
American beech	*Fagus grandifolia* Ehrh.	3.0	3	3
Black locust	*Robinia pseudoacacia* L.	3.0	37	71
Flowering dogwood	*Cornus florida* L.	3.0	42	47
Frazier magnolia	*Magnolia fraseri* Walter	3.0	0	6
White mulberry	*Morus alba* L.	3.0	0	3
Northern red oak	*Q. rubra* L.	3.0	31	35
Black cherry	*Prunus serotina* Ehrh.	3.5	0	3
Cucumber-tree	*Magnolia acuminata* (L.) L.	3.5	2	0
Eastern hemlock	*Tsuga canadensis* (L.) Carrière	3.5	3	21
American hornbeam	*Carpinus caroliniana* Walter	3.5	1	0
Sweet birch	*Betula lenta* L.	3.5	3	74
Yellow-poplar	*Liriodendron tulipifera* L.	3.5	39	100
Black walnut	*Juglans nigra* L.	4.0	0	3
White ash	*Fraxinus americana* L.	4.0	4	26
Yellow birch	*B. alleghaniensis* Britton	4.0	0	3

[a] One hundred fourteen 0.2-acre plots situated in stands dominated by xerophytic species, such as oaks.

[b] Thirty-four 0.25-acre plots situated in stands dominated by mesophytic species, such as yellow-poplar.

Table 3—Coefficients of correlation between oak site index and soil moisture regime calculated by all combinations of 3 size strata of trees present, 2 categories of red maple, and 2 categories of rare species on upland sites in Bent Creek Experimental Forest

Species ubiquity and rareness[a]	Size classes used in calculation of the moisture regime index		
	Trees	Trees + saplings	Trees + saplings + regeneration
	r		
With ubiquitous red maple			
With rare species	0.77	0.77	0.75
Without rare species	0.77	0.79	0.75
Without ubiquitous red maple			
With rare species	0.77	0.77	0.75
Without rare species	0.77	0.78	0.75

[a] Rareness class = rare: $n = 1$, common: $n > 1$.

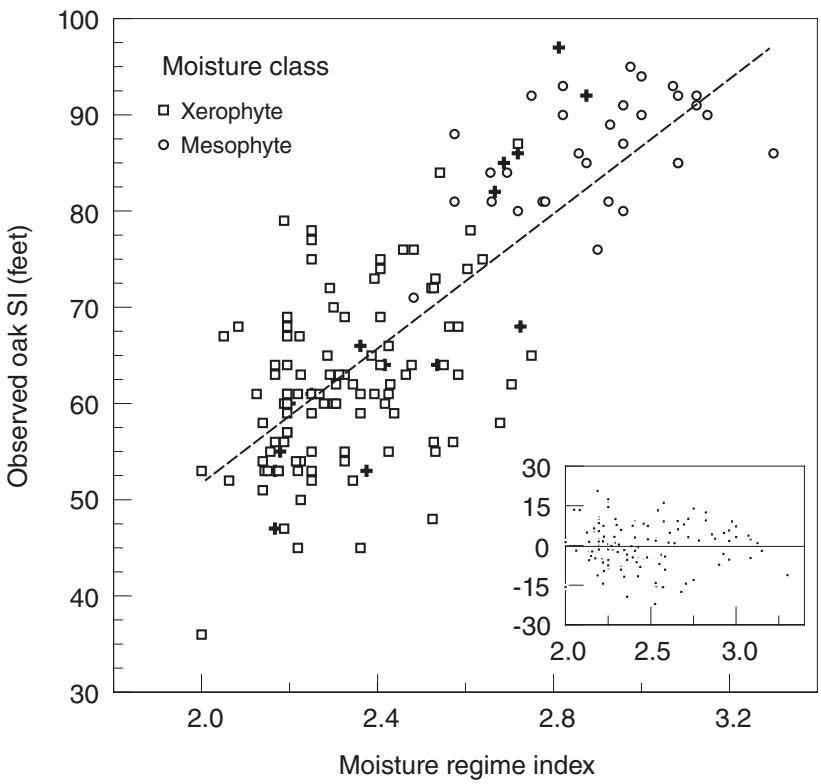

Figure 2—Relationship of observed oak site index with a moisture index on 133 sample plots dominated by species classified as either xerophytic or mesophytic. A simple linear trendline (dashed line) explained 62 percent of the variation in site index. Fifteen sample plots (+) were systematically excluded from the training dataset and reserved for validation. The inset shows residuals of predicted site index in relation to the moisture index.

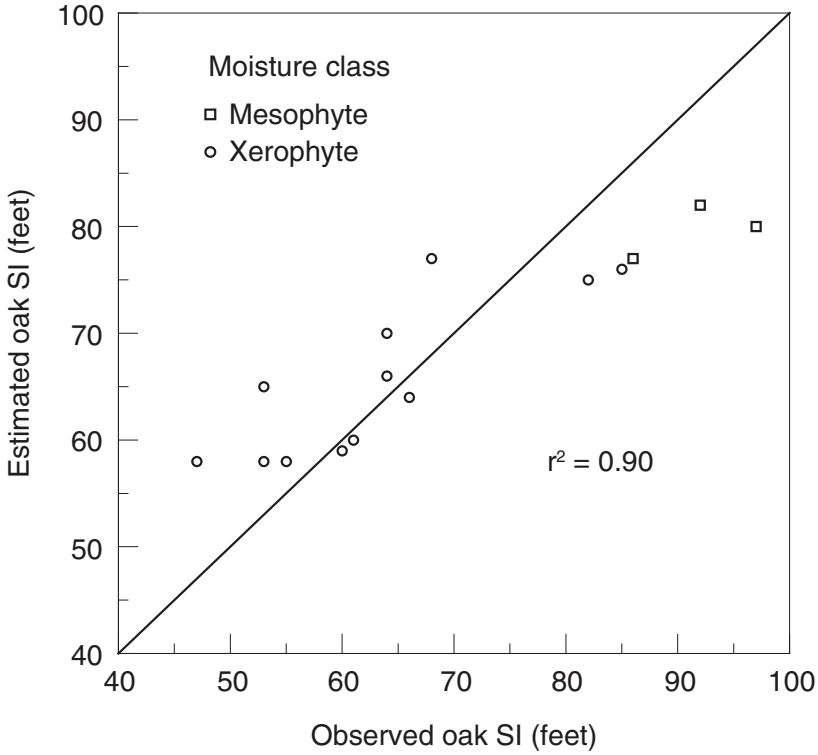

Figure 3—Correlation between oak site index observed on 15 validation plots and estimated from equation 2 was high (r^2 = 0.90). The diagonal line indicates perfect correlation between observed and estimated site index. The relationship was biased, however, with site index underestimated on high-quality sites and overestimated on low-quality sites. An explanation for the apparent bias of the prediction model is provided in the text.

arborescent species present to approximate the position of a site on the moisture gradient. In this species-rich region of the Southern Appalachian Mountains, therefore, an abundance of species associated with a moisture class is beneficial in placing a site in an appropriate position on the gradient and thus minimizes the influence of both ubiquitous and rare species. Although inclusion of shrub species was not part of this investigation, additional study is needed to determine the value of nonarborescent vegetation for estimation of site quality based on species composition.

Application of this method requires several considerations by the resource manager. First, a tree species list must be completed for the area of application. The list we presented in table 2 is limited to species encountered in our study area, which is about a third of those occurring in the Southern Appalachian region. Next, moisture weights assigned to each species should be adjusted for the region in which the method will be applied. The location of some species on the moisture gradient could change somewhat if other areas of application are near the limits of their natural range or compensating conditions are present, such as temperature or fertility. For example, Whittaker (1956) assigned yellow-poplar to his mesic class with a weight of 4. In our area of application, which is somewhat lower in elevation and drier than the region where Whittaker (1956) worked, we placed yellow-poplar between the mesic and submesic classes with a weight of 3.5. Finally, because site quality and species composition is variable in most stands, an adequate field inventory must be devised, such as a systematic grid of sample points or other design suitable for estimation of the timber resources.

In summary, results of this exploratory study demonstrate that arborescent species can be used for estimation of forest site quality in the Southern Appalachian Mountains. More specifically, we found that oak site index was highly correlated with an index based on species composition, which quantifies the location of a site on a xeric to mesic moisture gradient. An equation based only on the moisture index accounted for 62 percent of the variation in oak site index on sample plots in the study area. An unexplained bias in the validation test suggests, however, that additional study and refinement is needed before the method can be recommended for estimation of site index beyond the study area.

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LITERATURE CITED

Beck, D.E.; Della-Bianca, L. 1970. Yield of unthinned yellow-poplar. Res. Pap. SE-58. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 20 p.

Beck, D.E.; Hooper, R.M. 1986. Development of a Southern Appalachian hardwood stand after clearcutting. Southern Journal of Applied Forestry. 10: 168–172.

Beck, D.E.; Trousdell, K.B. 1973. Site index: accuracy of prediction. Res. Pap. SE-108. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 7 p.

Carmean, W.H. 1970. Site quality for eastern hardwoods. In: The silviculture of oaks and associated species. Res. Pap. NE-144. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station: 36–56.

Doolittle, W.T. 1957. Site index of scarlet and black oak in relation to Southern Appalachian soil and topography. Forest Science. 3: 114–124.

Doolittle, W.T. 1958. Site index comparisons for several forest species in the Southern Appalachians. Soil Science Society of America Proceedings. 22: 455–458.

Frothingham, E.H. 1931. Timber growing and logging practice in the Southern Appalachian region. Tech. Bull. 250. Washington, DC: U.S. Department of Agriculture Forest Service. 92 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. 3 p.

McNab, W.H.; Lloyd, F.T.; Loftis, D.L. 2002. Preliminary evaluation of methods for classifying forest site productivity based on species composition in western North Carolina. In: Doruska, P.F.; Bragg, D.C., eds. Proceedings of the 2001 southern mensurationists' conference. Monticello, AR: Arkansas Forest Resources Center: 10–18.

McNab, W.H.; Loftis, D.L.; Sheffield, R.M. 2003. Testing tree indicator species for classifying site productivity in Southern Appalachian forest types. In: Proceedings of 2002 national meeting of the Society of American Foresters. Bethesda, MD: Society of American Foresters: 350–356.

Schnur, G.L. 1937. Yield, stand, and volume tables for even-aged upland oak forest. Tech. Bull. 560. Washington, DC: U.S. Department of Agriculture. 88 p.

Smith, D.M. 1962. The practice of silviculture. New York: John Wiley & Sons, Inc. 578 p.

Strahler, A.H. 1977. Response of woody species to site factors in Maryland, USA: evaluation of sampling plans and continuous and binary measurement techniques. Vegetatio. 35: 1–19.

Whittaker, R.H. 1956. Vegetation of the Great Smoky Mountains. Ecological Monographs. 26: 2–80.

Whittaker, R.H. 1966. Forest dimensions and production in the Great Smoky Mountains. Ecology. 47: 103–121.

HYPOTHESES FOR COMMON PERSIMMON STAND DEVELOPMENT IN MIXED-SPECIES BOTTOMLAND HARDWOOD FORESTS

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Abstract—Common persimmon (*Diospyros virginiana* L.) is a shade-tolerant tree species found in southern bottomland hardwood forests. It is a desired species due primarily to its large fruit used by many wildlife species. While it has been observed as a component in natural reproduction, persimmon is rarely found as an overstory species in maturing bottomland hardwood stands. Unfortunately, little information exists regarding persimmon ecology and silviculture to develop silvicultural prescriptions to increase its stand density and development. Results from an archived dataset of stem analysis from a variety of bottomland hardwood species, personal observations of persimmon, and a conceptual model of tree species to plant with red oaks (*Quercus rubra* L.) in bottomland hardwood afforestation were used to develop hypotheses for future persimmon stand development research. These hypotheses are based on development in even-aged stands.

INTRODUCTION

Common persimmon (*Diospyros virginiana* L.) (hereafter referred to as persimmon) is a dioecious, shade-tolerant tree that occurs on a variety of sites throughout the Southeastern United States (Halls 1990, Skallerup 1953). Its best growth occurs on the rich, moist alluvial soils of river flood plains, where it can reach 70 to 80 feet tall and 20 to 25 inches d.b.h. (Halls 1990, Nix 2008). In the Lower Mississippi Alluvial Valley (LMAV), persimmon is most often found on clay or loamy flats (Putnam and Bull 1932).

Persimmon produces a true berry, also called a persimmon, that is highly desired by wildlife species (Perry and others 1999). Persimmon fruits are also edible for human consumption following ripening in the fall. They were a staple in the diets of Native Americans (Ohio Public Library Information Network 2001). Currently, the fruits are used in jellies, pudding, and pies (Anonymous 2008, Fletcher 1942). Persimmon also has a dense, hard, smooth wood suitable for golf club heads and shuttles for textile weaving (Das and others 2001, Maisenhelder 1971), but the loss of the golf club head market has reduced persimmon timber demand. Persimmon was also used for making flat-sliced veneer as face material in furniture (Maisenhelder 1971).

Interest in managing persimmon is increasing. It is commonly mentioned in forest management plans (Wilson and others 2007). In natural stand management, persimmon is considered a "hands off" species, or one usually left for wildlife habitat.[2] It is also a common, but minor, component of afforestation and reforestation efforts to meet wildlife habitat objectives (Aikman and Boyd 1941, Schweitzer and others 1999, Twedt 2004).

Persimmon, while sometimes establishing abundant natural reproduction, is rarely found as a component of the overstory canopy in a mature bottomland hardwood forest (Hepting 1935, Lentz 1929, Putnam and Bull 1932, Skallerup 1953). Early reports indicate that presettlement forests contained pure stands of persimmon, but this is no longer the case (see Skallerup 1953). A review of the literature reveals little information on persimmon ecology (especially stand development) and silviculture for developing silvicultural prescriptions to ensure development of this species to overstory prominence in bottomland hardwood forests. The objective of this study is to determine persimmon development patterns using an archived dataset that included stem analysis data from a variety of bottomland hardwood species. These results and personal observations will be used to develop hypotheses for future research in persimmon stand development.

METHODS

A hardwood growth-and-yield dataset developed between 1975 and 1977 is archived at the Southern Hardwoods Laboratory in Stoneville, MS. Dr. Bryce Schlaegel published a series of individual tree species volume and weight tables from this data (Schlaegel 1981, 1984a, 1984b, 1984c, 1984d; Schlaegel and Wilson 1983). Twenty-five stands were located in the LMAV and the adjacent Brown Loam Bluffs in westcentral Mississippi. A circular 0.2-acre plot was located in each stand. Four additional 0.2-acre plots were randomly located within a 5-acre circular area of the center of the first plot such that each plot fell within one of four quadrants of the first plot without overlapping any of the other plots. All trees >4.5 inches d.b.h. (diameter at breast height 4.5 feet above the ground) were tallied for species; d.b.h. (inches); crown class (dominant, codominant, intermediate, and suppressed); and distance and azimuth from plot center.

Trees for destructive sampling were selected after trees on all five plots in a given stand were measured. About 15 trees per stand were selected for sampling with no fewer than 13 trees

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per stand. Tables of limiting distances by tree diameters for different basal area factors (BAF) were used to select sample trees. An initial BAF 35 was used to determine the number of trees to be sampled from the five plots. If fewer than 13 trees were selected, then a BAF 30 was used and tree selection redone. A BAF 25 was used to gather the minimum number of trees if efforts with the BAF 30 were unsuccessful. Trees were selected as part of a bottomland hardwood growth-and-yield study, and not for stand development research objectives.

Each selected tree was mechanically felled and marked at 5-foot intervals to the top of the tree. One to one and one-half-inch thick discs were cut at each mark beginning at the base of the tree. Discs were labeled as to tree number and disc number and sealed in polyethylene bags. Discs for each tree were then placed in a burlap bag and labeled as to location and tree number, then taken to the laboratory for further analysis. Stem analysis followed standard techniques (Oliver 1978, 1982). Age was determined for each disc, then subtracted from the stump disc age to determine tree age at each 5-foot length interval. This data was plotted to determine individual tree length development and compared with the development of other trees sampled from the plot. Distances between the persimmon and the other selected trees were calculated using the law of cosines (Selby 1969):

$$a^2 = b^2 + c^2 - 2bc(\text{cosine } A) \qquad (1)$$

where

a = distance between two trees
b = distance from plot center to persimmon
c = distance from plot center to other selected tree
A = angle between the two trees calculated as the difference between the two azimuths

Crown radii were calculated using d.b.h. and equations developed by Francis (1986). A general species equation was used for willow oak (*Quercus phellos* L.), while equations by crown class were used for overcup oak (*Q. lyrata* Walter). The sweetgum (*Liquidambar styraciflua* L.) general equation was used for persimmon. Lockhart and others (2008) developed a conceptual model for trees that may act as trainers for bottomland red oaks (*Q. rubra* L.) during stand development. Sweetgum was used as the model species. Persimmon scored well as a potential trainer tree; therefore, it may have crown characteristics similar to sweetgum.

In reviewing the dataset, two plots were found that contained a destructively sampled persimmon. One plot was located on the Delta National Forest in Sharkey County, MS (32°57' N, 90°43' W). Soil is a Forestdale silty clay loam (fine, smectitic, thermic Typic Endoaqualfs). The stand was mature willow oak with 143 trees per acre (62 percent willow oak), 80 square feet of basal area per acre (69 percent willow oak), and an average stand diameter of 9.3 inches (9.8 inches for willow oak) (table 1).

Table 1—Tree species composition, number per acre, basal area per acre, and average d.b.h. for stands containing persimmon used in stem analysis

	Delta National Forest			Mahannah Plantation		
Species	*n*	Basal area	Average d.b.h.	*n*	Basal area	Average d.b.h.
	trees per acre	*square feet per acre*	*inches*	*trees per acre*	*square feet per acre*	*inches*
American elm	6.0 (1.0)	6.0 (1.1)	13.0 (4.2)	—	—	—
Bitter pecan	—	—	—	3.0 (0.7)	6.6 (0.1)	20.9 (5.4)
Cottonwood	—	—	—	2.0 (<0.1)	7.0 (0.8)	25.3 (2.8)
Green ash	15.0 (2.9)	4.8 (0.9)	7.5 (2.6)	39.0 (4.4)	17.8 (1.8)	8.5 (1.1)
Nuttall oak	5.0 (2.1)	1.4 (0.8)	6.2 (1.9)	1.0 (—)	0.5 (—)	9.1 (—)
Overcup oak	19.0 (1.9)	11.3 (0.8)	10.1 (2.3)	219.0 (16.4)	58.1 (4.5)	6.7 (0.1)
Persimmon	5.0 (0.6)	1.1 (0.1)	6.5 (1.3)	6.0 (0.6)	2.0 (0.3)	7.6 (0.4)
Sugarberry	3.0 (0.7)	0.5 (0.1)	5.4 (0.4)	3.0 (0.7)	2.3 (0.8)	13.8 (7.4)
Sweetgum	1.0 (—)	0.7 (—)	11.1 (—)	—	—	—
Willow oak	89.0 (5.6)	54.2 (3.9)	9.8 (2.2)	—	—	—
Stand[a]	143.0 (37.4)	80.0 (20.1)	9.3 (1.4)	273.0 (77.3)	94.7 (14.1)	7.4 (0.3)

Numbers in parentheses represent one standard deviation.

— = No trees for this species were present in the sampling for this stand.

[a] Stand values are based on plot averages and not the addition of individual species trees per acre, basal area per acre, or average d.b.h.

Four trees were utilized for stem analysis—one persimmon and three willow oaks (table 2).

The second plot was located on the Mahannah Plantation in Issaquena County, MS (32°32' N, 90°5' W), on what is now the Mahannah Wildlife Management Area. Soil in the stand is undifferentiated Sharkey clay (very-fine, smectitic, thermic Chromic Epiaquerts) and Dowling clay (very-fine, smectitic, nonacid, thermic Vertic Endoaquepts). The stand was largely composed of young overcup oak, probably resulting from an abandoned agriculture field or a complete harvest of the previous stand. The stand contained 273 trees per acre (80 percent overcup oak), 94.7 square feet of basal area per acre (61 percent overcup oak), and an average stand diameter of 7.4 inches (6.7 inches for overcup oak) (table 1). Six trees were utilized for stem analysis—one persimmon and five overcup oaks (table 3).

RESULTS

The plot age structure on the Delta National Forest contains multiple age classes (table 2). Willow oak 1 and willow oak 2 represent one age class, the persimmon represents a second age class, and willow oak 3 represents a third age class (fig. 1). The persimmon initiated 16 and 10 years after the first two willow oaks, respectively, and 14 years before the third willow oak. This persimmon was 73 percent smaller in d.b.h. than the two older willow oaks, and 56 percent smaller in d.b.h. than the younger willow oak. Further, the persimmon was 35 percent shorter in height than the willow oaks. All four trees showed steady length development, although the persimmon was slowing in growth during the 1960s and 1970s (fig. 1). Willow oak 4 displayed impressive growth throughout its life, averaging nearly 2 feet in length per year. Willow oak 1 was the closest to the persimmon at 14.9 feet, while the other two willow oaks were 24.8 feet away. Willow oak 1's crown was probably over the persimmon at the time of sampling, while the other two willow oaks were too far away (table 2).

Three trees showed a 5-foot difference between height (table 2) and length (fig. 1). Willow oak 3 had a difference of 10 feet. These differences are the result of comparing heights measured for standing trees using standard equipment, such as a clinometer (table 2), to adding the number of discs cut at 5-foot intervals during stem analysis. Tree length above the

Table 2—Tree characteristics on a destructively sampled stem analysis plot on the Delta National Forest, Sharkey County, MS, in 1977

Species	Age	D.b.h.	Height	Crown class	From plot center		Distance from persimmon	Crown radius
					Azimuth	Distance		
	years	*inches*	*feet*		*degrees*	*feet*		
Persimmon	53	5.6	50	Suppressed	60	10.4		6.5
Willow oak 1	69	22.6	80	Codominant	128	19.6	14.9	20.1
Willow oak 2	63	18.8	75	Codominant	77	34.6	24.8	17.0
Willow oak 3	39	12.8	75	Intermediate	215	15.0	24.8	12.0

Table 3—Tree characteristics on a destructively sampled stem analysis plot on the Mahannah Plantation, Issaquena County, MS, in 1977

Species	Age	D.b.h.	Height	Crown class	From plot center		Distance from persimmon	Crown radius
					Azimuth	Distance		
	years	*inches*	*feet*		*degrees*	*feet*		
Persimmon	42	10.6	66	Codominant	170	17.3		10.1
Overcup oak 1	44	6.4	58	Intermediate	36	35.6	49.2	7.1
Overcup oak 2	42	5.5	52	Intermediate	129	6.6	13.1	6.3
Overcup oak 3	44	10.8	66	Codominant	219	15.1	13.6	13.7
Overcup oak 4	44	7.5	60	Intermediate	276	10.0	17.4	8.0
Overcup oak 5	44	7.7	62	Codominant	340	7.7	24.9	11.6

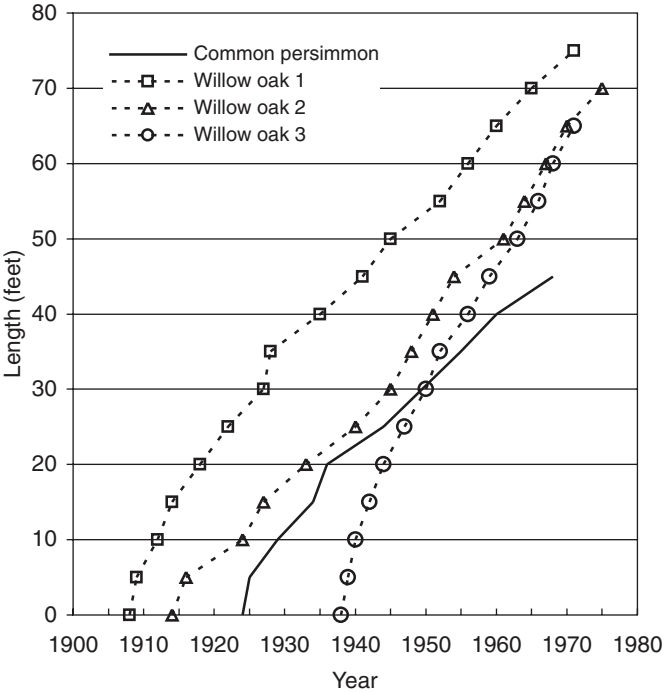

Figure 1—Persimmon and willow oak length development measured in a destructively sampled stem analysis plot on the Delta National Forest, Sharkey County, MS, in 1977.

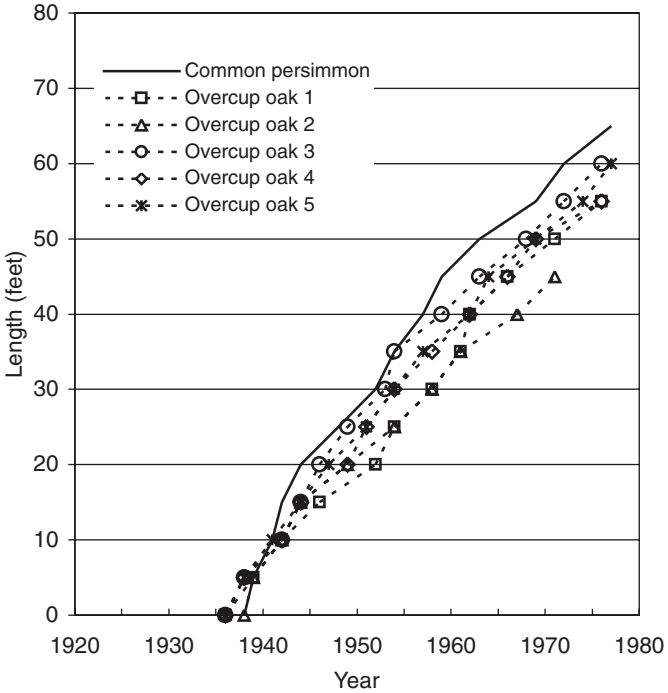

Figure 2—Persimmon and overcup oak length development measured in a destructively sampled stem analysis plot on the Mahannah Plantation, Issaquena County, MS, in 1977.

last disc, but <5 feet, is not used in stem analysis. Differences >5 feet are probably the result of a crooked main stem near the top of the tree or measurement error.

The plot age structure on the Mahannah Plantation differed by only 2 years among the six trees indicating an even-aged structure. The persimmon and overcup oak 3 were about 37 percent greater in d.b.h. than the other trees and slightly taller in height (table 3). All six trees were located in the main canopy based on their crown class. The persimmon, although 2 years younger than four of the five overcup oaks, has maintained canopy position throughout its life (fig. 2). Three overcup oaks were within 13 to 17 feet of the persimmon, while one overcup oak was nearly 50 feet away. The three closest overcup oaks were possibly in direct competition with the persimmon since their crown radii overlapped those of the persimmon (table 3). Tree height (table 3) and stem length (fig. 2) differences ranged from 1 to 7 feet.

DISCUSSION

The dataset used in this paper was developed for a bottomland hardwood growth-and-yield study. Trees selected for destructive sampling were not selected to test hypotheses of bottomland hardwood stand development. The nonpersimmon trees may or may not have been in competition with the persimmon. Further, trees that appear to be competing with persimmon at the time of sampling may not have been competing with persimmon earlier in stand development. Regardless, this dataset, along with a conceptual model of tree species to plant with red oaks in bottomland hardwood afforestation and personal observations of persimmon, does present interesting questions for further study involving persimmon stand development. Three hypothesis statements for further testing are listed below.

Hypothesis no. 1: Persimmon will not stratify above bottomland red oaks (section Erythrobalanus) during development in even-aged stands.

A conceptual model of species to plant in intimate mixtures with red oaks in bottomland afforestation indicates persimmon may be a useful species to “train” red oaks to develop better quality boles (Lockhart and others 2008). Concurrently, development of persimmon would probably be hindered in the presence of red oaks. The qualities of persimmon that are beneficial to red oaks include tree form, branching patterns, and relative twig diameter and durability. These characteristics would allow bottomland red oaks to stratify above persimmon and eventually suppress them. Therefore, persimmon may not be able to maintain overstory canopy position in competition with bottomland red oaks.

Hypothesis no. 2: Persimmon will maintain overstory canopy position with bottomland white oaks (*Q. alba* L.) (section Lepidobalanus) through early and midstages of development in even-aged stands.

Figure 2 shows persimmon was able to maintain canopy position in the presence of overcup oak in an even-aged stand. Overcup oak, as with the white oaks in general, are slower growing than bottomland red oaks. This slower growth, especially early height growth, may give persimmon a competitive advantage to stay slightly above the crowns of these species. In later stages of development, the white oaks will probably suppress persimmon through crown abrasion since these oaks have stouter twigs than persimmon.

Hypothesis no. 3: Persimmon will stratify above smaller diameter twig species, such as *Ulmus* spp. and sugarberry (*Celtis laevigata* Willd.), during development in even-aged stands.

The Sugarberry Natural Area (SNA) on the White River National Wildlife Refuge in eastcentral Arkansas has old-growth structure, with large trees and numerous canopy gaps (Lockhart and Kellum 2006). I have observed 5 to 10 persimmon trees per acre along low flats with strong intermediate or codominant canopy positions, a persimmon stand structure not often found in today's bottomland hardwood forests. Many of these trees had high-quality boles, and one tree had a measured 31-inch d.b.h. An obvious question is "How did these trees develop into the overstory canopy?" These trees were in the overstory canopy with American elm (*U. americana* L.) and sugarberry, species noted for small-diameter twigs. Persimmon may be able to maintain canopy position or even stratify above these species similar to stand development patterns found with cherrybark oak (*Q. pagoda* Raf.) and sweetgum (Clatterbuck and Hodges 1988, Lockhart and others 2006). Interspecific competition between these species would force persimmon trees to grow in height to maintain canopy position, resulting in high-quality boles. My observation though represented a snapshot of stand development. Individual trees and species that may have competed with persimmon in early stand development are now gone. Further, what is the age structure of the persimmon and competing species, or did these trees develop as an even-aged stand following a major disturbance?

In developing these hypotheses for future persimmon stand development research, I have focused on even-aged development in mixed-species stands. Additional persimmon stand development research questions involve development in pure, even-aged stands and development in uneven-aged stands.

Persimmon is often found as scattered individuals in natural bottomland hardwood forests. Is this the result of competitive pressure during stand development or past discriminate harvesting of persimmon when forest product markets for large, quality persimmon trees were good? Research is needed in the ecology of persimmon, especially stand development, to provide a basis for silvicultural decisions to promote the development of persimmon in future bottomland hardwood forests. Greater ecological knowledge is a prerequisite to successful management of this species.

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LITERATURE CITED

Anonymous. 2008. American persimmons (*Diospyros virginia*) [sic]. http://home.earthlink.net/~tabletophomestead/persimmons.html. [Date accessed: July 2].

Aikman, J.M.; Boyd, I.L. 1941. Adaptation studies of plants for soil conservation purposes in southern Iowa. Iowa Academy of Science Proceedings. 48: 117–121.

Clatterbuck, W.K.; Hodges, J.D. 1988. Development of cherrybark oak and sweetgum in mixed, even-aged bottomland stands in central Mississippi, USA. Canadian Journal of Forest Research. 18: 12–18.

Das, S.; Shillington, L.; Hammett, T. 2001. Persimmon. Virginia Tech Non-Timber For. Prod. Fact Sheet 13. Blacksburg, VA: Virginia Tech University. 5 p.

Fletcher, W.F. 1942. The native persimmon. Farmer's Bull. 685. Washington, DC: U.S. Department of Agriculture. 22 p.

Francis, J.K. 1986. The relationship of bole diameters and crown widths of seven bottomland hardwood species. Res. Note SO-328. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 3 p.

Halls, L.K. 1990. Common persimmon. In: Burns, R.M.; Honkala, B.H., tech. coords. In: Silvics of North America. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service: 294-298. Vol. 2.

Hepting, G. 1935. Decay following fire in young Mississippi Delta hardwoods. Tech. Bull. 494. Washington, D.C.: U.S. Department of Agriculture Forest Service. 32 p.

Lentz, G.H. 1929. Summary of first year's hardwood investigations in Louisiana. Journal of Forestry. 27: 486–494.

Lockhart, B.R.; Gardiner, E.; Leininger, T.; Stanturf, J. 2008. A stand development approach to oak afforestation in the Lower Mississippi Alluvial Valley. Southern Journal of Applied Forestry. 32: 120–129.

Lockhart, B.R.; Hodges, J.D.; Ezell, A.W.; Clatterbuck, W.K. 2006. Using natural stand development patterns in artificial mixtures: a case study with cherrybark oak and sweetgum in east-central Mississippi, USA. Forest Ecology and Management. 222: 202–210.

Lockhart, B.R.; Kellum, J.E. 2006. A complex stand on the White River National Wildlife Refuge: implications for bottomland hardwood old growth. Journal of the Arkansas Academy of Science. 60: 181–186.

Maisenhelder, L.C. 1971. Common persimmon (*Diospyros virginiana*). American Woods FS-25. Washington, DC: U.S. Department of Agriculture Forest Service. 6 p.

Nix, S. 2008. How to ID and manage persimmon. http://forestry.about.com/od/hardwoods/p/persimmon.htm?p=1. [Date accessed: July 2].

Ohio Public Library Information Network. 2001. Persimmon. http://new.oplin.org/tree/fact%20pages/persimmon/persimmon.html. [Date accessed: July 2, 2008].

Oliver, C.D. 1978. Development of northern red oak in mixed-species stands in central New England. Yale School of For. and Environ. Studies Bull. 91. 63 p.

Oliver, C.D. 1982. Stand development - its uses and methods of study. In: Means, J.E., ed. In: Forest succession and stand development research in the Northwest. Corvallis, OR: Oregon State University, Forest Research Laboratory: 100–112.

Perry, R.W.; Thill, R.E.; Peitz, D.G.; Tappe, P.A. 1999. Effects of different silvicultural systems on initial soft mast production. Wildlife Society Bulletin. 27: 915–923.

Putnam, J.A.; Bull, H. 1932. The trees of the bottomlands of the Mississippi River Delta region: their distribution, occurrence by site and cover type, utilization in the Delta, and the distinguishing characteristics of the principal and most common species and groups of species. Occas. Pap. 27. New Orleans: U.S. Department of Agriculture Forest Service. 207 p.

Schlaegel, B.E. 1981. Willow oak volume and weight tables. Res. Pap. SO-173. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E. 1984a. Green ash volume and weight tables. Res. Pap. SO-206. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E. 1984b. Overcup oak volume and weight tables. Res. Pap. SO-207. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E. 1984c. Sugarberry volume and weight tables. Res. Pap. SO-205. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E. 1984d. Sweetgum volume and weight tables. Res. Pap. SO-204. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B.E.; Willson, R.B. 1983. Nuttall oak volume and weight tables. Res. Pap. SO-186. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 14 p.

Schweitzer, C.J.; Gardiner, E.S.; Stanturf, J.A.; Ezell, A.W. 1999. Methods to improve establishment and growth of bottomland hardwood artificial regeneration. In: Stringer, J.W.; Loftis, D.L., eds. In: Proceedings 12th central hardwood forest conference. Gen. Tech. Rep. SRS-24. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 209–214.

Selby, S.M., ed. 1969. Standard mathematical tables. 17th ed. Cleveland, OH: The Chemical Rubber Co. 724 p.

Skallerup, H.R. 1953. The distribution of *Diospyros virginiana* L. Annals of the Missouri Botanical Garden. 40: 211–225.

Twedt, D.J. 2004. Stand development on reforested bottomlands in the Mississippi Alluvial Valley. Plant Ecology. 172: 251–263.

Wilson, R.; Ribbeck, K.; King, S.; Twedt, D., eds. 2007. Restoration, management, and monitoring of forest resources in the Mississippi Alluvial Valley: recommendations for enhancing wildlife habitat. Vicksburg, MS: Lower Mississippi Valley Joint Venture Forest Resource Conservation Working Group. 88 p.

LONGLEAF PINE

Longleaf pine-wiregrass community on the Apalachicola National Forest south of Tallahassee, Florida. (Photo by James M. Guldin)

IMPACT OF FIRE IN TWO OLD-GROWTH MONTANE LONGLEAF PINE STANDS

John S. Kush, John C. Gilbert, Crystal Lupo, Na Zhou, and Becky Barlow[1]

Abstract—The structure of longleaf pine (*Pinus palustris* Mill.) forests of the Southeastern United States Coastal Plains has been the focus of numerous studies. By comparison, the forests in the mountains of Alabama and Georgia are not well understood. Less than 1 percent of longleaf pine stands found in the montane portion of longleaf's range are considered old growth. Several of these stands occur on the Mountain Longleaf National Wildlife Refuge located in northeastern Alabama. A 1998 study documented the conditions in two old-growth longleaf pine stands on the refuge. The 1998 study described the age and stand structure, and shed light on the past disturbance and replacement patterns of two remnant old-growth longleaf pine stands. In 2005 and 2008, these stands were remeasured to document changes. In 2004, one stand was subjected to a relatively intense prescribed fire. In 2006, the other stand was burned with a more conservative approach.

INTRODUCTION

Mountain longleaf pine (*Pinus palustris* Mill.) forests are a critically endangered component of the once vast longleaf pine forests of the Southeast. Stretching from coastal Virginia to the pineywoods of east Texas, the longleaf pine forest, maintained by frequent fire, has dwindled in acreage and integrity. Several small pockets of this once vast forest remain in the Coastal Plain, but in the mountain region only a small national wildlife refuge in northeastern Alabama contains a forest that approaches the landscape witnessed by European settlers—Mountain Longleaf National Wildlife Refuge (MLNWR).

It is well known and accepted that Coastal Plain longleaf pine forests were fire maintained, but what about the montane longleaf pine forests? Based on observations, it appears that longleaf pine is currently confined to ridgetops and slopes with south/southwesterly aspects. Elsewhere, the tree is found in mixed pine and pine-hardwood mixtures. Historical accounts provided by Sargent (1884), Mohr (1897), Reed (1905), Andrews (1917), and Harper (1905, 1913, 1928, 1943) present a different picture. Each of these has accounts which indicated longleaf pine was occurring in pure stands up to 2,000 feet in elevation and on all aspects of many of the mountains in northeastern Alabama and northwestern Georgia.

Today few stands of longleaf like those described by the above authors remain. Several years of extensive field and laboratory work in the late 1990s on what was once Fort McClellan Army Base found 12 old-growth tracts, lush herbaceous communities, and several management concerns. With the closure of the fort in 1998, the new MLNWR was established on the eastern half of the base in 2003, which now contains nine of these stands. Located just northeast of Anniston, AL, and lying in the growing Birmingham, AL, to Atlanta, GA, corridor, the refuge is 9,016 acres in size. This is the first mountain national wildlife refuge in the Southeast and contains the third highest mountain ridge in Alabama.

Today it is known that MLNWR contains the finest extant of mountain longleaf pine. The refuge objective is to preserve and enhance the natural mountain longleaf pine ecosystem and preserve a natural diversity and abundance of native fauna and flora with an emphasis towards red-cockaded woodpeckers (*Picoides borealis*). Two of the most intact longleaf pine ecosystems and best examples of fire-maintained old-growth stands are found on Caffey Hill and Red-tail Ridge areas of the refuge.

Why is there so much longleaf pine left on the MLNWR and why are Caffey Hill and Red-tail Ridge in such good shape? The fire regime! When the MLNWR was still Fort McClellan, wildfires were allowed to burn on the fort until the early 1960s, then fire exclusion occurred over most of the fort. However, Red-tail Ridge was allowed to burn every year from 1987 until Fort McClellan closed in 1998, often twice per year and Caffey Hill burned less frequently and often at night.

Caffey Hill and Red-tail Ridge were mapped and measured in 1998. The stands were resurveyed in 2005 and again in 2008. This paper will report on the changes in each stand and their responses to being prescribed burned.

METHODS

All longleaf pine trees >0.6 inches diameter at breast height (d.b.h.) were stem mapped and measured for d.b.h. Crown and total height were measured on each tree. All trees over 4 inches d.b.h. were cored to determine ring count at 4 feet. Longleaf pine regeneration and nonlongleaf pine species were subsampled using milacre plots randomly located in each stand.

Caffey Hill

This stand is 3.7 acres in size and has a south/southeasterly aspect, with slopes running from 40 to 60 percent. Located on the upper slope of a ridge with elevations from 1,350 to 1,500 feet, it was burned in May 2004 where the fire was ignited at the bottom of the slope and allowed to burn upslope.

Red-tail Ridge

It is 4.4 acres in size. It has a west/southwesterly aspect, with slopes running from 30 to 45 percent. The stand is found upper to midslope on a ridge with elevations from 1,100 to 1,250 feet. It was burned in March 2006 with aerial ignition by helicopter.

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RESULTS

Caffey Hill

The density and basal area dropped between the 1998 and 2005 measurements. Density went from 121 to 95 trees per acre and basal area fell from 34.9 square feet per acre to 32.7 square feet per acre. There was only the loss of 1 foot per acre between 2005 and 2008 with basal area increasing to 36.1 square feet per acre. The loss in density between 1998 and 2005 occurred in the lower d.b.h. classes (fig. 1). During this time over 40 percent of the trees in the 1- to 3-inch d.b.h. class were lost. Most of these were killed by the fire, which often burned up entire trees so that only metal tags remained. Over one-fifth of trees >9.0 inches d.b.h. perished in the fire. This is often the size at which trees begin to have fertile cones. Due to fire intensity, there were also instances in this area where trees >20 inches d.b.h. were killed by the fire.

Red-tail Ridge

The story was a little different for Red-tail Ridge. Despite the fact that this stand was not burned between 1998 and 2005, there was a loss in density from 110 trees per acre to 104 trees per acre (fig. 2). However, basal area increased from 55.4 to 58.6 square feet per acre. With a similar loss in density, from 104 trees per acre to 96 trees per acre in 2008, there was a loss in basal area which dropped to 56.8 square feet per acre. The prescribed fire in March 2006 had a more pronounced effect on stand density, removing trees in the upper diameter classes as well as many of the smaller diameter trees.

DISCUSSION

Since 1994, field reconnaissance on Fort McClellan, now MLNWR, by Auburn University's School of Forestry & Wildlife Sciences identified a number of old-growth longleaf pine stands. Many of these stands have undergone various lengths of fire suppression and degradation. MLNWR's longleaf pine forests provide the "missing link" to scientists, land managers, and conservationists in the mountain region, providing the only information on (1) age and stand structure and dynamics of frequently burned old-growth forests, (2) composition of pristine plant communities, and (3) landscape extent of mountain longleaf pine forests.

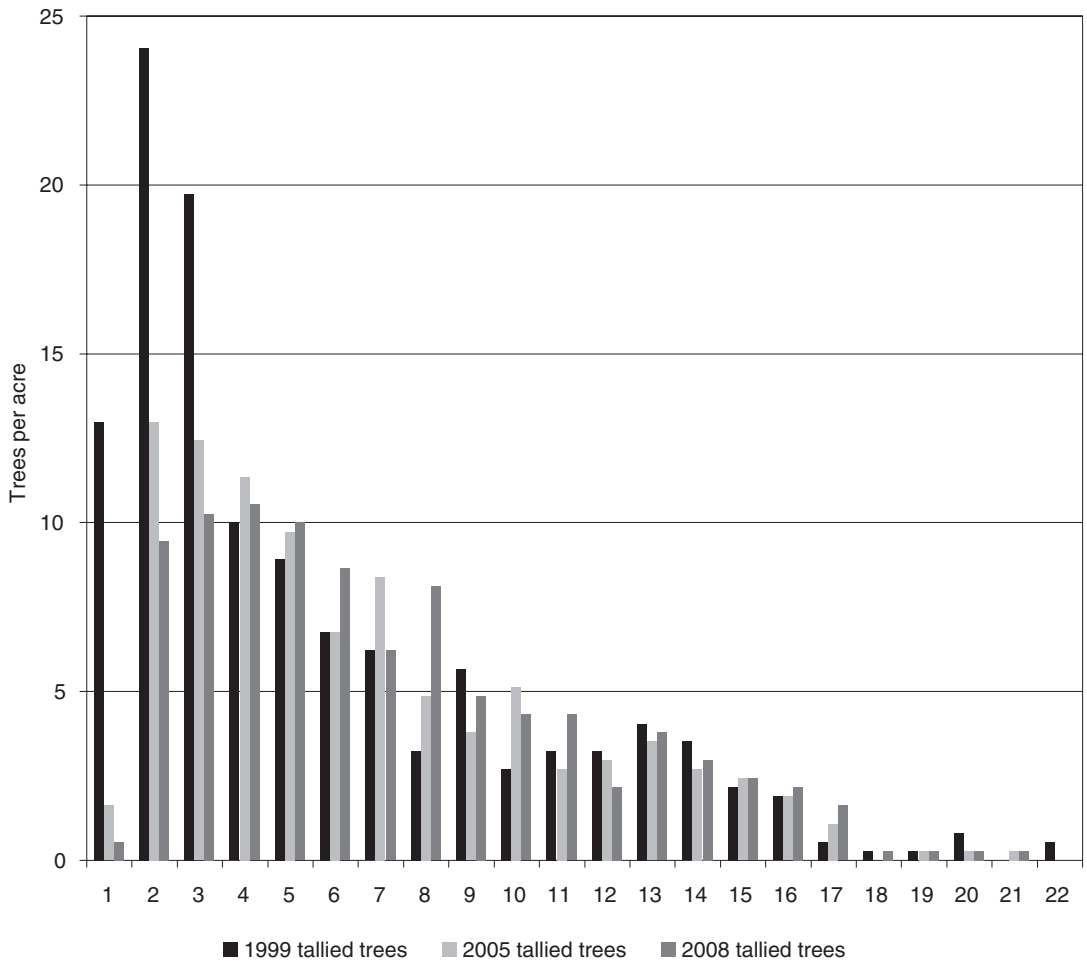

Figure 1—Longleaf pine diameter distribution (trees per acre) for the Caffey Hill location at the Mountain Longleaf National Wildlife Refuge near Anniston, AL.

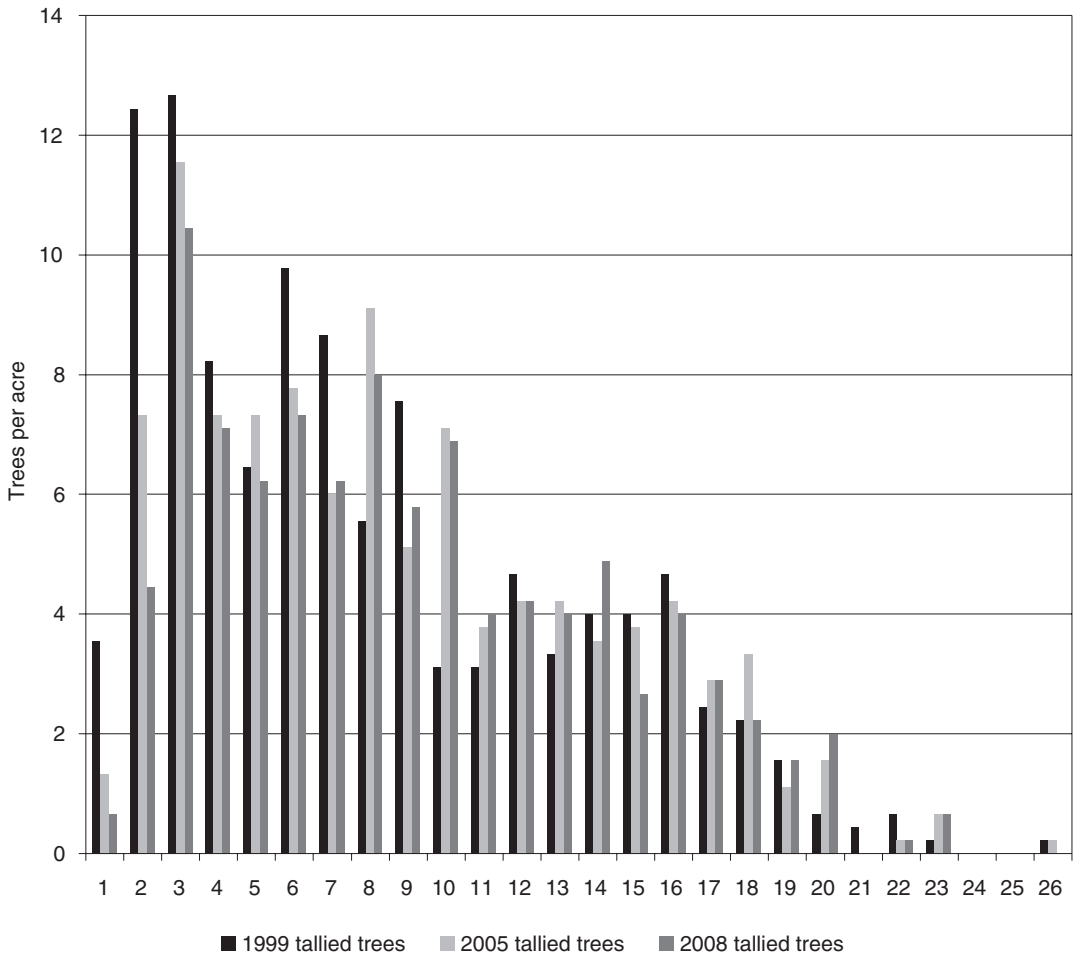

Figure 2—Longleaf pine diameter distribution (trees per acre) for the Red-tail Ridge location at the Mountain Longleaf National Wildlife Refuge near Anniston, AL.

In the past, Fort McClellan's forest management program was based on wildfire prevention, endangered species protection, timber harvesting, and forest regeneration. Contact with the public, other than in conjunction with the robust hunter program, was minimal. Prescribed fire and forest management restrictions were based on Army and U.S. Department of Defense policies. Yesterday's Fort McClellan is gone, and the challenges that face this landscape are increasingly complex. First, MLNWR's mountain longleaf pine forests will face new management objectives. Restoration and management of MLNWR should become the highest priority. Natural area management will likely take the place of the former timber/fire management program of what was once Fort McClellan.

Next, and most importantly, conditions acceptable for performing prescribed fires will drastically change. As a U.S. Army post, Fort McClellan's prescribed fire "windows" were more liberal—minimal smoke concerns, minimal public involvement. With changing ownership, both smoke and the public's perception of forest fire will reduce the window available for fire management of the MLNWR landscape. At the same time, forests on the refuge are now viewed by the public as a preferred neighbor for housing development. This has resulted in an increasing challenge involving wildland fire-urban interface issues.

Smoke sensitivity will increase not only from changing ownership, but also from in-holdings and potential highway access. A bypass route adjacent to MLNWR will limit the smoke window further, making times and conditions suitable for burning extremely limited. Development along this corridor would only decrease the fire window. Proposed industrial development in the northwest portion of what was Fort McClellan may likewise limit burning anywhere along Choccolocco Mountain.

Finally, as Anniston and the Birmingham–Atlanta corridor continue their expansion, problems mentioned above (smoke and public perception of fire) will only intensify without an

aggressive extension and education effort. As an Army post, Fort McClellan was considered the "Military Showplace of the South." With closure and military evacuation in 1999, natural resource managers have the opportunity to maintain and recreate a piece of southern biological history—pristine mountain longleaf pine forests.

Several publications describing the site, the restoration goals and process, and early findings have been published (Maceina 1997; Maceina and others 1997, 2000; Varner 2000; Varner and others 2000, 2003a, 2003b).

CONCLUSION

The MLNWR has some of the best fire-maintained old-growth montane longleaf pine stands left. However, fire intervals have been increasing since the refuge was created from what was part of Fort McClellan. This increase in fire interval has potentially led to problems when fire is applied to the stands. It appears fire is the underlying cause for mortality of longleaf pine in the steeper areas of the MLNWR. The refuge managers need to be careful not to lose any/many more overstory trees. There is a need to monitor cone crops to catch seed and then monitor regeneration, if any. When longleaf regeneration gets established then there needs to be a more cautious approach to applying fire. The MLNWR without longleaf pine cannot be called the Mountain Longleaf National Wildlife Refuge!

ACKNOWLEDGMENTS

The authors wish to thank Mr. Bill Garland for his years of service to Fort McClellan and then the Mountain Longleaf National Wildlife Refuge.

LITERATURE CITED

Andrews, E.F. 1917. Agency of fire in propagation of longleaf pines. Botanical Gazette 64: 497–508.

Harper, R.M. 1905. Some noteworthy stations for Pinus palustris. Torreya 5: 55–60.

Harper, R.M. 1913. Economic botany of Alabama. Part I—geographical report on forests. Monograph 8. University, Alabama: Geological Survey of Alabama. 228 p.

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. Monograph 9. University, Alabama: 357 p.

Harper, R.M. 1943. Forests of Alabama—revision of Monograph 8, Geographical Report, including descriptions of the natural divisions of the state, their forests and forest industries. Monograph 10. University, Alabama: Geological Survey of Alabama. 230 p.

Maceina, E.C. 1997. Characterization of a montane longleaf pine community type on Fort McClellan, Alabama: community structure within pine-hardwood forest type. Auburn, AL: Auburn University. 180 p. M.S. thesis.

Maceina, E.C.; Kush, J.S.; Meldahl, R.S. 1997. Longleaf pine restoration plan for Fort McClellan, Alabama: a final report. On file with: Mountain Longleaf National Wildlife Refuge, Fort McClellan, AL, and the Longleaf Pine Stand Dynamics Laboratory, Auburn University, AL.

Maceina, E.C.; Kush, J.S.; Meldahl, R.S. 2000. A floristic survey of a Blue Ridge montane longleaf pine (*Pinus palustris* Mill.) community at Fort McClellan, Alabama. Castanea. 65(2): 147–154.

Mohr, C. 1897. The timber pines of the United States. Bulletin #13. Washington, D.C.: U.S. Department of Agriculture, Division of Forestry. 172 p.

Reed, F.W. 1905. A working plan for forest lands in central Alabama. Bulletin No. 68. Washington, D.C. : U.S. Department of Agriculture, Forest Service. 71 p.

Sargent, Charles S. 1994. Report on the Forests of North America (exclusive of Mexico). Washington, D.C.: U.S. Department of the Interior, Census Office. Government Printing Office. 612 p.

Varner, J.M.; Kush, J.S.; Meldahl, R.S. 2003a. Structure of old-growth longleaf pine (*Pinus palustris* Mill.) forests in the mountains of Alabama. Castanea. 68(3): 211–221.

Varner, J.M.; Kush, J.S.; Meldahl, R.S. 2003b. Vegetation of frequently burned old-growth longleaf pine (*Pinus palustris* Mill.) savannas on Choccolocco Mountain, Alabama, USA. Natural Areas Journal. 23: 43–52.

Varner, J.M., III. 2000. Species composition, structure, and dynamics of old-growth mountain longleaf pine forests of Fort McClellan, Alabama. 128 p. Auburn, AL: Auburn University. M.S. thesis.

Varner, J.M., III; Kush, J.S.; Meldahl, R.S. 2000. The mountain longleaf pine resources of Fort McClellan, Alabama: final report on their status, ecology, and management needs. On file with: Mountain Longleaf National Wildlife Refuge, Fort McClellan, AL, and the Longleaf Pine Stand Dynamics Laboratory, Auburn University, AL.

A DECISION TREE APPROACH USING SILVICS TO GUIDE PLANNING FOR FOREST RESTORATION

Sharon M. Hermann, John S. Kush, and John C. Gilbert[1]

Abstract—We created a decision tree based on silvics of longleaf pine (*Pinus palustris*) and historical descriptions to develop approaches for restoration management at Horseshoe Bend National Military Park located in central Alabama. A National Park Service goal is to promote structure and composition of a forest that likely surrounded the 1814 battlefield. We considered options for (1) stands currently supporting second-growth montane longleaf, (2) isolated trees, and (3) areas beyond dispersal range of longleaf seed. We estimate >50 percent of the property is appropriate for longleaf but <20 percent is within dispersal range of existing seed trees. There are three major areas with current densities sufficient to be classified as longleaf stands—in these high-priority stands, fuel reduction burns are being applied to maintain adult trees and create appropriate seedbeds. Criteria applied to isolated trees categorize them as medium priority. Areas beyond seed dispersal range of longleaf are low priority and require planting.

INTRODUCTION

Forest restoration efforts are largely focused on reestablishing trees. A current issue of concern in the Southeastern United States is how best to increase acreage of longleaf pine (*Pinus palustris*). With at least a 97-percent loss in this once-dominant ecosystem, there is interest in restoring both the species and the forest type. In some cases a primary interest is to regenerate existing stands (Croker and Boyer 1975) and in other cases a forester must decide which tree species is most appropriate to use to initiate a new stand (Moser and others 2003). Although there have been advances in restoration much of the effort has been limited to planting trees, especially in the Coastal Plain. Planting is critical to the ultimate goal of recovering the forest. However, there are additional approaches that could be pursued (where appropriate). There is growing interest in longleaf forest restoration that promotes ecological values that include forest composition and structure. In the current paper we review a traditional approach for planning forest restoration projects and we propose to augment previous efforts. The additions are designed, in part, to maximize benefits conferred by residual longleaf trees that are not dense enough to be of value in traditional stand regeneration efforts. In addition we propose factors to aid in prioritizing areas within the landscape for (1) relying on natural regeneration, (2) enhancing the value of residual longleaf, and (3) artificial regeneration (seedling planting). To make this approach useful, we apply information based on ecology of the species and incorporate knowledge derived from longleaf pine silvics.

OVERVIEW OF STEPS IN TRADITIONAL FOREST RESTORATION

Steps in forest restoration often include:

1. Articulation of goal or desired future condition for the site. When there is an emphasis on conservation values this often results in the need to understand what tree species dominated the landscape during some reference period (often representing pre-European settlement)
2. Determination of the existing conditions (restoration starting point)
3. Steps 1 and 2 set in motion application of a decision tree (ex. Johnson 1998) that includes such actions as
   a. removal of offsite trees
   b. site preparation (control of competing species, etc.) and, if necessary, remediation treatments such as subsoiling, etc.
   c. seedling selection
   d. planting technique
   e. release actions

Under a traditional, formulaic approach to forest restoration planting is the primary focus because it meets the primary goal of establishing a target tree species. When the primary goal is driven by conservation concerns and when there are residual components of the forest that are retained on the site then a broader range of actions may be warranted. In addition, when longleaf pine is the target of forest restoration there may need to be additional actions such as (1) reintroduction of fire and/or (2) reintroduction of ground-layer species.

Many recommendations included in the steps for forest restoration are most appropriate for homogeneous management units and/or where restoration goal is uniform across the landscape. Because the process is designed to produce an even-sized stand, any residual longleaf trees are eliminated before planting begins. The Longleaf Alliance developed an informal decision tree to guide successful creation of a longleaf pine plantation but also has options for making effective use of longleaf currently existing under undesirable conditions (Johnson 1998). If goals for a site include ecological values, then the additions to the process of forest restoration proposed by the Longleaf Alliance will promote those goals better than more traditional restoration options. In the current paper we propose additional

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options that should enhance ecological goals even more. Specifically, we focus on enhancing the ecological value of residual longleaf pine trees rather than discarding them in favor of a more uniform landscape. In addition we propose a set of considerations to aid in prioritizing areas for different management actions including enhancing natural regeneration opportunities vs. reliance on planting.

EXPANDING TRADITIONAL APPROACHES TO LONGLEAF FOREST RESTORATION

There is no doubt that there is a vital role for tree planting in an effort to recover some of the 97 percent of the original longleaf forest lost to conversion and development. However, there are additional opportunities to maximize the potential of sites that support residual components of the forest. Sites that retain some longleaf trees, especially those of cone-bearing size may be relatively small in acreage but the small area is offset by having trees that are decades older (often 40 to 70 years older) than newly planted stands. One obvious expansion of traditional forest restoration is to promote efforts to recover fire excluded stands. Using Forest Inventory and Analysis plot data, Outcalt (2000) determined that approximately 50 percent of the existing stands of longleaf pine are inadequately burned as measured by presence of fire in the past 5 years. Reintroduction of fire requires effort but also may yield high benefits. Planted stands with young, even-aged trees will require decades to reach the ecological complexity offered by multiple-aged stands.

The decision tree offered by the Longleaf Alliance (Johnson 1998) appropriately stresses careful assessment of the starting point. This is the basis for selection of appropriate types of site prep techniques, planting methods, release actions, and subsequent management. Focus is on stands or management units where the restoration goal is uniform across the landscape and this is often highly appropriate for a small private landowner.

We propose ways to broaden the decision tree of the Longleaf Alliance. As interest in complex conservation values of longleaf restoration grows, the range of desired future conditions expands. Start points become more multifaceted and less homogeneous. This is especially true north of the fall line where topography is more complex and the natural landscape may not be as homogeneous as the Coastal Plains. Broader consideration of starting conditions provides additional opportunities for conservation efforts, especially for recovery of degraded sites. In the current paper we describe the initial phases of a case study of expanded planning for forest restoration at the landscape level.

THE SITE

The target area is Horseshoe Bend National Military Park (HOBE), Tallapoosa County in northcentral Alabama. This 2,000+-acre site is north of the fall line, in the montane region of longleaf. The National Park Service (NPS) acquired HOBE in the 1950s because it is the site of the final battle of the Creek War of 1813 to 1814. Across the region the majority of uplands were cutover prior to 1930s and HOBE appears to be no exception. The NPS is interested in restoring uplands surrounding the battlefield to a condition that represents the time of the battle. A battlefield letter provides some evidence for an open-canopy forest in 1814 (see Hermann and Kush 2006). Almost a century after the battle, there is documentation from a nearby county that confirms open-canopy forest structure as well as describes the composition of native longleaf forest (Reed 1905).

Hermann and Kush (2006, in press) provide an overview of HOBE uplands today that supports residual longleaf in a long (50+ years) fire-excluded landscape dominated by hardwoods and offsite pines. The number of mature longleaf is not insignificant. There are 1,000+ scattered longleaf that have 50 to 70 rings at breast height, with approximately half the trees with diameter at breast height (d.b.h.) of >6 inches. In addition, there are three dense stands, each approximately 5 acres in size. Hermann and Kush (2006) provide information on size-class distribution of HOBE residual longleaf pine compared to data from nearby uncut forest of 1905 as well as descriptions of the first attempts to reintroduce fire. Evaluation of the site provides the basis for establishing meaningful objectives appropriate for pursing the goal of longleaf forest restoration.

CASE STUDY OF AN EXPANDED DECISION TREE APPROACH TO RESTORATION

1. Information gathered on the site indicates substantial residual longleaf pine that is embedded in a matrix of a degraded forest structure. Based on this assessment, we developed two objectives. The first and immediate objective is to restore the look (structure) of the forest at the time of the battle. The second and more long-term objective is to restore species composition of the forest. The order was determined by ease and time required to tackle each objective.

2. Once the objective of restoring forest structure was established we considered the placement of residual longleaf across the HOBE landscape. Figure 1 displays the Global Positioning System's (GPS) locations of the large (>6 inches d.b.h.) longleaf.

Identification of Longleaf Positions Over the Landscape

1. Visual assessment of figure 1 reveals numerous scattered residual longleaf suggesting that maintaining these trees will be useful in a restoration effort with ecological and conservation values. It also illustrates that simple removal of offsite hardwoods and reintroduction of fire will not be sufficient to restore forest structure because the population of residual longleaf does not meet expectations of a native forest based on what is described in the literature.

   It should be noted that use of GPS technology permitted us to evaluate tree locations on a fine scale but application of the decision process and silvics-based restoration is not dependent on use of the technology. However, it does enhance the value of the graphical displays of this paper.

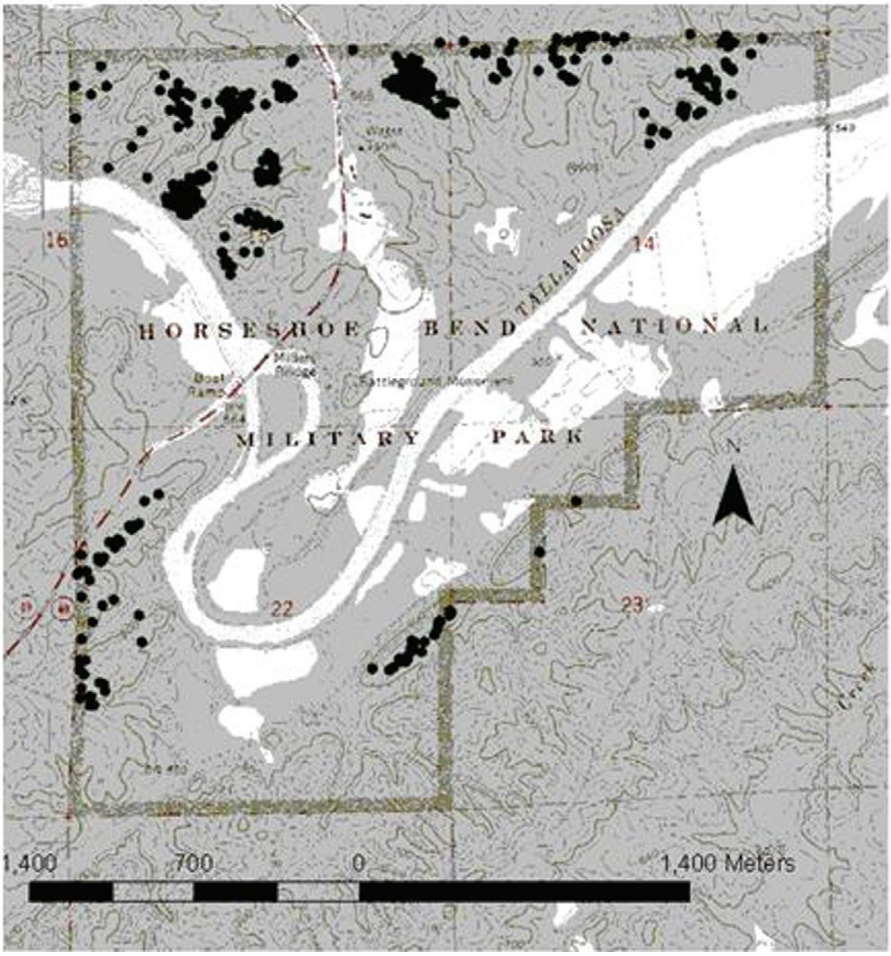

Figure 1—Map of longleaf pine stems (>15 cm d.b.h.) at Horseshoe Bend National Military Park in Tallapoosa County, AL. Map was created using ArcGIS version 9.2.

2. Figure 1 also reveals three dense stands of residual longleaf. This divides the property into areas of dense longleaf trees and no dense stands.

Assessment of Dense Stands of Longleaf

1. Assessment of these stands determined that basal area ranged from 35.6 to 50.7 square feet per acre. Croker and Boyer (1975) suggested that 30 square feet basal area per acre is required for successful natural regeneration of longleaf. This silvic information indicates that the three dense stands do not require supplemental planting in the near future.

2. These stands require reintroduction of fire to eliminate excessive litter and accumulated duff (Hermann and Kush 2010). In the future they may benefit from selective removal of offsite trees. Figure 2 provides a stem map of one of the three stands. Visual inspection as well as measurements of the polygons between longleaf suggests that there are opportunities for future small gap natural regeneration. Neither longleaf nor hardwoods are positioned in a uniform fashion over the stand. In figure 2, one natural gap is obvious and may be sufficient for natural regeneration once bare mineral soil is exposed; minimum gap size is estimated as 0.25 to 0.50 acre (McGuire and others 2001). Over the remainder of the stand selective offsite tree removal eventually may be required if stems have not been eliminated by fire.

Consideration of Hardwoods in the Uplands

1. Many hardwood species are correctly viewed as offsite in longleaf forests, however, Reed (1905) indicated that at sites near HOBE there were scattered hardwood trees prior to regionwide logging and exclusion of fire. To confirm that some hardwood individuals might be appropriate to leave, we cored selected trees.

2. Cores from hardwood trees revealed ages of 150+ for eight trees, primarily post oak (*Quercus stellata*). This information modified initial plans and scattered individual selective hardwoods will be retained. Although rare in 1905 upland forest, hardwoods should be part of a restored landscape.

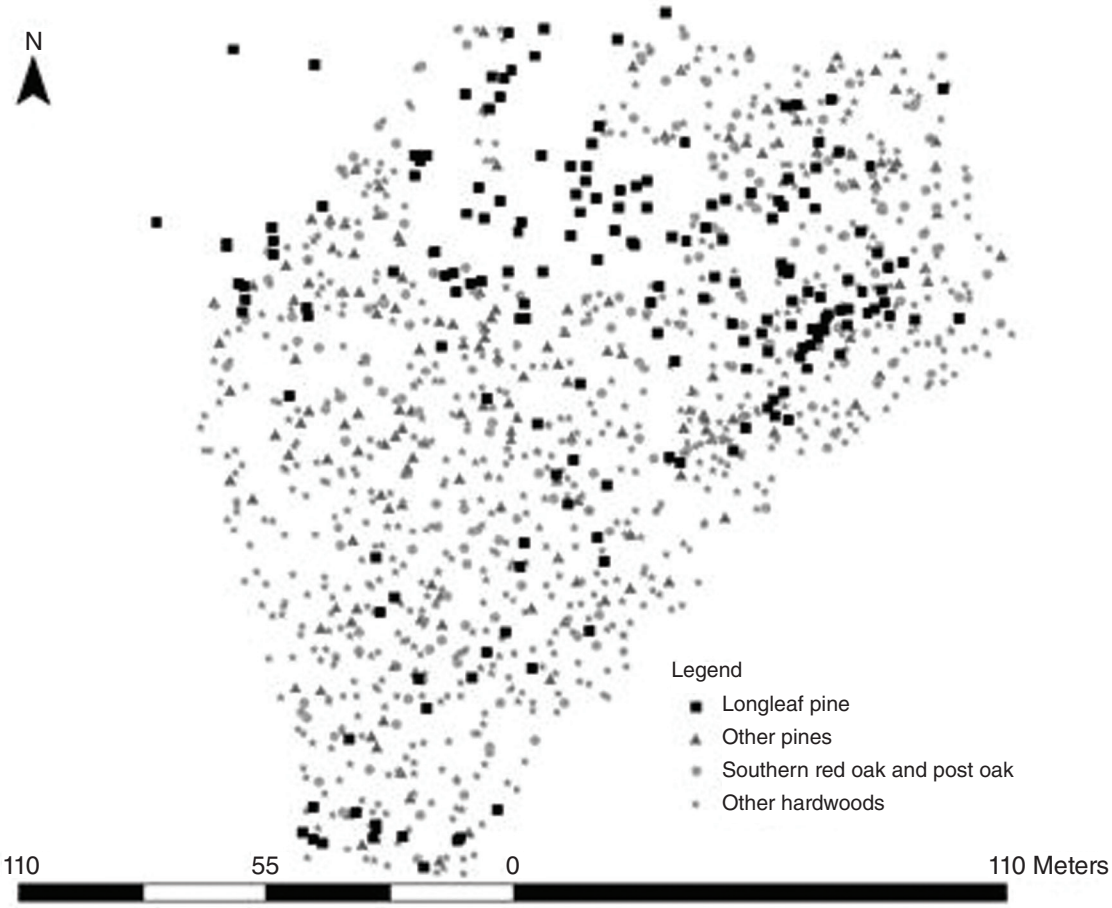

Figure 2—Stem map of a mixed pine hardwood stand that is one of three areas on Horseshoe Bend National Military Park with a dense coverage of longleaf pine. The perimeter of the stand was determined by the presence of longleaf pine.

Assessment of Longleaf Locations Based on Soil Series

1. The dense stands revealed in figure 1 suggest priority areas to target for maintaining residual longleaf on HOBE but it does not address a way to consider areas away from the dense stands. Examination of soil series currently occupied by HOBE longleaf is one way to address this issue. Longleaf is known to occupy a wide range of soil series (Craul and others 2005) but in an effort to target the most likely areas for restoration we determined what soils HOBE longleaf currently occupies. Figure 3 reveals that the HOBE soil landscape is complex with 21 soil series represented and provides no well-defined starting point for restoration. This level of complexity makes it difficult to assess possibilities and so we collapsed soil series into two simple categories: (a) those that currently support longleaf and (b) those that do not (fig. 4).

2. Figure 4 reveals that residual large longleaf are scattered over seven soil series types that occupy approximately 50 percent of HOBE uplands. Although it is very possible that some if not many of the soil types currently without longleaf once supported the tree, we use this distinction to prioritize efforts and elect to burn but not consider non-longleaf soil areas for further immediate consideration for restoration activities.

Do Isolated Longleaf Pines Have Value for Forest Restoration Efforts?

1. Isolated trees are rarely considered as contributing to landscape-level restoration efforts. Their isolation appears to add little to recreating forest structure and they may require time and effort to manage. To assess potential value of isolated longleaf pine trees we consider seed dispersal distance. Boyer (1963) determined that approximately 75 percent of seed would fall within approximately 60 feet of the cone tree and approximately 95 percent of seed was likely to fall within approximately 120 feet of the cone tree. This information serves as the basis for determining what areas are likely to benefit from natural regeneration.

2. Silvics information on longleaf seed dispersal distance permits estimation of areas that will benefit from isolated trees serving as seed sources. Figure 5 zooms in on a small area of residual longleaf. The largest circles surrounding trees illustrates the dispersal limit for 95

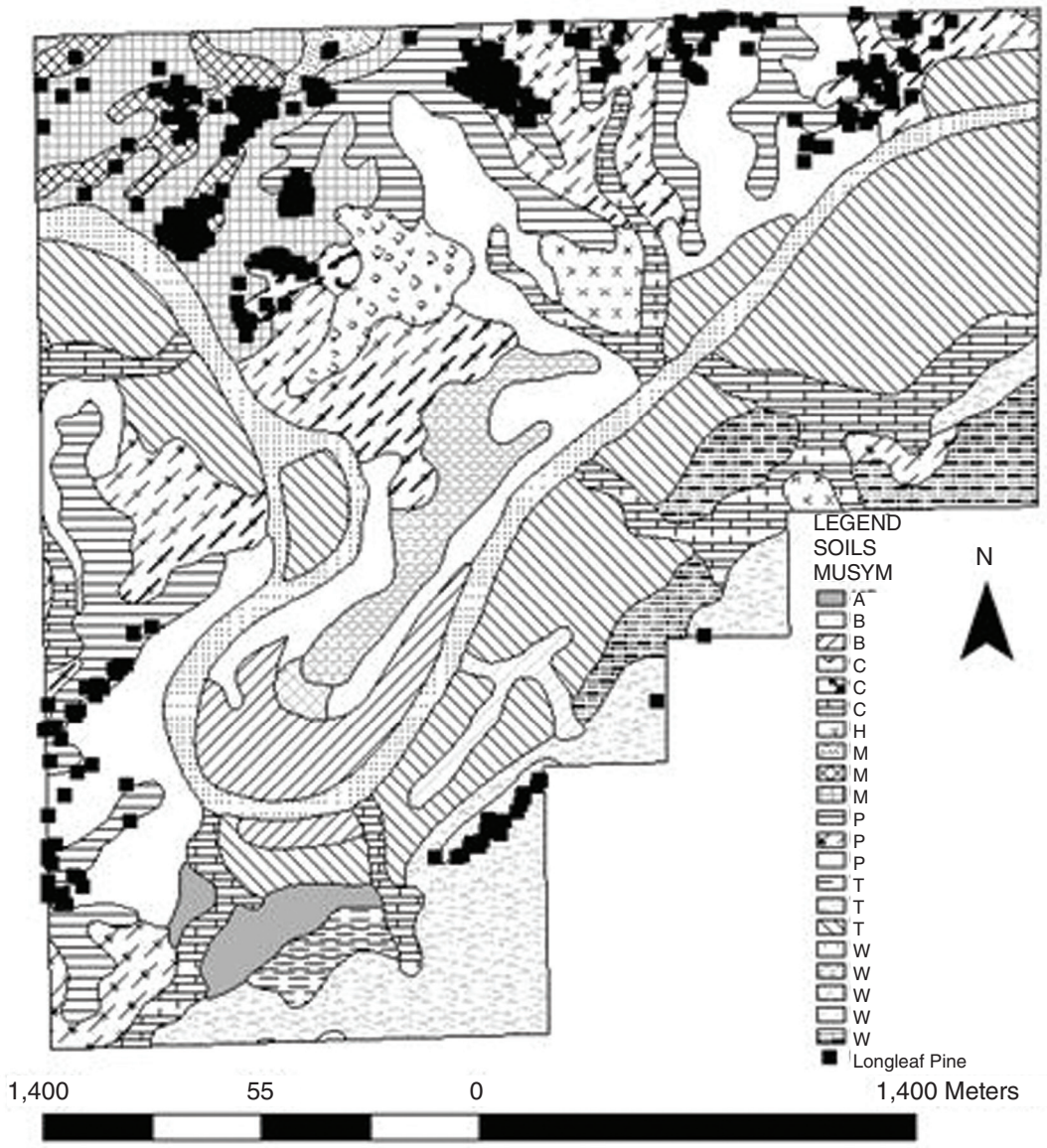

Figure 3—Longleaf pine (>15 cm d.b.h.) mapped with the soil series found at Horseshoe Bend National Military Park. Each dot represents a tree with the black center encompassing the stem plus 20 m (the expected distance of 75 percent of seedfall); the lighter gray circle represents a 40-m radius (the expected distance of 95 percent of seedfall). The soil component of the figure is derived from Web-based Natural Resources Conservation Service material. Map was created using ArcGIS version 9.2.

percent of the seed from each stem. The result is a map of small areas likely to benefit from natural regeneration imbedded in a mosaic of appropriate soils that have little or no possibility of getting seed. The areas outside of seed dispersal become potential areas for additional management effort and perhaps seedling planting. Figure 4 illustrates the entire site with areas likely to benefit from seed dispersal of residual trees. We estimate that slightly <20 percent of the appropriate area will benefit from natural regeneration over the next 40 years (approximate time until there are cone producers derived from natural regeneration). This apparently small benefit from isolated trees coupled with potential challenges of reintroduction of fire may generate concerns that isolated residual longleaf are more trouble than they are worth because they will be expensive to protect during the process of reintroducing fire. This could be true if heroic efforts were made. However, this need not be the case. If no extra effort is made to safeguard them as duff is reduced, any trees that survive are an extra bonus to restoration efforts. Even if only half of the 20 percent benefits from natural regeneration, the HOBE landscape is that much farther along in restoration of forest structure. It should be noted that although figures 5 and 4 are based on GPS data, this approach to assessment could be completed in a low-tech manner relying on trained eyes.

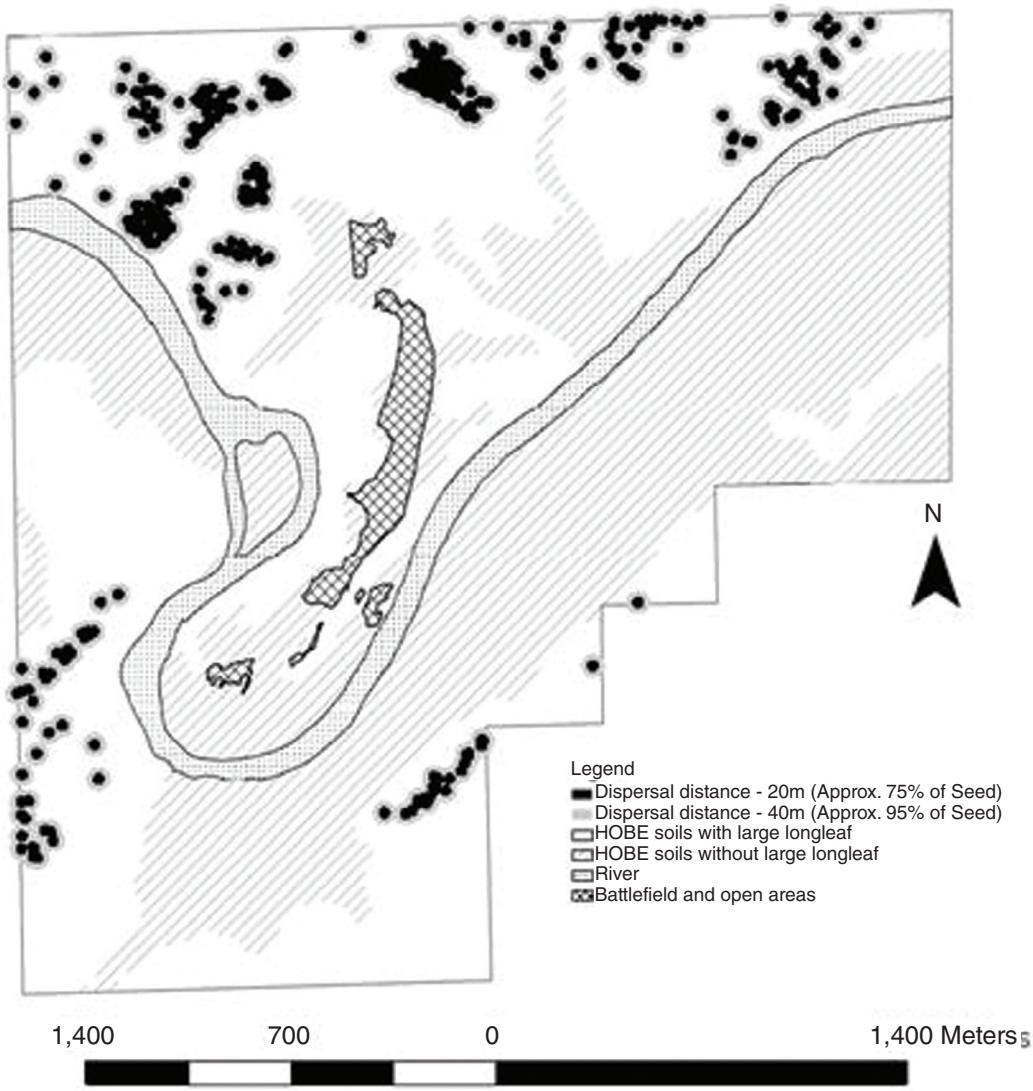

Figure 4—Longleaf pine (>15 cm d.b.h.) mapped with the soil series found at Horseshoe Bend National Military Park. Each dot represents a tree with the black center encompassing the stem plus 20 m (the expected distance of 75 percent of seedfall); the lighter gray circle represents a 40-m radius (the expected distance of 95 percent of seedfall). Map was created using ArcGIS version 9.2.

CONCLUSIONS

Application of the decision tree proposed by the Longleaf Alliance (Johnson 1998) coupled with our recommendations for additional considerations provides a basis for identifying needs of areas within the landscape to be restored as well as a way to prioritize areas and associated actions. Our application of silvics information (seed dispersal distance) coupled with a conservative view of appropriate soils is expected to result in enhanced use of residual trees to meet conservation goals and a more efficient approach to forest restoration planning. Natural regeneration can benefit restoration at HOBE but will not be the primary means for meeting the goal of reestablishing a longleaf forest.

A decision tree approach that includes our additions will identify areas where (a) longleaf pine seedlings should eventually be planted and (b) restoration efforts (duff and litter consumption) coupled with offsite stem removal can create appropriate regeneration gaps with as little effort as possible. Future refinement of this landscape model for restoration could include slope and/or aspect. Reed (1905) noted few longleaf pines on north- or east-facing slopes.

Our additional considerations identified areas with potential to enhance conservation goals that would not have been included in a traditional forest restoration plan. Many plans do not confer value on isolated trees and this is understandable because goals for many sites are not driven by conservation values. In addition cost-effective site prep and planting are dependent on economy of scale. Generally if a major goal for property includes need for income, leaving isolated trees may not generate adequate benefit in the short term. However, if goals are driven in large part by conservation interests, value of isolated trees may be in creating uneven-aged stands as

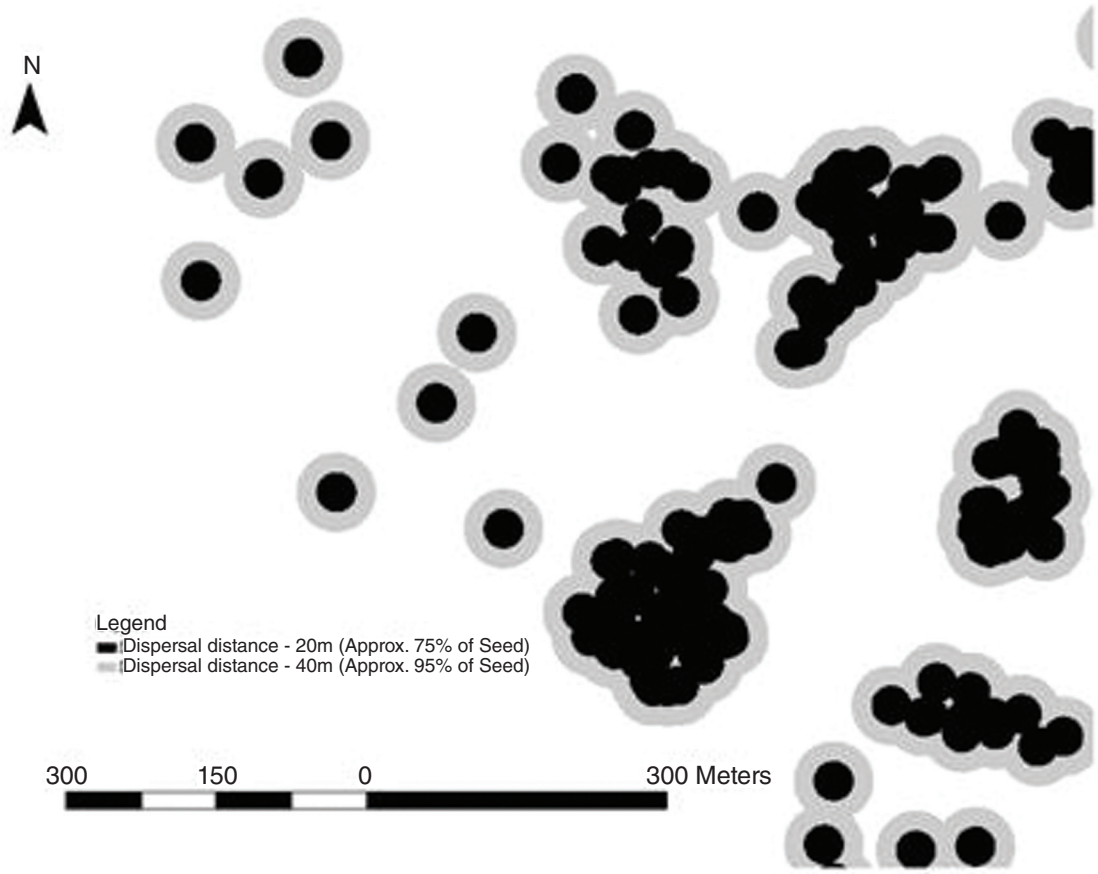

Figure 5—Longleaf pine (>15 cm d.b.h.) in a selected area of Horseshoe Bend National Military Park. Each dot represents a tree with the black center encompassing the stem plus 20 m (the expected distance of 75 percent of seedfall); the lighter gray circle represents a 40-m radius (the expected distance of 95 percent of seedfall).

well as creating potential of areas of natural regeneration. Retention and promotion of isolated trees will decrease time and potentially cost of restoration of a complex multiaged longleaf forest.

In the case of HOBE, discovery of residual dense stands identified areas of high priority for aggressive but careful reintroduction of fire to reduce litter and duff (Kush and others 1998), removing small woody stems and perhaps thinning canopy offsite stems is needed to produce small regeneration gaps. If offsite tree elimination does happen with fire, mechanical treatment should be considered. As a category, isolated longleaf have a moderate likelihood of contributing to forest appearance (structure) restoration. They may be difficult to protect during fire reintroduction but some have moderate potential for assisting in restoration of forest multiage structure, especially on edge of gaps.

Areas outside of dense longleaf stands and beyond dispersal distance of residual trees should be awarded relatively low priority for management actions because with frequent fire they are not likely to degrade further. However action necessary to plant some of them may be relatively easy to carry out, especially with a combination of mechanical and chemical treatments.

By considering the full range of start conditions across the HOBE landscape, we are better able to prioritize action for each condition, plan efficiently, and maximize restoration value of residual resources. In doing so we added a step (consideration) to decision tree developed by Longleaf Alliance (Johnson 1998). We followed their lead and (1) articulated desired future condition, and (2) determined starting points but we then considered how to make the best use of each starting point to achieve various components of the desired future condition. By pursuing this process we hope to restore longleaf pine forest structure to some areas of HOBE within the decade. Multiple states of starting points for restoration are also likely in other open-canopy forests; it may be useful to expand the evaluation and planning to species such as shortleaf (*P. echinata*) and slash pine (*P. elliottii*).

We urge consideration of recovery attempts for degraded stands where possible. This requires effort but may also have significant benefits. Our take-home message is that

in mature fire-excluded stands, even if only some residual trees are retained after reintroduction of fire, stands can be multiaged in a few years. Planted stands with all trees young and the same age requires decades to reach complexity of multiple ages.

ACKNOWLEDGMENTS

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LITERATURE CITED

Boyer, W.D. 1963. Longleaf pine seed dispersal. Res. Note SO-3. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 2 p.

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.

Croker, T.C., Jr.; 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. 26 p.

Hermann, S.M.; Kush, J.S. 2006. Assessment of restoration potential of residual stands of mountain (Piedmont) longleaf pine at Horseshoe Bend National Military Park. In: Cipollini, M.L., comp. Proceedings of the second montane longleaf pine conference workshop. Longleaf Alliance Rep. 9. Andalusia, AL: The Longleaf Alliance: 39–42.

Hermann, S.M.; Kush, J.S. 2010. Initial effects from reintroducing fire in Alabama montane longleaf stands: fifty years since last burn. In: Stanturf, J.A., ed. Proceedings of the fourteenth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-121. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 263–266.

Johnson, R. 1998. Longleaf pine forest restoration. www.longleafalliance.org. [July 1, 2011].

Kush, J.S.; Varner, J.M.; Meldahl, R.S. 1998. Slow down, don't burn too fast ... got to make that old-growth last. In: Proceedings of the 2nd Longleaf Alliance conference. Longleaf Alliance Rep. 4. Andalusia, AL: The Longleaf Alliance: 109–111.

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.

Moser, W.K.; Treiman, T.; Johnson, R. 2003. Species choice and the risk of disease and insect attack: evaluating two methods of choosing between longleaf and other pines. Forestry. 76(2): 137–147.

Outcalt, K.W. 2000. Occurrence of fire in longleaf pine stands in the Southeastern United States. In: Moser, W.K.; Moser, C.F., 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: 178–182.

Reed, F.W. 1905. A working plan for the forest lands in the central Alabama. Bull. 68. Washington, DC: U.S. Department of Agriculture Forest Service.

EFFECTS OF PRESCRIBED BURNING ON SMALL MAMMAL, REPTILE, AND TICK POPULATIONS ON THE TALLADEGA NATIONAL FOREST, ALABAMA

Jonathan Adams, Chris Edmondson, Damien Willis, and Robert Carter[1]

Abstract—A study of the relationship between prescribed burning and tick populations was conducted in the Talladega National Forest, AL. The study area for mammal and tick sampling consisted of 12 plots ranging from the unburned control site to sites burned within the previous 5 years. The study area for reptile sampling consisted of four plots ranging from the unburned control site to sites burned within the previous 5 years. Small mammal trapping yielded a total of 66 individuals and 5 species captured over 2,160 trap nights. One-way analysis of variance (ANOVA) results showed significant difference between *Peromyscus leucopus* populations across burn treatments. A total of 107 reptiles were captured, representing 12 species. Both species richness and Shannon index of diversity were significantly lower in plot 1 (1 month postburn). Both values increased in plot 2 (1 year postburn). A total of 321 individual ticks were collected. Over 90 percent were lone star ticks (*Ambylomma americanum*), with the remaining ticks being American dog ticks (*Dermacentor variabilis*). ANOVA results showed statistically significant changes in tick populations between sites with different burn treatments.

INTRODUCTION

Prescribed burning removes accumulated fuels and, therefore, reduces the risk of intense fires. Many public agencies and some private landowners conduct prescribed burns to restore or improve natural forest conditions (Long 2002). Forests containing longleaf pine (*Pinus palustris*) are frequently burned to maintain habitat suitable for red-cockaded woodpeckers (*Picoides borealis*) and other species. Burning also promotes seed germination, flowering, and resprouting of fire-adapted native plants and generally improves wildlife habitat. Regular burning of rangelands and understory plants improves forage quality and quantity for wildlife and livestock. Fire can also change the physical conditions of a habitat, which select for the types of organisms that inhabit the area (Pilliod and others 2003).

Reptiles and small mammals are reservoirs for many tick-borne pathogens. The objective of this study was to determine how prescribed burning influences small mammal, reptile, and tick populations.

METHODS

Study Area

Talladega National Forest is at the southern terminus of the Appalachian Mountain chain. The Shoal Creek District is located in northeastern Alabama and lies in Calhoun and Cleburne Counties. The Shoal Creek District is comprised of hills and low mountains with steep slopes.

The study area is located near Coleman Lake in the Shoal Creek District. It was selected based on the availability of burn data. The study area consisted of 12 plots ranging from the unburned control sites to sites burned within the previous year.

Small Mammal Collection

Twelve sites were chosen based on length of time elapsed since a prescribed burn—1 month, 1 year, 5 years, and an unburned control. Three sites were selected for each time period. Small mammals were collected on each site using Sherman live traps arranged in a 3 by 3 grid with trap spacing at 20 m apart. Traps were baited with sunflower seeds and checked each morning of the trapping period. Trapping was conducted for 5 nights for 4 months during the new-to-quarter moon phase to minimize the possible effect of moonlight on capture. Animals collected were identified to species then tagged and released at the site of capture. Animal collection data were separated by treatment type.

Due to small sample size, an index of population size was used instead of a statistical population estimator. The number of individuals captured per 100 trap nights (*n* individuals/*n* trap nights * 100) was calculated for each species and for all species combined (Yates and others 1997). One-way ANOVA was used to compare the *Peromyscus leucopus* populations between treatments. Also the Shannon-Weiner diversity index was calculated and compared using a t-test.

Reptile Collection

Beginning in April 2008, two unbaited pitfall arrays with funnel traps were installed in each of four plots. These plots had varying burn histories—plot 1 was burned 1 month prior to study, plot 2 was burned 1 year prior to study, plot 3 was burned 5 years prior to study, and plot 4 was the unburned control site. Traps were opened beginning in May of 2008 and remained open throughout the duration of the study. The traps were checked approximately every 4 days from May to October. Any reptiles that were captured were identified, measured, and marked using toe-clipping method.

Plots were located along, or in the vicinity of, Forest Service Road 500 located in the Shoal Creek Ranger District in the Talladega National Forest. Plot 1 was a predominantly longleaf pine community with a 22-percent canopy cover. Plot 2 was also a longleaf community. Canopy cover in this plot was 33 percent. Plot 3 was a mixed hardwood-pine forest with a canopy cover of 88 percent. Plot 4 had a closed-canopy mixed hardwood-pine forest.

[1]Graduate students and Associate Professor, Jacksonville State University, Department of Biology, Jacksonville, AL, respectively.

Raw data was analyzed using species richness, Kruskal-Wallis one-way ANOVA, and Shannon index of diversity.

Tick Collection

Tick samples were collected at the same 12 sites used for small mammal trapping. Ticks were collected using the drag cloth method (Goddard 2007, Milne 1943). The cloths used were approximately 1- by 1-m squares of white flannel material attached to a dowel on one end and lightly weighted on the other to maximize contact with the vegetation. Each site was dragged for 20 minutes each month (May through September). The cloths were inspected approximately every 10 m during the sampling, and any ticks found on the cloths were removed and preserved in 70 percent ethanol for future processing. Ticks were identified by species and life stage (adult or nymph).

Tick collection data was separated by treatment type. One-way ANOVA analysis was used to compare populations between treatment types.

RESULTS

Small Mammal

A total of 66 individuals and 5 species were captured over 2,160 trap nights. *Peromyscus leucopus* was the most common species captured with 48 captures. This species was captured on all treatment types. One-way ANOVA results showed significant differences between *Peromyscus leucopus* populations when comparing the unburned control vs. 5-year treatments, unburned control vs. 1-year treatments, and unburned control vs. 1-month treatments (table 1).

Combined species abundance was highest in plots that were burned within the previous year (6.11 individuals per 100 trap nights). Species abundance was lowest in plots that were unburned (0.56 individuals per 100 trap nights) (table 2). Species diversity and richness was greater in plots that were burned 1 month and 1 year ago (0.9434 and 0.8223, respectively) than those that were unburned and burned 5 years ago (0.6365 and 0.5196, respectively) (table 3).

Reptile

A total of 107 reptiles were captured, representing 12 species. Traps located in plot 1 (burned 1 month prior to study) captured 39 reptiles from 5 species. Dominant species were eastern fence lizard (*Sceloporus undulatus*) and broadhead skink (*Eumeces laticeps*). Traps located in plot 2 (burned 1 year prior to study) captured 19 reptiles from 8 species. Dominant species were the southeastern five-lined skink (*Eumeces inexpectatus*) and broadhead skink. Traps located in plot 3 (burned 5 years prior to study) captured 18 reptiles from 7 species. Dominant species were eastern fence lizard, eastern worm snake (*Carphophis amoenus*) and green

Table 1—Number of small mammals captured by treatment type in Shoal Creek District of Talladega National Forest, AL

Treatment	Total individuals	*Peromyscus leucopus*	*Mus musculus*	*Sigmodon hispidus*	*Tamias striatus*	*Ochrotomys nuttalli*
Unburned control	3	2	1	0	0	0
5 years	14	11	0	0	0	3
1 year	16	12	0	1	2	1
1 month	33	23	1	6	2	1
Total	66	48	2	7	4	5

Table 2—Number of small mammals captured per 100 trap nights by treatment type in Shoal Creek District of Talladega National Forest, AL

Treatment	*Peromyscus leucopus*	*Mus musculus*	*Sigmodon hispidus*	*Tamias striatus*	*Ochrotomys nuttalli*	All species
Unburned control	0.37	0.19	0	0	0	0.56
5 years	2.04	0	0	0	0.56	2.59
1 year	2.22	0	0.19	0.37	0.19	2.96
1 month	4.26	0.19	1.11	0.37	0.19	6.11

Table 3—Species richness and Shannon's index of diversity for small mammals by treatment type in Shoal Creek District of Talladega National Forest, AL

Treatment	Species richness	Shannon index of diversity
Unburned control	2	0.6365
5 years	2	0.5196
1 year	4	0.8223
1 month	5	0.9434

anole (*Anolis carolinensis*). Traps located in plot 4 (unburned control) captured 31 reptiles from 8 species. Dominant species were eastern fence lizard and green anole (table 4).

According to Kruskal-Wallis one-way analysis of variance, plot 1 was found to be significantly different from plot 2 and plot 4 ($P = 0.041$ and $P = 0.099$, respectively). Comparisons of all other sites showed no significant differences.

Both species richness and Shannon index of diversity were lower in plot 1, which was burned 1 month prior to study. Both values increased in plot 2, with species richness stabilizing and species diversity steadily decreasing (table 5).

Tick

A total of 321 individuals were collected. Over 90 percent were lone star ticks (*Ambylomma americanum*), with the remaining ticks being American dog ticks (*Dermacentor variabilis*). ANOVA results showed statistically significant changes in tick populations between sites with 1-month burn treatments and sites with 5-year burn treatments. There was also a significant difference in the populations at sites 1-month postburn and unburned treatments. There was little to no difference between populations at sites with 1-month and 1-year burn treatments. There was also no significant difference in tick populations between the sites with 1-year burn treatments and sites with 5-year or unburned treatments (table 6).

Table 4—Number of reptiles captured in each plot in Shoal Creek District of Talladega National Forest, AL

Species	Plot 1	Plot 2	Plot 3	Plot 4	Total
Anolis carolinensis	3	0	3	5	11
Carphophis amoenus	0	0	3	0	3
Coluber constrictor	0	1	0	0	1
Diadophis punctatus	0	0	1	0	1
Elaphe obsoleta	0	1	0	1	2
Eumeces fasciatus	2	2	0	2	6
Eumeces inexpectatus	3	5	0	1	9
Eumeces laticeps	7	4	1	4	16
Sceloporus undulatus	24	3	7	16	50
Scincella lateralis	0	2	2	1	5
Tantilla corona	0	1	0	0	1
Thamnophis sartalis	0	0	1	1	2
Total	39	19	18	31	107

Table 5—Species richness and Shannon's index of diversity by treatment type for reptiles in Shoal Creek District of Talladega National Forest, AL

Treatment	Species richness	Shannon index of diversity
Plot 1	5	1.226
Plot 2	8	1.875
Plot 3	7	1.69
Plot 4	8	1.528

CONCLUSIONS

Mammal

Recently burned plots had a high number of *Peromyscus leucopus* captured. According to Dickson (2001), white-footed mice quickly invaded pine plantations and were the most abundant small mammals the first year after a burn. *Peromyscus* spp. usually are the earliest invader of young stands due to increased ground vegetation postburn. Higher *Peromyscus* spp. populations on burned sites have been attributed to better visibility and abundance of seed, a food source for the mice, after reductions in litter cover and depth (Greenberg and others 2006).

In unburned sites, there were a low number of animal captures. Small mammal populations have been shown to be lower in older, more open forested areas that have little vegetation close to the ground. Mature forest stands with closed canopies and little herbaceous vegetation usually support a relatively low density of small mammals (Dickson 2001).

Reptile

High capture numbers found in plot 1 may be attributed to higher summer temperatures associated with open canopies. However, low species diversity in plot 1 can be attributed to a dominance of *Sceloporus undulatus*, a species which may seek refuge under objects during fires (Kahn 1960, Lillywhite and North 1974).

Richness and diversity initially increase after a prescribed burning treatment. If left undisturbed, species diversity decreases over time. This is due to the closing of the forest canopy, and the subsequent dominance of reptile species associated with this environment (Greenberg and others 1994).

Table 6—Number of ticks collected by month and treatment type in Shoal Creek District of Talladega National Forest, AL

Treatment	May	June	July	August	September	Total
	----------	----------	---------- *number*	----------	----------	----------
1-month burn	69	5	3	3	0	80
1-year burn	15	9	3	23	4	54
5-year burns	19	50	14	1	4	88
Unburned control	44	14	30	7	4	99

Tick

Burning significantly reduced populations temporarily, but numbers returned to normal within 2 to 5 years after fire disturbances. Annual and biennial prescribed burning has been shown to significantly reduce tick populations at all life stages (Davidson and others 1994). Fire mortality and reduction of leaf litter in burned areas are possible reasons for the reduction in tick populations.

While burn history is important, there are likely multiple factors determining tick populations at a given location.

LITERATURE CITED

Davidson, W.R.; Siefken, D.A.; Creekmore, L.H. 1994. Influence of annual and biennial prescribed burning during March on the abundance of *Ambylomma americanum* (Acari: Ixodidae) in central Georgia. Journal of Medical Entomology. 31(1): 72–81.

Dickson, James G. 2003. Wildlife of southern forests habitat & management: terrestrial small mammals. In: Dickson, James G., comp. ed. Wildlife of southern forests habitat & management. Blaine, WA: Hancock House Publishers: 350–358. Chapter 26.

Goddard, J. 2007. Seasonal activity of *Amblyomma* spp. in Mississippi. Journal of Vector Ecology. 32: 157–158.

Greenburg, C.H.; Neary, D.G.; Harris, L.D. 1994. Effect of high-intensity wildfire and silviculture treatments on reptile communities in sand-pine scrubs. Conservation Biology. 8: 1047–1057.

Greenberg, Cathryn; Otis, D.L.; Waldrop, T.A. 2006. Response of white-footed mice (*Peromyscus leucopus*) to fire and fire surrogate fuel reduction treatments in a Southern Appalachian hardwood forest. Forest Ecology and Management. 234(1–3): 355–362.

Kahn, W.C. 1960. Observations on the effect of a burn on a population of *Sceloporus occidentalis*. Ecology. 41: 358–359.

Lillywhite, H.B.; North, F. 1974. Perching behavior of *Sceloporus occidentalis* in recently burned chaparral. Copeia. 1977: 115–121.

Long, Alan. 2002. Benefits of prescribed burning. Publ. FOR 70. Gainesville, FL: University of Florida IFAS Extension. http://edis.ifas.ufl.edu/fr061. [Date accessed: July 5, 2011].

Milne, A. 1943. The comparison of sheep-tick populations (*Ixodes ricinus* L.). Annals of Applied Biology. 30: 240–250.

Pilliod, D.S.; Bury, R.B.; Hyde, E.J. [and others]. 2003. Fire and amphibians in North America. Forest Ecology and Management. 178: 163–181.

Yates, Mark; Loeb, S.C.; Guynn, D.C., Jr. 1997. The effect of habitat patch size on small mammal populations. Proceedings of the Annual Conference of Southeastern Association of Fish and Wildlife Agencies. 51: 501–510.

SITE INDEX MODEL FOR NATURALLY REGENERATED EVEN-AGED LONGLEAF PINE

Dwight K. Lauer and John S. Kush[1]

Abstract—Data from the Regional Longleaf Growth Study (339 permanent sample plots) were used to develop a site index model for naturally regenerated, even-aged longleaf pine (*Pinus palustris* Mill.). The site index equation was derived using the generalized algebraic difference approach and is base-age invariant. Using height as a measure of site productivity in naturally regenerated longleaf pine is confounded by the variability in the number of years an individual tree remains in the grass stage. This site index equation uses ring count age to model height development from 4.5 feet but can be modified to use tree age based on ring count age and years it takes trees to reach 4.5 feet.

INTRODUCTION

Height development in naturally regenerated even-aged longleaf pine (*Pinus palustris* Mill.) under shelterwood management is complicated by the fact that ring count age at d.b.h. for a given cohort includes variability in year of germination, length of time in the grass stage, and number of years it takes a tree to reach 4.5 feet. Site index equations typically model the grass stage of longleaf pine as slow early growth and/or make assumptions about the length of the grass stage. Early height predictions and estimates of site productivity would be improved by treating early longleaf development as a discrete event before the initiation of height growth. The difficulty is that site index typically uses stand age.

The objective of this project was to improve current site index models using more recent modeling methodology. Base-age invariant site index modeling methods allow the fitting of curves using ring count age and the substitution of stand age based on measurements or assumptions about years added to ring count age. Estimates based on early height development were improved through the use of a polymorphic equation and the modeling of individual tree height trajectories instead of using plot average height and average ring count age.

DATA

A subsample of dominant and codominant trees were measured on permanent plots since 1964 by Auburn University, Mississippi State University, and other public owners as part of a U.S. Forest Service, Southern Research Station cooperative study investigating production of thinned, even-aged, naturally regenerated stands in the east Gulf region of the Southern United States. Plots were initially selected to fill an array of cells with five 20-year classes, five 10-foot site index classes, and five 30-square-foot basal area classes (Farrar 1993). Plots were measured every 5 years and additional plots were added as the study progressed. A total of 2,014 plot measurements have been completed covering a wide range of age and height classes.

Dominant and codominant trees selected from this population of sampled trees had to have been measured at least three times, measured on at least half the number of all measurements taken on a given plot, and must have been measured at least to the second-to-the-last plot measurement. The trees must have been classified as dominant or codominant at all measurements. This resulted in a dataset with 19,527 measurements on 3,267 trees distributed over 285 plots.

ANALYSIS

Longleaf pine height development in young stands is characterized by nearly linear and rapid height growth once trees are >4.5 feet tall. Height growth slows with age but continues at a very slow rate at ages >90 years. A model was selected that was appropriate for this height development pattern. The generalized algebraic difference approach was used to develop a base-age invariant polymorphic model as discussed in detail by Cieszewski and Bailey (2000).

The model was fit to individual tree data using a dummy variable approach. This method estimates global parameters that are the same for all trees and define the shape of the site index curve. A parameter, or random effect, was estimated for each tree to determine the instance of the site index curve for an individual tree. A first-order autoregressive error process was used to account for serial correlation of repeated measurements on a tree. Models were fit using the SAS/ETS model procedure (SAS Institute Inc. 2004).

GENERALIZED EQUATIONS

The fitted models that describe height development >4.5 feet based on ring count age were generalized to use total height and stand age. The site index equation is:

$$S_{rc+G} = 4.5 + \frac{b_1 + X_0}{1 + \frac{b_2}{X_0}(B_{rc+G} - G)^{-b_3}} \qquad (1)$$

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where

$$X_0 = 0.5\left(H - 4.5 - b_1 + \sqrt{(H - 4.5 - b_1)^2 + 4b_2(H - 4.5)(age - G)^{-b3}}\right)$$

The height equation is:

$$H = 4.5 + \frac{b_1 + X_0}{1 + \frac{b_2}{X_0}(age - G)^{-b_3}} \qquad (2)$$

where

$$X_0 = 0.5\left(\left(S_{rc+G} - 4.5\right) - b_1 + \sqrt{((S_{rc+G} - 4.5) - b_1)^2 + 4b_2(S_{rc+G} - 4.5)(B_{rc+G} - G)^{-b3}}\right)$$

In these equations, *age* is stand age in years, *G* is the age at which trees reach 4.5 feet (or stand age minus ring count age), *rc* is ring count age, *S* is site index, *B* is site index base age, and *H* is total height. Estimates of the parameters are b_1 = 77.080, b_2 = 1723.39, and b_3 = 1.235. The subscripts for *S* and *B* indicate how base age is referenced. S_{43+7} would indicate site index base age 50 with a ring count of 43 years and 7 years to reach 4.5 feet.

These equations allow the user to clearly define height development in longleaf stands if both stand age and ring count age are known. Note that base age 50 has very different implications if stands are out of the grass stage in 2 years as opposed to 5 years. Care must be exercised when using this equation to compare management regimes or comparing longleaf height development to that of other pine species. The subscripting of site index and base age allows the clear understanding of what is being assumed.

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LITERATURE CITED

Cieszewski, C.J.; Bailey, R.L. 2000. Generalized algebraic difference approach: theory based derivation of dynamic site equations with polymorphism and variable asymptotes. Forest Science. 46(1): 116–126.

Farrar, R.M., Jr. 1993. Growth and yield in naturally regenerated longleaf pine stands. In: Hermann, Sharon M., ed. Proceedings of the Tall Timbers fire ecology conference. No. 18. The longleaf pine ecosystem: ecology, restoration and management. Tallahassee, FL: Tall Timbers Research Station: 311–335.

SAS Institute Inc. 2004. SAS/ETS 9.1 user's guide. Cary, NC: SAS Institute Inc.

SURFACE SOIL ROOT DISTRIBUTION AND POSSIBLE INTERACTION WITH SITE FACTORS IN A YOUNG LONGLEAF PINE STAND

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Abstract—Interaction between soil bulk density and low soil water content may create root growth-limiting soil strengths. In a Louisiana longleaf pine (*Pinus palustris* Mill.) stand, soil strength at the zero- to 20.0-cm depth was assessed in response to no fire or biennial fires in May. At the 5.0- to 20.0-cm depth, one-half of the measurements were characterized by root growth-limiting soil strengths regardless of fire history. Where soil strengths were root growth limiting, pine fine root biomass was about 24 percent lower than where soil strengths were not root growth limiting. Correlation between soil strength and pine fine root biomass was only observed where samples were collected distal to the longleaf pine trees, where soil strengths were high, and where biennial fire was applied. Further research is needed to determine whether repeated fire interacts with the relationship between soil strength and longleaf pine root growth on the west Gulf Coastal Plain.

INTRODUCTION

Soils in the western range of longleaf pine (*Pinus palustris* Mill.) are frequently characterized as poorly drained and fine textured (Peet 2006). High bulk densities are likely when soil texture is dominated by silt and clay (Fisher and Binkley 2000). Recently on the Kisatchie National Forest in central Louisiana, bulk densities of typical silt loam soils averaged 1.4, 1.5 to 1.6, and 1.6 to 1.7 g/cm^3 for the A, B1, and B2 horizons, respectively (Patterson and others 2004, Sword Sayer 2007). Bulk densities >1.6 g/cm^3 are known to restrict pine root elongation (Kelting and others 1999, Pritchett 1979). These root growth-limiting bulk densities are countered by root elongation along interped spaces and in macropores created by old roots and soil fauna (Van Lear and others 2000). Fortunately, these attributes also introduce spatial variation into bulk density measurements so that extreme values are not constant over large areas.

During periods of sparse precipitation, low soil water content (SWC) interacts with bulk density to increase soil strength. Soil strength is the force required to advance through soil (Bennie 1996), and values >2000 kilopascals (kPa) are known to inhibit root elongation (da Silva and others 1994, Taylor and others 1966). When low precipitation evolves into drought, the negative effects of soil properties on pine root elongation are potentially far reaching on the west Gulf Coastal Plain. In effect, the land base with root-restricting soil characteristics is widened to include not only areas with high bulk densities but also areas that develop root growth-limiting soil strengths as the soil dries. Once again, in this situation, conduits produced by interped spaces, old root channels, and soil fauna allow root foraging for water and mineral nutrients.

Efforts to restore longleaf pine ecosystems have been successful, in part, by the renewed use of fire as a management tool (Brockway and Lewis 1997). In some situations, repeated prescribed fire reduces understory woody vegetation but increases the growth of herbaceous plants and grasses (Haywood and others 2001). It is hypothesized that by manipulating understory vegetation, repeated fire also changes the amount and distribution of soil macropores that serve as conduits for pine root elongation. This, in turn, could affect soil strength, its spatial variability, and the relationship between soil strength and pine root elongation. As an initial step toward understanding the relationship between soil strength and longleaf pine root growth, the present study was conducted to survey soil strength, longleaf pine fine root biomass (FRB), and their relationship where competing vegetation was not controlled and where biennial prescribed fire was applied in May.

MATERIALS AND MEHODS

Study Site

The study is located on the Kisatchie National Forest in central Louisiana at latitude 31°0'42.45" N, longitude 92°37'8.54" W. The soil is a Beauregard silt loam and Malbis fine sandy loam complex. A mixed pine-hardwood forest originally occupying the site was clearcut harvested in the mid-1980s, repeatedly burned, sheared and windrowed in 1991, and rotary-mowed in 1992 (Haywood 2002).

In 1992, 15 treatment plots [22 by 22 m (0.048 ha)] were established and assigned 1 of 3 vegetation management treatments (no plant control, herbicide application, or mulching after planting) (Haywood 2002). In February 1993 and January 1994, one-half of each plot was planted at 1.8 by 1.8 m with container-grown longleaf pine seedlings from a Mississippi source. By age 3 to 4 years, seedlings were overtopped by competing vegetation in spite of the vegetation management treatments (Haywood 2002). Competing vegetation was manually and chemically eradicated in 1997 and 1998, respectively.

In 1998, analyses of variance indicated that tree growth was significantly affected by vegetation management treatment but not by block, age, or their interaction with vegetation management treatment (Haywood 2002). The study was subsequently reconfigured with each of the original vegetation management treatments as one of three blocks, and random assignment of one of five treatments to each plot per block.

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The new study utilized a randomized complete block design with three blocks. Blocks were delineated by the former vegetation management treatments. New treatments were one of five management activities: (1) control (C)—no vegetation management after 1998, (2) herbicide—biennial application as needed beginning in 1999 at age 5 to 6 years of triclopyr herbicide to competing woody vegetation as a direct foliar spray in May, (3) prescribed fire in March—biennial burning in March, (4) prescribed fire in May (MB)—biennial burning in May, and (5) prescribed fire in July—biennial burning in July. Fires were applied as strip headfires in 1999, 2001, 2003, and 2005.

Before application of the prescribed fire scheduled for March 2007 and at age 13 to 14 years, a wildfire burned the entire study area on March 22, 2007. Based on a postfire survey, the fire burned intensely over the entire study area consuming nearly all living foliage and small woody stems within 1 m of the ground, and longleaf pine crown scorch was over 50 percent.[2] In each of the C and MB plots, 4 subplots were delineated, i.e., 2 treatments, 3 blocks, and 4 subplots per plot for a total of 24 subplots. Subplots contained four adjacent trees in two interior rows of two trees each, so that the dimension of the subplots was 1.8 by 1.8 m. Subplots contained four live trees with some live, unscorched crown, and avoided areas where the majority of the trees were missing or where stump holes and animal burrows were found.

Root Measurements

Root distribution was evaluated with 12 soil cores that were nearby, i.e., proximal, and 12 soil cores that were distant from, i.e., distal, the 4 corner trees of each subplot. The six proximal soil cores were collected from around the two trees in each subplot having the most similar diameters at breast height. Proximal core locations were equidistant around the circumference, and 30 to 45 cm from the base of each of these two trees. On the interior of each subplot, distal soil cores were collected at 12 locations >45 cm from the base of the 4 corner trees. All soil coring for root biomass was done in November 2007.

Proximal and distal soil cores were 20 cm deep and were extracted with a tractor-mounted hydraulic probe (5.1 cm diameter). Cores were partitioned into six depth intervals—i.e., zero to 2.5 cm, 2.5 to 5.0 cm, 5.0 to 7.5 cm, 7.5 to 10.0 cm, 10.0 to 15.0 cm, and 15.0 to 20.0 cm—in the field with a box cutter knife. Soil samples in each subplot were pooled by proximity, i.e., proximal and distal samples, and depth so that 48 soil samples were collected per plot.

Root biomass was removed from soil samples by wet sieving (1 mm^2 mesh). Pine roots were distinguished from nonpine roots based on diameter, color, plasticity, and the appearance of lateral roots and ectomycorrhizae. Using digital calipers, fine plus small pine roots, zero ≤2 mm in diameter, i.e., fine roots, were separated from root samples, ovendried to equilibrium at 70 °C, ground in a Wiley mill (1 mm^2 mesh), and combusted (450 °C, 8 hours) to obtain ash-free dry weights. Fine pine root biomass was expressed as mg/cm^3.

Soil Strength Measurements

Pairs of soil strength and SWC measurements were taken six times in June through September 2008. Soil strength was measured with a CP40II cone penetrometer equipped with a 130-mm^2 tip (Agridry Rimik Pty Ltd., Queensland, Australia). At each of the six measurement times, a soil strength profile was generated at one location around the circumference of each of the two measurement trees per subplot. Soil strength profiles were the average of five inserts within a 20-cm radius, 25 to 30 cm from the base of the stem, and at least 20 cm away from where soil cores were extracted for root samples. The soil strength data for each insert was recorded at 1-cm intervals to a 20-cm depth, and averaged by depth interval and measurement tree. For each measurement tree and time, the soil strength of each of the six depths where roots were sampled was calculated as the average of the appropriate 1-cm interval data—i.e., zero to 2.5 cm: depth intervals 1, 2, and 3 cm; 2.5 to 5.0 cm: depth intervals 3, 4, and 5 cm; 5.0 to 7.5 cm: depth intervals 6, 7, and 8 cm; 7.5 to 10.0 cm: depth intervals 8, 9, and 10 cm; 10.0 to 15.0 cm: depth intervals 11, 12, 13, 14, and 15 cm; 15.0 to 20.0 cm: depth intervals 16, 17, 18, 19, and 20 cm.

Air pockets, plinthite, charcoal, and abrupt changes in resistance caused by semi-impenetrable soil layers led to outliers in the raw data which were eliminated in three ways. First, for each set of raw soil strength data, “0” values were changed to missing data. Second, for each set of raw soil strength data per 1-cm interval and measurement time, i.e., 240 observations, data outside two standard deviations of the mean were changed to missing data. Third, for each of the six soil layers and measurement times, mean soil strength outside two standard deviations of the mean was changed to missing data. These three actions deleted approximately 4 percent of the raw soil strength data that was paired with SWC data.

At each measurement interval, one soil core (25 cm long, 6.5 cm diameter) was extracted from the 20-cm radius where five soil strength inserts were performed. This was done manually with a metal coring device (Veihmeyer 1929). Cores were partitioned into six depth increments using a box cutter knife, i.e., zero to 2.5, 2.5 to 5.0, 5.0 to 7.5, 7.5 to 10.0, 10.0 to 15.0, 15.0 to 20.0 cm. Mineral soil from each depth increment was put into preweighed tins. Capped tins containing wet soil were weighed, uncapped and dried at 105 °C for 24 hours, and reweighed. Gravimetric SWC was calculated which represented SWC when soil strength measurements were taken. Again, artifacts, e.g., large decomposing roots and old root channels, caused outliers in the SWC data. Data quality was refined by excluding data outside two standard deviations

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of the mean for each depth and measurement time. This action affected approximately 5 percent of the raw SWC data.

With the refined soil strength and SWC data, a linear regression equation describing the relationship between soil strength and SWC was developed for the 6 depths in each of the 24 subplots. Each equation was based on 6 data points from each of the 2 measurement trees per subplot, i.e., 12 observations. For regression equations with coefficients of determination, i.e., R^2, that were significant at the alpha-level of 0.05, the soil strength at 16 percent SWC, i.e., SS_{16}, was predicted. The 95-percent prediction interval for SS_{16} was determined, i.e., SS_{16} PI (Neter and Wasserman 1974), and SS_{16} PI were scaled by equation 1 so that the variation associated with SS_{16} could be compared across the range of predicted SS_{16} values.

$$SS_{16}\ PI_{scaled} = (SS_{16}\ PI/2)/SS_{16} \qquad (1)$$

Statistical Analyses

The soil strength profile of each subplot was visually assessed, and subplots were partitioned into two groups: those with the majority of SS_{16} ≤2000 kPa, i.e., low SS_{16} subplots, and those with the majority of SS_{16} >2000 kPa, i.e., high SS_{16} subplots. For the low and high SS_{16} subplots and C and MB treatments, the mean and standard deviation of SS_{16}, SS_{16} PI_{scaled}, and proximal and distal FRB were calculated for each of the six depth intervals.

Because all SS_{16} values at the 2.5- to 5.0-cm depth and 20 percent of the SS_{16} values at the 15.0- to 20.0-cm depth were <2000 kPa on the high SS_{16} subplots, regressions between SS_{16} and FRB excluded data from the 2.5- to 5.0- and 15.0- to 20.0-cm depths. Simple linear relationships between SS_{16} and either distal or proximal FRB at the 5.0- to 15.0-cm depth were evaluated by ordinary least squares regression for the low and high SS_{16} subplots on the C and MB plots. Residuals were assessed for normality by the Shapiro-Wilk statistic (SAS Institute Inc. 2000), and as a result, FRB was transformed to natural logarithm values to insure that residuals were normally distributed. The F statistics associated with R^2 values were considered significant at an alpha-level of 0.05.

RESULTS

Evaluation of relationships between soil strength and FRB was done using SS_{16} that was predicted with equations exhibiting a significant R^2. The R^2 value of these equations was significant for 5 of the 6 depths and for the majority of the 24 subplots (table 1).

There were 12 low SS_{16} subplots, i.e., 5 C and 7 MB subplots, and 12 high SS_{16} subplots, i.e., 7 C and 5 MB subplots. For the low and high SS_{16} subplots, mean SS_{16} was higher at the 5.0- to 15.0-cm depth than at the 2.5- to 5.0- or 15.0- to 20.0-cm depths (fig. 1A). Across the 2.5- to 20.0-cm depth, SS_{16} averaged 31 percent less on the low SS_{16} subplots compared to the high SS_{16} subplots. Within the low and high SS_{16} subplots, mean SS_{16} at each depth was similar between the C and MB treatments. For the low and high SS_{16} subplots, mean SS_{16} PI_{scaled} at each depth was similar between the C and MB treatments (fig. 1B). At the 10.0- to 15.0-cm and 15.0- to 20.0-cm depths, there was a trend for mean SS_{16} PI_{scaled} to be larger on the low SS_{16} subplots compared to the high SS_{16} subplots.

Values of FRB were greatest in the zero- to 5.0-cm depth and decreased with depth to 20.0 cm (figs. 2A and 2B). Across the C and MB treatments and the distal and proximal locations 17, 27, 12, 33, 36, and 17 percent more FRB was observed on the low SS_{16} subplots than the high SS_{16} subplots for the six depth intervals, respectively.

Among the four linear regressions between SS_{16} and proximal FRB, none were significant (figs. 3A and 3B). Where subplot SS_{16} was low, the two linear regressions between SS_{16} and distal FRB were not significant (fig. 3C). Where subplot SS_{16} was high, linear regression between SS_{16} and distal FRB was significant for the MB treatment ($R^2 = 0.3261$, $P = 0.0262$) but not significant for the C treatment (fig. 3D).

Table 1—For the control (C) and May burn (MB) treatments and at each of 6 soil depth intervals, number of subplots out of 12 subplots with significant linear regressions that predicted soil strength at 16 percent soil water content, and number of these subplots that had a coefficient of determination (R^2) >0.50

	Soil depth (cm)					
Treatment	0 to 2.5	2.5 to 5.0	5.0 to 7.5	7.5 to 10.0	10.0 to 15.0	15.0 to 20.0
	Number of subplots with a significant soil strength–soil water content regression					
C	6	10	11	10	10	9
MB	1	9	10	11	9	7
	Number of subplots with a significant regression with R^2 >0.50					
C	3	8	9	8	8	8
MB	0	5	9	9	5	6

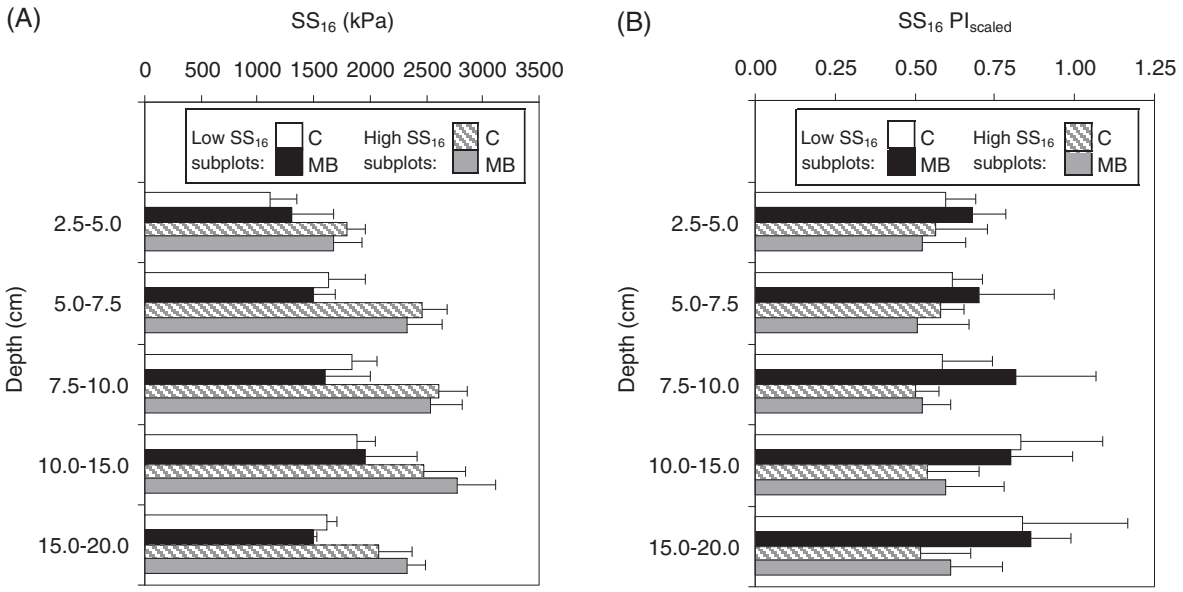

Figure 1—(A) Mean predicted values of soil strength at 16 percent soil water content (SS_{16}) and (B) scaled prediction intervals of SS_{16} in the 2.5- to 20.0-cm depth for subplots with low and high SS_{16} and in response to no vegetation management (C) or biennial fire in May (MB). Values of SS_{16} for the zero- to 2.5-cm depth were excluded because the majority of predictive equations at this depth interval were not significant. Bars represent the standard deviation of the mean.

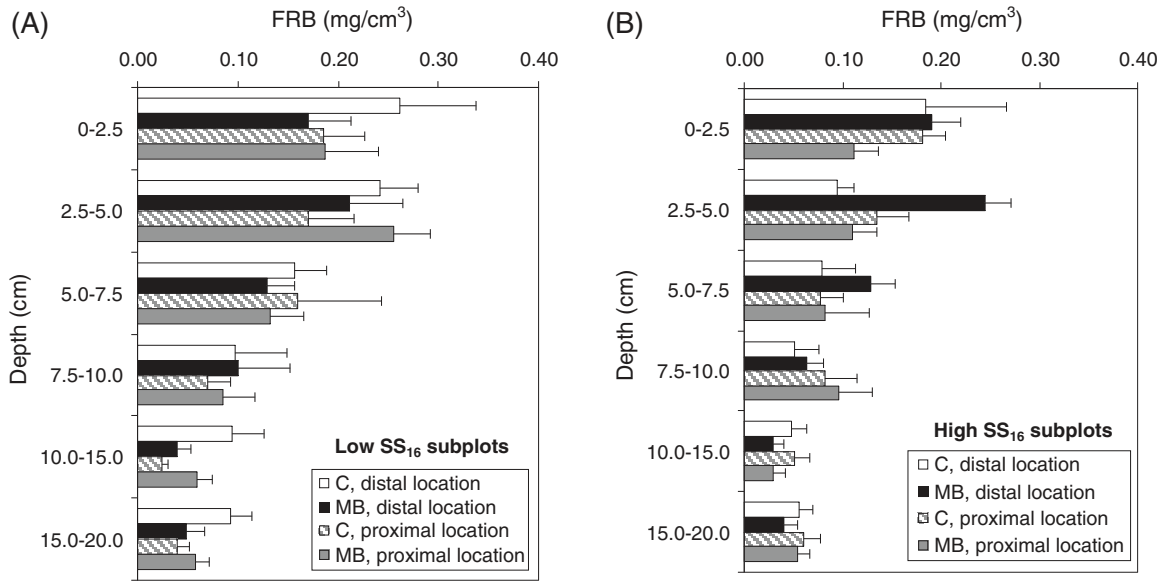

Figure 2—Mean pine fine root biomass (FRB) in the zero- to 20.0-cm depth for subplots with (A) low SS_{16} and (B) high SS_{16}. Sampling was done at locations that were distal or proximal to the measurement trees and in response to no vegetation management (C) or biennial fire in May (MB). Bars represent the standard deviation of the mean.

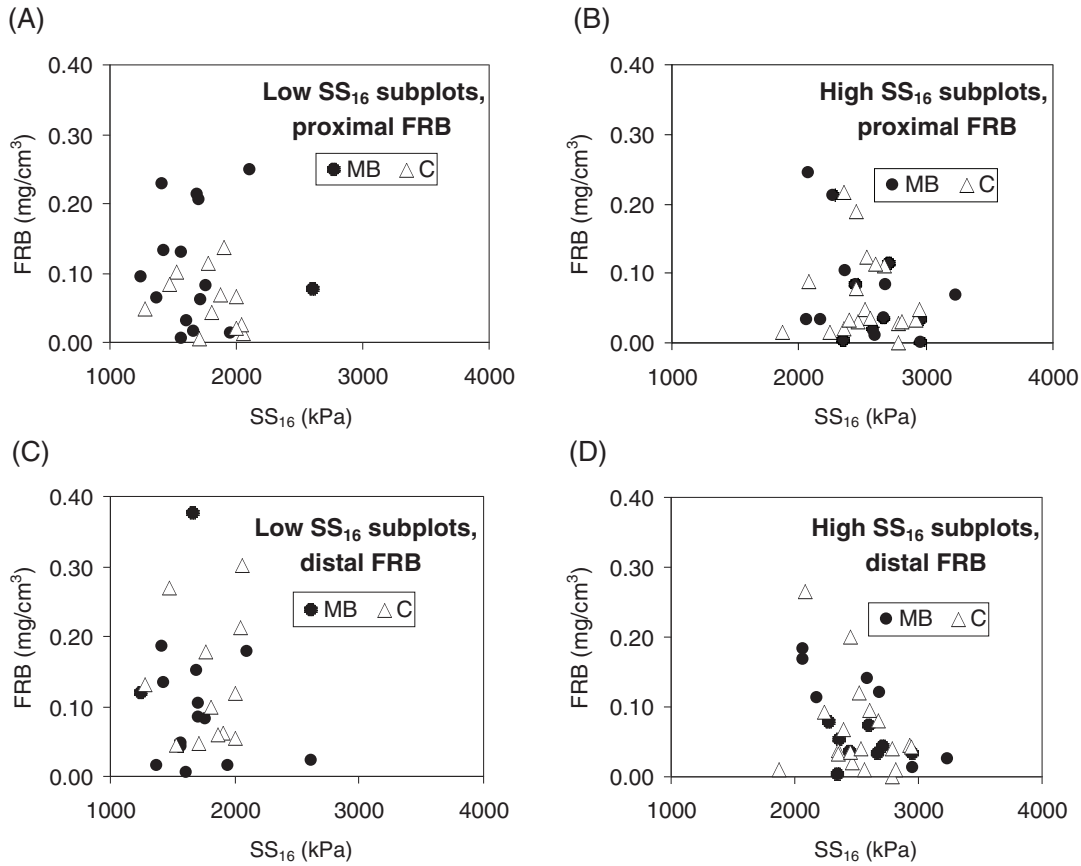

Figure 3—Scatter plots describing the relationship between soil strength at 16 percent soil water content (SS_{16}) and pine fine root biomass (FRB) across the 5.0- to 15.0-cm soil depth and in response to no vegetation management (C) or biennial fire in May (MB) for proximal FRB on the (A) low SS_{16} subplots and the (B) high SS_{16} subplots, and for distal FRB on the (C) low SS_{16} subplots and the (D) high SS_{16} subplots.

DISCUSSION

The study site is representative of typical pine forests on the west Gulf Coastal Plain that seem relatively homogenous with regard to aboveground features such as slope and vegetation. The present results indicate that this appearance may be deceiving when belowground variables such as soil strength are considered. Overall, soil strength at 16 percent SWC averaged 1952±474 kPa which is representative of soil strengths for similar soils (Sword and Tiarks 2002). At this study area, one-half of the 24 subplots were characterized by values of SS_{16} at the 5.0- to 20.0-cm depth that were >2000 kPa. Soil strengths ≥2000 kPa are known to limit pine root elongation (Taylor and others 1966). Therefore, there was notable variation associated with soil strength, and in some locations, the volume of soil accessed by pine roots for water and mineral nutrients may have been considerably smaller than its potential.

The FRB in the zero- to 20-cm depth averaged 0.11 mg/cm^3 which is 63 percent less than that found at an adjacent study site (Sword Sayer and Kuehler 2010). This discrepancy may be attributed to the time of root sampling and the fact that in both studies, live, senescent, and dead but not visibly decomposing root biomass were combined. Higher values were obtained when sampling was done in September and October, while lower values were obtained when sampling was done in November. Silt loam soils in central Louisiana tend to be relatively dry in late summer and early fall. Often, as winter approaches, rainfall saturates the soil which reduces the supply of oxygen to roots (Sword and Tiarks 2002). Natural root mortality in response to dry soil conditions (Caldwell 1977) followed by wet soil conditions may have accelerated root decomposition leading to low FRB in November.

More FRB was found at the 0- to 20-cm depth on the low SS_{16} subplots compared to the high SS_{16} subplots. This suggests that pine root elongation in the upper portion of the soil profile was restricted by either soil strength or other variables that interfere with root growth, e.g., inadequate water or carbohydrate. Siegel-Issem and others (2005) have also found southern pine root growth limitations when soil bulk density and water content interact to increase soil strength. At this point, however, there is no evidence that tree growth

suffered from less FRB in the zero- to 20.0-cm soil depth where SS_{16} was high. Rather, it is likely that tree growth was sustained by roots growing in portions of the soil where soil strength was not root growth limiting. If climate dictates other constraints to root growth such as inadequate plant-available water, however, the sum of all root growth limitations could reduce whole root system function and, therefore, tree growth.

Regardless of C or MB treatment, the high SS_{16} subplots produced less FRB in the 0- to 20-cm depth compared to the low SS_{16} subplots. Correlation between SS_{16} and FRB, however, was only significant on the MB plots when FRB was sampled in distal locations on the high SS_{16} subplots. This suggests that in addition to soil strength, pine root elongation was controlled by other site variables that differed between C and MB plots, proximal and distal locations, and low and high SS_{16} subplots. An obvious difference between burned and unburned stands is the production of understory woody vegetation (Haywood and others 2001). With repeated burning, understory woody vegetation is reduced, leading to less forest floor accumulation (Wells and others 1979) which has the potential to increase surface soil evaporation (Neary and others 1999, Wells and others 1979). Also, the uptake of water near the surface of the soil may be accelerated if repeated fire increases grass and herbaceous cover. Pine trees respond to this situation by increasing the uptake of deeper soil water (Fernández and others 2008). However, it is possible that low surface SWC indirectly inhibited pine root elongation on the high SS_{16} subplots of the MB plots by its inverse relationship with soil strength. Repeated fire may have also affected FRB by reducing resource foraging by pine and nonpine woody roots. Over time, this would lower soil perturbation which could increase soil strength and reduce its variability (Bennie 1996, Fisher and Binkley 2000). These speculations indicate that where soil strength is potentially root growth limiting, information on the composition and distribution of understory vegetation will benefit the evaluation of pine root responses to silvicultural treatments.

This preliminary survey of the relationship between soil strength and longleaf pine FRB suggests that soil strength has the potential to reduce pine root elongation on the west Gulf Coastal Plain. Furthermore, it appears as though repeated prescribed fire interacts with this relationship. These results set the stage for further research designed to evaluate the effect of nonpine woody vegetation on soil strength and its variation, and how pine root responses to soil strength are manifested by aboveground production.

LITERATURE CITED

Bennie, A.T.P. 1996. Growth and mechanical impedance. In: Waisel, Y.; Eshel, A.; Kafkafi, U., eds. Plant roots: the hidden half. 2nd ed. New York: Marcel Dekker: 453–470.

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.

Caldwell, M.W. 1977. Root structure: the considerable cost of belowground function. In: Solbrig, O.T., ed. Topics in plant population biology. New York: Columbia University Press: 408–427.

Da Silva, A.P.; Kay, B.D.; Perfect, E. 1994. Characterization of the least limiting water range of soils. Soil Science Society of America Journal. 58: 1775–1781.

Fernandez, M.E.; Gyenge, J.; Licata, J. [and others]. 2008. Belowground interactions for water between trees and grasses in a temperate semiarid agroforestry system. Agroforestry Systems. 74: 185–197.

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

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: 103–107.

Haywood, J.D.; Harris, F.L.; Grelen, H.E. [and others]. 2001. Vegetative response to 37 years of seasonal burning on a Louisiana longleaf pine site. Southern Journal of Applied Forestry. 25: 122–130.

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.

Neary, D.G.; Klopatek, C.C.; DeBano, L.F. [and others]. 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management. 122: 51–71.

Neter, J.; Wasserman, W. 1974. Applied linear statistical models. Homewood, IL: Richard D Irwin, Inc. 482 p.

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: Conner, K., ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 74–78.

Peet, R.K. 2006. Ecological classification of longleaf pine woodlands. In: Jose, S.; Jokela, E.J.; Miller, D.L., eds. The longleaf pine ecosystem, ecology, silviculture, and restoration. New York: Springer Verlag, Inc: 51–93.

Pritchett, W.L. 1979. Properties and management of forest soils. New York: John Wiley. 500 p.

SAS Institute Inc. 2000. The SAS system. Version 8. Cary, NC: SAS Institute Inc.

Siegel-Issem, C.M.; Burger, R.F.; Powers, F. [and others]. 2005. Seedling root growth as a function of soil density and water content. Soil Science Society of America Journal. 69: 215–226.

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. 2007. Soil physical property response to prescribed fire in two young longleaf pine stands on the western Gulf Coastal Plain. In: Butler, B.W.; Cook, W., comps. Proceedings of the 2nd fire behavior and fuels conference. Proceedings RMRS-P-46CD. Ogden, UT: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 487–495.

Sword Sayer, M.A.; Kuehler, E.A. 2010. Influence of repeated prescribed fire and herbicide application on the root biomass of young longleaf pine. In: Stanturf, John 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: 267–273.

Taylor, H.M.; Robertson, G.M.; Parker, J.J., Jr. 1966. Soil strength-root penetration relations for medium to coarse-textured soil materials. Soil Science. 102: 18–22.

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.

Veihmeyer, F.J. 1929. An improved soil sampling tube. Soil Science. 27: 147–152.

Wells, C.G.; Campbell, R.E.; DeBano, L.F. [and others]. 1979. Effect of fire on soil, a state-of-knowledge review. Gen. Tech. Rep. WO-7. Washington, DC: U.S. Department of Agriculture Forest Service. 34 p.

LONGLEAF PINE REGENERATION FOLLOWING HURRICANE IVAN UTILIZING THE RLGS PLOTS

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Abstract—On September 16, 2004, Hurricane Ivan hit the Alabama coast and severely impacted numerous plots in the U.S. Forest Service's Regional Longleaf Growth Study (RLGS). The Escambia Experimental Forest (EEF) has 201 of the 325 RLGS plots. Nearly one-third of the EEF was impacted. Nine plots with pole-sized trees were entirely lost. Another 54 plots had some type of damage. Following the hurricane, a salvage logging operation was conducted to recover any damaged merchantable timber, which created an opportunity to examine longleaf pine (*Pinus palustris* Mill.) regeneration and its development. Several regeneration-monitoring plots have been installed on the RLGS plots, and an initial inventory of the regeneration has been documented. The location of the monitoring plot, seedling density, and measurements of seedling size have been documented. With this information, longleaf pine regeneration and its development on plots impacted by Hurricane Ivan can be followed.

INTRODUCTION

Natural regeneration of longleaf pine (*Pinus palustris* Mill.) is one of the most important tools natural resource managers have at their disposal to regenerate existing longleaf pine stands in the Southern United States. However, adequate cone crops for natural regeneration typically occur every 5 to 7 years and often longer.

Longleaf pine ecosystems are considered to be in a perilous condition. A report by the U.S. Department of the Interior lists the longleaf pine ecosystem as the second-most threatened ecosystem in the United States (Noss 1989). The original longleaf pine forest was self-perpetuating. It reproduced itself in openings in the overstory where young stands developed. The result was a parklike, uneven-aged forest, composed of many even-aged stands of varying sizes. Wahlenberg (1946) described the original longleaf pine forests as made up mainly of pure, even-aged, irregularly open stands. The even-aged character was the result of relatively infrequent but heavy seedfalls and the ability of reproduction to survive only in openings free of an overstory.

Many of the factors governing the ability of longleaf pine to reproduce are obscure, and the innumerable ecological influences are so interrelated as to make their interpretation difficult. Solutions depend on understanding the prerequisites of the process, the characteristics of seed-bearing trees and longleaf pine seed crops, and the possible causes of failure after seedfall. Predicting seedling performance under varying levels of overstory competition is important for understanding the consequences of silvicultural systems.

One major regeneration problem is irregular seed production. Seed crops considered adequate for regeneration occur at 5- to 7-year intervals, on average, with exceptions. Longleaf pine is generally considered the most intolerant of the southern pines (Baker 1949). It is intolerant of competition from any source especially overstory competition. Survival and growth are closely related to longleaf pine's two unique silvical characteristics—its grass stage and its high resistance to fire. The grass stage usually lasts 4 to 5 years but may range from 2 to 20 years. If competing species are allowed to grow freely, they will completely dominate the site while longleaf seedlings are still in the grass stage. Once this has occurred, a longleaf pine stand can never regain dominance without some type of intervention. Unsatisfactory regeneration in longleaf pine forests may be attributed largely to the lack of management or unwise management. Mismanagement may be the rule rather than the exception, due to ignorance of the unique life history of the species and incomplete knowledge of factors determining the life and death of seedlings.

Research has been conducted on the Escambia Experimental Forest (EEF) since 1947. It is operated by the U.S. Forest Service (Forest Service) in cooperation with the T.R. Miller Mill Company in Brewton, AL. The EEF contains nearly 60 percent of the Regional Longleaf Pine Growth Study (RLGS) plots. From 1964 to 1967, the Forest Service established the RLGS in the Gulf States (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. The RLGS consists of 292 1/5-acre and 13 1/10-acre permanent measurement plots located in central and southern Alabama, southern Mississippi, southwest Georgia, northern Florida, and the sandhills of North Carolina. The plots are inventoried on a 5-year cycle and are thinned at each inventory, as needed, to maintain the assigned density level. Plots cover a range of age classes from 20 to 120 years, five site index classes ranging from 40 to 80 feet at 50 years, and five density classes ranging from 30 to 150 square feet per acre, with a new class recently added of "free to grow" to see what is the maximum density longleaf pine stands can attain prior to extensive mortality setting in. Densities are established and maintained by low thinning. The study accounts for growth change over time by adding a new set of plots in the youngest age class every 10 years. Within this distribution are five time replications of the youngest age class. All five replications are located on

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the EEF. Plots are burned once every 3 years in the dormant season (Kush and others 1987, 1998).

On September 16, 2004, Hurricane Ivan hit the Alabama coast and severely impacted numerous plots in the RLGS. Thirteen plots were entirely destroyed, where 9 of those plots had pole-sized trees >40 years old. Another 54 plots had some type of damage. Following the hurricane, a salvage logging operation was conducted to recover any damaged merchantable timber. Kush and Gilbert (2010) documented and evaluated the damage Hurricane Ivan caused on the RLGS plots. The damage done by Hurricane Ivan and the disturbance from the salvage logging operation allowed for an opportunity to examine longleaf pine regeneration and its development as a result of the openings created by the hurricane. The RLGS database contains a history of burn frequency, plot density, and the location of each tree on a plot. Several regeneration monitoring plots have been installed on the RLGS plots.

METHODS

All 13 of the destroyed RLGS plots were visited in the same dormant season. Monitoring plots were installed on these plots to document an initial inventory of the regeneration following the hurricane and salvage logging operation. The location of the monitoring plot, seedling density, and measurements of seedling size have been documented. With this information, longleaf pine regeneration and its development on plots impacted by Hurricane Ivan can be followed.

The RLGS plots are 0.2 acres in size. All trees on the plot with a diameter at breast height (d.b.h.) of 0.6 inches and greater were stem mapped, and the d.b.h. was recorded. To account for smaller regeneration, a regeneration subsample was taken. Four randomly selected subplots, 56 by 56 inches, were installed in each of the destroyed RLGS plots. All longleaf pine seedlings were counted and labeled as grass-stage seedlings, out-of-grass-stage seedlings, or saplings. Grass-stage seedlings were characterized as seedlings that had not initiated height growth. Seedlings out of the grass stage were characterized as having a root-collar diameter ≥1 inch and having initiated height growth. Saplings were characterized as having a d.b.h. <0.6 inches. In an effort to continue monitoring the visual aspects of the plots, photo points were also established at each plot center, and photos were taken facing each cardinal direction (north, east, south, and west).

RESULTS

The destroyed plots covered the range of basal area classes for the RLGS plots from 30 to 120 square feet per acre. Regeneration was found across all basal area classes, but the amounts and sizes varied. One plot was removed due to the amount of damage caused by the salvage logging operation. A plot in the 60-square-feet-per-acre class, which had been damaged by a tornado prior to Hurricane Ivan, originally had 120 square feet per acre. The tornado created openings or gaps for regeneration, which enabled regeneration to become established on the plot. This was the only plot where an overstory tree survived the hurricane and the salvage logging operation but died before this study was installed.

The plots were summarized by the average number of stems per acre by regeneration type. The average number of stems per acre was 1,708 grass seedlings per acre, 375 out-of-grass-stage seedlings per acre, 41saplings per acre, and 152 trees per acre. The plots were also summarized by the size of the trees recorded. Trees were recorded in the 1-, 2-, 3-, 4-, and 6-inch d.b.h. classes. The average number of trees per acre recorded in each d.b.h. class was 67, 30, 7, 1, and 0.4 trees per acre, respectively.

Since basal area provides a measure of density that relates to the amount of sunlight reaching the forest floor, the plots were also summarized by the maintained basal area classes prior to the two disturbances. The average number of grass-stage seedlings per acre recorded in the 30-, 60-, 90-, and 120-square-feet-per-acre basal area classes was 1,800, 2,750, 250, and zero grass-stage seedlings per acre, respectively. The average number of out-of-grass-stage seedlings per acre by basal area class included 200, 750, zero, and 500 seedlings per acre in the 30-, 60-, 90-, and 120-square-feet-per-acre basal area classes, respectively. Saplings were only recorded for one basal area class. The average number of saplings per acre for the 60-square-feet-per-acre basal area class was 125 saplings per acre. Trees were only recorded on the 30- and 60-square-feet-per-acre basal area classes with an average of 152 and 134 trees per acre, respectively.

Plots were also summarized by the remaining basal area following the hurricane and the salvage logging operation to determine if openings caused by the disturbances seemed to influence regeneration more than the previously maintained basal area classes. The residual basal areas were classified into five classes including zero, 10, 20, 30, and 40 square feet per acre. The average number of grass-stage seedlings per acre was 2,000, zero, 2,000, 3,334, and 500 for each basal area class, respectively. The average number of out-of-grass-stage seedlings per acre was 1,000, 167, 333, 333, and zero for the zero-, 10-, 20-, 30-, and 40-square-feet-per-acre basal area classes, respectively. Saplings were only recorded in the zero-square-feet-per-acre class with an average of 250 saplings per acre. The average number of trees per acre was 405, 32, 75, 55, and zero for the zero-, 10-, 20-, 30-, and 40-square-feet-per-acre basal area classes, respectively.

DISCUSSION

The amount of and type of regeneration varied across the maintained basal area classes and basal area per acre following the hurricane and the salvage logging operation. The RLGS does not account for regeneration. The plots undergo numerous disturbances like harvest operations to maintain basal area classes, natural mortality, weather damage, and prescribed fires. Therefore, it is not possible to determine exactly how old the regeneration is or how many cohorts exist on each plot. However, the basal area classes

maintained in the RLGS provided an interesting way to look at regeneration following the disturbances. These findings show that, on average, grass-stage seedlings were the most abundant type of observed regeneration followed by out-of-grass-stage seedlings, trees, and then saplings. The saplings and trees, which only existed in the understory and midstory of the 30- and 60-square-feet-per-acre plots, had to be present before the hurricane. These densities and regular prescribed fire provided enough sunlight and bare mineral soil for seed to establish and grow. The high densities of 90 and 120 square feet per acre did not contain the large types of regeneration, but they did contain grass-stage and out-of-grass-stage seedlings. The majority of the out-of-grass-stage seedlings were likely grass-stage seedlings before the hurricane and were released after the overstory was damaged or removed by the two disturbances. Looking at the residual basal area classes following the disturbances does show that grass-stage seedlings were present across all of the residual basal area classes except the 10-square-feet-per-acre class. However, there is not a clear pattern showing the plots with no- or low-residual basal areas having the largest amounts of grass-stage seedlings. It is likely that many of the grass-stage seedlings were also present before the disturbances. The amounts and types of regeneration did not consistently show that as the basal area increased, the size and abundance of regeneration decreased or that the inverse is true. This is potentially due to the disturbances that have affected the plots like the 60-square-feet-per-acre plot that had earlier been affected by a tornado. These disturbances change the spatial distribution of the plots creating gaps from trees being removed or fire not carrying evenly across the plot. A more intense spatial analysis including trees surrounding the plots is needed to determine what is driving regeneration patterns.

CONCLUSIONS

The relationship between target basal area classes in combination with residual basal areas following the disturbances seem to be influential to the patterns of regeneration on the plots, but a more complex spatial dynamic exists. Regeneration was not seen in areas with thick patches of inkberry (*Ilex glabra* L.) and/or hardwood regeneration which suggests that gaps have been opened from other disturbances like prescribed fire, natural tree mortality, and harvest operations to maintain basal area classes. This also suggests the potential need to switch to growing season fires which can help control competing vegetation. Monitoring the plots will allow the opportunity to follow the progress of natural regeneration in the destroyed RLGS plots over time. The character of the ecosystem is best maintained with natural regeneration, using the processes that have long maintained longleaf ecosystems over the millennia. No phase of longleaf pine management presents more complex and critical problems than does its reproduction.

ACKNOWLEDGMENTS

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LITERATURE CITED

Baker, F.S. 1949. A revised tolerance table. Journal of Forestry. 47(3): 179–181.

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 structure measures. Athens, GA: University of Georgia. 132 p. Ph.D. dissertation.

Kush, J.S.; Gilbert, J.C. 2010. Impact of Hurricane Ivan on the Regional Longleaf Pine Growth Study: is there a relation to site and conditions? In: Stanturf, John 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, Southeastern Forest Experiment Station: 259–261.

Kush, J.S.; Meldahl, R.S.; Dwyer, S.P.; Farrar, R.M., Jr. 1987. Naturally regenerated longleaf pine growth and yield research. In: Phillips, D.R., comp. Proceedings of the fourth biennial southern silvicultural research conference. Gen. Tech. Rep. SE-42. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 343–344.

Kush, J.S.; Meldahl, R.S.; McMahon, C.K. 1998. Thirty years old—the Regional Longleaf Pine Growth Study. In: Waldrop, T.A., ed. Proceedings of ninth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 113–117.

Noss, R.F. 1989. Longleaf pine and wiregrass: keystone components of an endangered ecosystem. Natural Areas Journal. 9: 211–213.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth, and management. Washington, DC: Charles Lathrop Pack Forestry Foundation. 429 p.

2008 INTERIM GUIDELINES FOR GROWING LONGLEAF PINE SEEDLINGS IN CONTAINER NURSERIES

R. Kasten Dumroese, James P. Barnett, D. Paul Jackson, and Mark J. Hainds[1]

Abstract—Production of container longleaf pine (*Pinus palustris* Mill.) seedlings for reforestation and restoration exceeds that of bare-root production, but information on container production techniques has been slow to develop. Because outplanting success requires quality seedlings, interim guidelines were proposed in 2002 to assist nursery managers and tree planters in developing and using the best stock possible. The guidelines were intended to be updated as new information was generated. During the past 6 years, additional studies have confirmed most provisions of the interim guidelines, except that presence of buds (number and color) as originally described in the guidelines does not appear to be a useful metric. In addition, some new attributes have been added. This report synthesizes that new information and presents revised guidelines.

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.) forests were once a dominate ecosystem across the Southeastern United States, but intense harvesting during the past century reduced this forest type from nearly 36 million ha (90 million acres) to about 800,000 ha (2 million acres). As a result, many species found in longleaf pine-dominated forests have become threatened or endangered (Barnett 2002, Jose and others 2006, Noss and others 1995, Outcalt 2000). Restoration of this forest type has been encouraged by Federal incentive programs (Hainds 2002), especially through afforestation or reforestation using planting. Because survival and growth of container longleaf pine planting stock is often better than bare-root stock after outplanting (Barnett and McGilvray 1997, Boyer 1989, South and others 2005), use of container stock has increased dramatically. For example, in 2008 about 64 million container seedlings were produced compared to about 12 million bare root.[2]

Despite demand for container longleaf pine, very little detailed research exists concerning the production of this relatively new stock type. This information gap led to a major problem: an absence of container seedling standards and subsequent variation in stock quality (Hainds 2004). Although stock quality can be described in the nursery, what really matters is how well it performs on the outplanting site (Landis and Dumroese 2006). On one hand, plants characterized as "poor" in the nursery may perform well in the field if site factors are favorable, for example, completion of proper site preparation, planting technique, weed control, and/or ample precipitation. On the other hand, "high" quality plants may do poorly if those same treatments are done improperly, or if precipitation is below normal. Despite these existing information gaps, Barnett and others (2002a, 2002b) published interim guidelines to help growers identify container types and seedling quality attributes for growing longleaf pine seedlings in containers. These guidelines were generated based on the available completed or ongoing research, experience of growers, and the expertise of regional specialists with the intention that they would be revised as new information became available.

Since 2002, more information has been published, including Dumroese and others (2005), Hainds and Barnett (2006), Jackson (2006), and Jackson and others (2007, 2010); practical experience has also expanded over the past several years. Thus, it is timely to revisit the 2002 standards with an update.

2002 INTERIM GUIDELINES

The 2002 interim guidelines focused on needles, roots and root-collar diameter (RCD), buds, container size, and other important attributes, such as presence of "sondereggers" [*P.* x *sondereggeri* H.H. Chapm., a naturally occurring hybrid of *P. palustris* and *P. taeda* L. (Little 1979)] (Barnett and others 2002a, 2002b). For each attribute, we summarize the "2002 interim guideline" as published in Barnett and others (2002a), describe the "rationale" behind each original guideline, and provide a "2008 update" that synthesizes the new information that corroborates or refines the 2002 guidelines.

Needles

2002 Interim Guideline—If clipped, needles should be 6 to 10 inches (15 to 25 cm) long but not <4 inches (10 cm). If not clipped, needles should be 8 to 12 inches (20 to 30 cm) long. The appearance of many fascicles is preferred, and needles should have a pale-green-to-dark-green color.

Rationale: Barnett (1984) showed that repeated clipping of longleaf needles to maintain a length of 2 inches (5 cm) reduced RCD, shoot weight, and root weight during nursery production. But seedlings given single or multiple clippings to maintain a needle length of 10 inches (25 cm) were similar

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[2] Unpublished data, on file with The Longleaf Alliance, 12130 Dixon Center Road, Andalusia, AL 36420

to their nonclipped cohorts. In addition, survival of seedlings clipped to maintain the 2-inch length was poorer under higher levels of moisture stress than seedlings with longer needles. Barnett (1984) also reported that seedlings clipped once to 10 inches (25 cm), immediately before outplanting under severe moisture stress conditions, survived better than control seedlings and seedlings clipped too frequently. These results are similar to the conclusions of South (1998) who noted that clipping needles of bare-root seedlings improved survival, presumably because of reduced transpiration on sites where seedlings are under significant moisture stress. Clipping needles in the nursery can prevent their lodging and reduce subsequent susceptibility to disease by improving air circulation, reducing humidity levels, and allowing more uniform irrigation. Poor irrigation uniformity leads to overwatering and can increase root disease (Enebak and Carey 2002). Barnett (1989) found that seedlings grown in shade during nursery production were much smaller and suggested that clipping could allow more uniform light exposure (Barnett 1984). Seedlings with fascicles are preferred; Wakeley (1954) and Barnett (1980) reported that seedlings with fascicles perform better after outplanting. A healthy "green" color is indicative of proper nutrient status, rather than the "yellow" (chlorotic) foliage resulting from nutrient deficiencies.

2008 Update—To our knowledge, no new work has been published on clipping. However, we found that needle length of container seedlings is a function of nitrogen fertilizer rate (Jackson 2006, Jackson and others 2007). We also determined that a rate of 2 to 3 mg nitrogen per seedling per week for 20 weeks produced seedlings in Ropak® Multipot #3-96® containers [depth = 4.8 inches (12 cm); volume = 6 cubic inches (98 cm^3); density = 41 per square foot (441/m^2)] with needles within the original interim guidelines without the need for clipping. After outplanting, these seedlings survived and grew well (Jackson 2006, Jackson and others 2007). Seedlings given 4 mg nitrogen per week for 20 weeks had needles that would have required clipping under operational conditions to prevent lodging (we did not clip them, however, in the experiment). No additional benefit in terms of seedling survival or growth was seen for this stock type. It should be noted that many other fertilizer regimes appear to produce longleaf seedlings without the need for clipping (Dumroese and others 2005). It may be, however, that nutrient loading of longleaf pine seedlings in the nursery (Dumroese 2003, Hinesley and Maki 1980) in concert with clipping may improve outplanting performance, particularly because of unpublished work conducted at Auburn University. Researchers there found that clipping longleaf pine seedlings to 8 inches (20 cm) reduced water loss in a greenhouse during the first 4 days after clipping.[3] This short-term effect may be beneficial to outplanting performance.

Roots

2002 Interim Guideline—RCD, measured at the base of the needles, should be one-fourth inch (6.35 mm) or more, and no less than three-sixteenth inch (4.75 mm). Roots should be light brown in color with white root tips, free of disease symptoms, and without circling. Presence of mycorrhizae is encouraged.

Rationale: Because longleaf pine seedlings generally exit the grass stage when their RCDs are about 1 inch (25 mm) (Wahlenberg 1946), obtaining large RCDs in the nursery could shorten the grass stage after outplanting. In addition, larger RCDs are associated with better survival of bareroot stock (White 1981). The minimum value was based on observations that seedlings with < three-sixteenth inch (4.75 mm) diameter grown in Ropak® Multipot #6-45® containers [depth = 4.8 inches (12 cm); volume = 6 cubic inches (98 cm^3); density = 54 per square foot (581/m^2)] were "floppy" and had reduced survival. ["Floppy" seedlings, when held horizontally by the terminal bud, "flopped" over because of insufficient development of roots within the root plug (Hainds and Barnett 2004, 2006).] Light brown roots with white root tips indicate a healthy root system and show potential for new root development. Black roots require close scrutiny because they are likely diseased, particularly if a large portion of the root system is black. Modern, commercially available containers typically used to produce reforestation seedlings have modifications (ribs, slits, chemical coating) to prevent circling. Presence of mycorrhizae indicates a healthy root system but applying inoculant is usually unnecessary because windborne spores typically inoculate seedlings naturally (Barnett and Brissette 1986).

2008 Update—In general, the recommendation for RCDs being greater than three-sixteenth inch (4.75 mm) for typical 6 cubic inches (100 cm^3) seems acceptable. In this stock type, we note that most fertilizer regimes produce seedlings above this threshold (Jackson 2006, Jackson and others 2007). Seedlings below this threshold have reduced survival (Hainds and Barnett 2004, 2006), and it appears that seedlings with increasing RCDs have increasingly better performance in terms of reduced time in the grass stage (Jackson and others 2007, 2010). South and others (2005) report a critical threshold of 5.5 mm; seedlings with less RCD had poorer survival across a variety of sites than those with greater RCD. Recent work shows, however, that RCD cannot be increased indefinitely without a decline in survival and growth—when the ratio of RCD to the diameter of the growing container, the Root Bound Index, was >27 percent, seedling survival was compromised (fig. 1) (South and Mitchell 2006, South and others 2005). Our observation is that this critical threshold may be difficult to achieve in a 20- to 30-week growing cycle for seedlings in Ropak® Multipot #3-96® containers, but as Salonius and others (2002) point out, it could be easily achieved when seedlings are grown too long in the containers, or "held over" in the nursery in anticipation of being sold the following year. Most typical, commercially available containers used for reforestation have design features to prevent root circling. Some containers are treated with copper to prevent root spiraling, which also

[3] Personal communication. David B. South, Professor, Auburn University, School of Forestry and Wildlife Sciences, Alabama Agricultural Experiment Station, Auburn, AL.

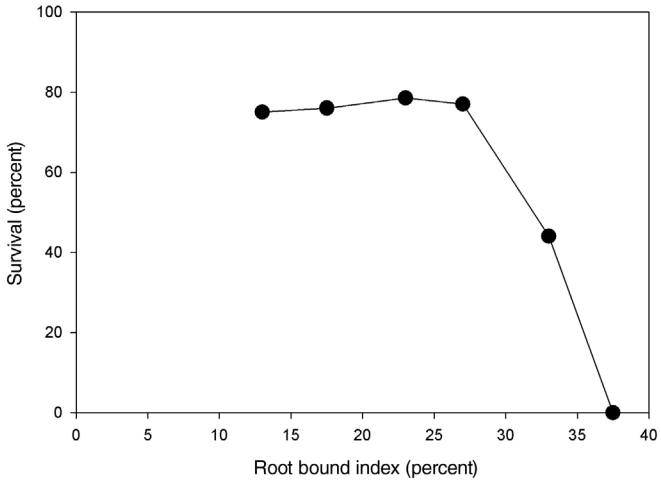

Figure 1—Effect of the root bound index (root-collar diameter/cell diameter) on second-year survival of container longleaf pine seedlings (South and others 2005).

prevents lateral roots from growing downward on the exterior of the plug and forming a "bird cage." This copper treatment is associated with changes in root system morphology, shoot and root biomass (Barnett and McGilvray 2002), and root growth potential (South and others 2005). In general, these seedlings are easier to extract from containers, especially those made of Styrofoam, and fresh copper on container walls decreases the level of potential disease inoculum (Dumroese and others 2002). Copper-coated containers yield seedlings with better, more uniform root distribution higher on the initial root plug, which is believed to improve resistance to windthrow (Burdett 1978, Burdett and others 1986). Neither South and others (2005) nor Sung and others (2010) noted any short-term benefit, in terms of survival or growth, from growing seedlings in copper-treated containers. Tinus and others (2002) determined that exposing longleaf roots to temperatures below 25 °F (–4 °C) caused significant damage. South (2006) reports damage is more severe if that temperature is achieved before seedlings have acclimated to cold temperatures (early winter) or the frost is preceded by warm temperatures that cause deacclimation of seedling tissues to cold.

Buds

2002 Interim Guideline—Buds should be present on 90 percent of the crop. Seedlings outplanted in late October or early November are more likely to have green buds, whereas seedlings outplanted in late December or January are more likely to have brown buds. Brown buds are thought to be more mature, but outplanting should not be delayed to obtain better bud development.

Rationale: Personal observations of quality seedling crops grown during a variety of research projects indicated that seedlings at the end of the growing cycle in late fall had a cessation of needle growth, hardening of tissue, and formation of notable, green, terminal buds, which then became brown during winter.

2008 Update—Early researchers noted that longleaf pine seedlings in the grass stage exhibit a progression of bud types (Pessin 1939, Wahlenberg 1946). Wakeley (1954) noted that bud status during a single growing season changed as terminal buds formed, opened, reformed, and reopened. We have observed development of the apex during several studies and have attempted some quantification. Attempting to use the bud descriptions (pincushion, round, and elongated) of Pessin (1939), Wahlenberg (1946), and Wakeley (1954) during nursery production has been problematic, as nursery stock shows a wide variation in apex characters not necessarily meeting those descriptions. Jackson (2006) found that increasing rates of fertilizer resulted in larger, more robust buds. At deficient nitrogen rates, buds were small and brownish, whereas seedlings given high doses of nitrogen had larger, green buds. In another trial, we observed that frequency of terminal buds varied by month, generally increasing from September through December and then decreasing dramatically in January (fig. 2), whereas in another study more than 90 percent of the crop still had firm terminal buds in January. Larson (2002) points out that dormant buds may be difficult to see. Therefore, additional quantification, and perhaps a new framework for describing/measuring bud development during nursery culture, would help identify if, and what, the effect of differing bud/apex condition on longleaf pine seedling quality might be. Because we have outplanted groups of longleaf pine seedlings with wide variation in the presence of terminal buds [ranging from 20 (Jackson and others 2007) to 100 percent (fig. 2)] and survival and growth have been similar, it appears that the bud criteria in the 2002 guidelines is not useful.

Container Size

2002 Interim Guideline—Container diameter should be no less than an inch (25 mm) with 1.5 inches (38 mm) or greater

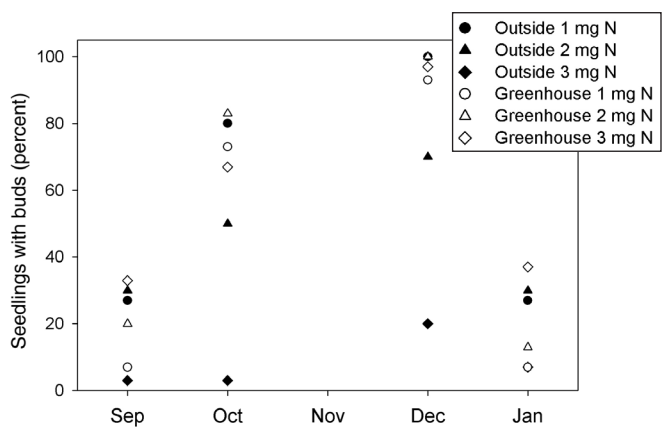

Figure 2—Bud occurrence from September through January for longleaf pine seedlings in a recent fertilizer trial completed by the authors. Although no pattern was observed between seedlings grown in a greenhouse or outside, or among three levels of nitrogen fertilizer, pooled data showed that buds formed from September, with most of the crop having discernable terminal buds in December, followed by an opening of terminal buds in January.

desired. Container depth should be no less than 3.5 inches (9 cm) with 4.5 inches (11.5 cm) or more preferred. Container volume should be no less than 5.5 cubic inches (90 cm^3) with 6 cubic inches (100 cm^3) or more recommended.

Rationale: The guidelines were based on observations from a variety of studies (Amidon and others 1982; Barnett 1974, 1984, 1988, 1991; Barnett and McGilvray 1997).

2008 Update—Since the interim guidelines were published, most of our work has focused on seedlings grown in Ropak® Multipot #3-96® (Jackson 2006; Jackson and others 2007; Jackson and others 2010) or Ropak® Multipot #6-45® (Dumroese and others 2005) containers (described above). Seedlings grown in Ropak® Multipot #3-96® containers have been evaluated up to 3 years in the field; preliminary data shows excellent survival and growth (Jackson and others 2007; Jackson and others, in press). South and others (2005) evaluated six different containers differing in depth from 2.6 to 6 inches (6.5 to 15 cm), in volume from 4 to 6 cubic inches (60 to 120 cm^3), and in container material, outplanted on four field sites. They concluded that container type (Styrofoam, hard plastic, or mesh) may not affect survival on easy-to-regenerate sites, but mesh-type containers (such as Jiffy pellets) performed poorer on harsher sites than Styrofoam and hard plastic containers (the containers had characteristics consistent with the original guidelines). Sung and others (2010) found reduced survival, height growth, and exit from the grass stage for seedlings grown in small volume [4-cubic-inch (54-cm^3)] containers compared to larger cohorts. A study examining a wider range of container sizes [4 to 20 cubic inches (60 to 340 cm^3)] was outplanted on the U.S. Forest Service Palustris Experimental Forest (Rapides Parish, LA) in December 2008.

Other Important Attributes

2002 Interim Guideline—Root plugs should remain intact (no loss of medium) when extracted and during handling, and they should always be moist. Seedlings should lack competing weeds and insect pests. The nursery manager and the buyer should agree whether to cull sonderegger seedlings.

Rationale: Firm root plugs indicate good root development, and seedlings with firm plugs and appropriate RCD for the container diameter are not "floppy" as described in the "roots" section. Furthermore, firm plugs facilitate handling in the nursery and outplanting because they do not fall apart, and losing a portion of the root plug during the process of extraction through outplanting was associated with a decrease in survival and subsequent growth in a conifer species (Tinus 1974). Moisture held in the growing medium prevents root desiccation. A seedling sharing its container with a competing weed has less access to nutrients and water, resulting in reduced growth (Pessin and Chapman 1944). Seedlings that begin height growth during nursery production are usually sonderegger pines. These seedlings produce poorly formed trees in plantations and are less desirable than longleaf pine.

2008 Update—Many growers irrigate their seedlings just prior to extraction (Dumroese and Barnett 2004). Seedlings may be hot planted (no or very limited storage) or cooler stored for a week to a few months (Dumroese and Barnett 2004). Regardless, having moist plugs when shipped to the field is important. This may be especially true for seedlings outplanted during the April through October planting window because these seedlings likely have more exposure to greater vapor pressure deficits than seedlings hot planted, or outplanted after cooler storage, during the relatively mild "winter" season. Luoranen and others (2004) found that mortality of silver birch (*Betula pendula* Roth) increased with decreasing plug moisture content; rate of mortality with decreasing plug moisture was greatest on dry sites. More detailed observations by Hainds and Barnett (2006) suggest that seedlings with as much as 4 inches (10 cm) of height

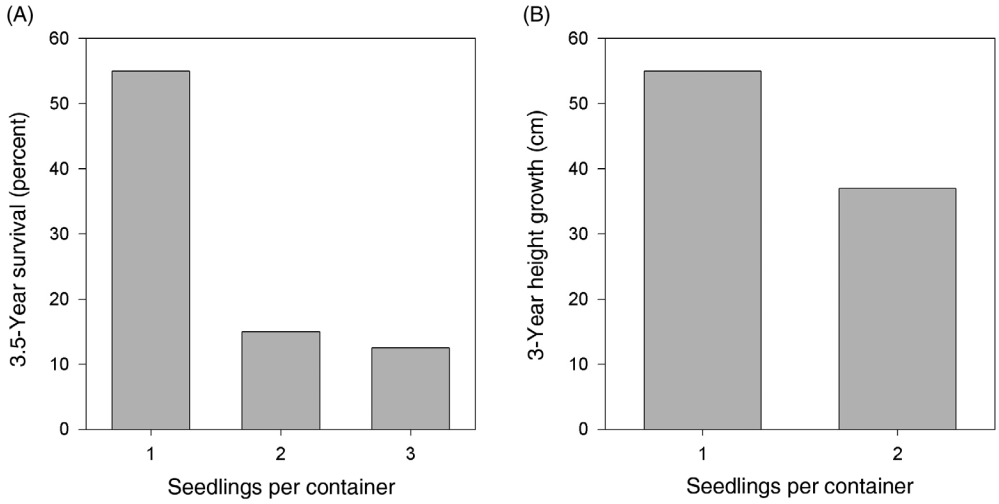

Figure 3—(A) Survival of longleaf pine seedlings decreases when multiple seedlings exist within a single container (Barnett and Brissette 1986) and (B) height growth of seedlings at the Samson site in Alabama (see Hainds and Barnett 2006 for a more details). Note: This figure was presented incorrectly in Hainds and Barnett (2006).

Table 1—2008 interim guidelines for nursery production of longleaf pine seedlings

Needles	Needles should be 6 to 12 inches (15 to 30 cm) long and not less than 4 inches (10 cm). Needles should have a "medium-to-dark" green color. Avoid yellow or brown seedlings.
Roots	Root-collar diameter (RCD), measured at the base of the needles, should be no less than 3/16 inch (4.75 mm). Larger RCDs are encouraged as long as the ratio of seedling RCD to container diameter is <27 percent to avoid root binding. Roots should be light brown in color with white root tips, free of disease symptoms, and without circling. Cambium at or near the root collar should be whitish or greenish, never orange or brown. Plugs should be firm and moist and stay intact during extraction and outplanting. Avoid "floppy" seedlings—these seedlings, when held horizontally by the terminal bud, bend or flop, unable to maintain a straight horizontal alignment. Seedlings with a very large callus at the tip of the air-pruned taproot might not form a strong taproot.
Buds	May or may not be present.
Container size	Diameter ≥1 inch (25 mm) with 1.5 inches (38 mm) or greater desired. Depth ≥3.5 inches (9 cm) with 4.5 inches (11.5) or more preferred. Volume ≥5.5 cubic inches (90 cm^3) with 6 cubic inches (100 cm^3) or more recommended.
Other important attributes	Seedlings should be free of weeds and insects. Avoid multiple seedlings within a single container. Sonderegger pines retained or removed pending decision by grower and buyer in agreement.

growth in the nursery may not necessarily be sonderegger pines. This may complicate identification of hybrid seedlings in the nursery; as always, the best solution is for the grower and the buyer to communicate about this beforehand.

Not discussed in the original guidelines were "double seedlings," two seedlings growing in a single container. During nursery production, a "single" seedling can have twice the dry weight of a "double" seedling (Barnett and Brissette 1986), which affects outplanting performance. After outplanting, Barnett and Brissette (1986) showed that survival was greatly reduced when two or three seedlings occupied the same container (fig. 3A), and Hainds and Barnett (2006) report that height growth was also diminished (fig. 3B).

SUMMARY

Results from recent studies confirm that most of the recommendations made when the 2002 interim guidelines were developed are still sound (table 1). The main exception is related to the presence of terminal buds and its effect on outplanting performance. Additional information regarding "floppy" seedlings, double seedlings, and classification of sonderegger pines has also been included.

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LITERATURE CITED

Amidon, T.E.; Barnett, J.P.; Gallagher, H.P.; McGilvray, J.M. 1982. A field test of containerized seedlings under drought conditions. In: Guldin, R.W.; Barnett, J.P., eds. Proceedings of the southern containerized forest tree seedling conference. Gen. Tech. Rep. SO-37. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 139–144.

Barnett, J.P. 1974. Tube lengths, site treatments, and seedling ages affect survival and growth of containerized southern pine seedlings. Res. Note SO-174. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 3 p.

Barnett, J.P. 1980. Density and age affect performance of containerized loblolly pine seedlings. Res. Note SO-256. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 5 p.

Barnett, J.P. 1984. Top pruning and needle clipping of container-grown southern pine seedlings. In: Lantz, C., comp. Proceedings: southern nursery conferences. Atlanta: U.S. Department of Agriculture Forest Service, State and Private Forestry, Southern Region: 39–45.

Barnett, J.P. 1989. Site preparation, containers, and soil types affect field performance of loblolly and longleaf pine seedlings. In: Proceedings, 5th biennial southern silvicultural research conference. Gen. Tech. Rep. SO-74. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 155–158.

Barnett, J.P. 1989. Shading reduces growth of longleaf and loblolly pine seedlings in containers. Tree Planters' Notes. 40(1): 23–26.

Barnett, J.P. 1991. Effects of morphological grade on field performance of container-grown southern pine seedings.

In: Coleman, S.S.; Neary, D.G., eds. In: Proceedings, 5th biennial southern silvicultural research conference. Gen. Tech. Rep. SO-74. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 94–99.

Barnett, J.P. 2002. Longleaf pine: why plant it? Why use containers? In: Barnett, J.P.; Dumroese, R.K.; Moorhead, D.J., eds. Proceedings of workshops on growing longleaf pine in containers—1999 and 2001. Gen. Tech. Rep. SRS-56. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 5–7.

Barnett, J.P.; Brissette, J.C. 1986. Producing southern pine seedlings in containers. Gen. Tech. Rep. SO-59. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 71 p.

Barnett, J.P.; Hainds, M.J.; Hernandez, G.A. 2002a. Interim guidelines for growing longleaf seedlings in 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: 27–29.

Barnett, J.P.; Hainds, M.J.; Hernandez, G.A. 2002b. Interim guidelines for growing longleaf seedlings in containers [Leaflet]. Gen. Tech. Rep. SRS-60. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 3 p.

Barnett, J.P.; McGilvray, J.M. 1997. Practical guidelines for producing longleaf pine seedlings in containers. Gen. Tech. Rep. SRS-14. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 28 p.

Barnett, J.P.; McGilvray, J.M. 2002. Copper-treated containers influence root development of longleaf pine seedlings. 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: 24–26.

Boyer, W.D. 1989. Response of planted longleaf pine bare-root and container stock to site preparation and release: fifth year results. In: Proceedings, fifth biennial southern silvicultural research conference. Gen. Tech. Rep. SO-74. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 165–168.

Burdett, A.N. 1978. Control of root morphogenesis for improved mechanical stability in container-grown lodgepole pine. Canadian Journal of Forest Research. 8: 483–486.

Burdett, A.N.; Coates, H.; Eremko, R.; Martin P.A.F. 1986. Toppling in British Columbia's lodgepole pine plantations: significance, cause and prevention. Forestry Chronicle. 62: 433–439.

Dumroese, R.K. 2003. Hardening fertilization and nutrient loading of conifer seedlings. In: Riley, L.E.; Dumroese, R.K.; Landis, T.D., tech. coords. National proceedings, forest and conservation nursery associations—2002. Proceedings RMRS-P-28. Ogden, UT: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 31–36.

Dumroese, R.K.; Barnett, J.P. 2004. Container seedling handling and storage in the Southeastern States. In: Riley, L.E.; Dumroese, R.K.; Landis, T.D., tech. coords. National proceedings, forest and conservation nursery associations—2003. Proceedings RMRS-P-33. Ogden, UT: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 22–25.

Dumroese, R.K.; James, R.L.; Wenny, D.L. 2002. Hot water and copper coatings in reused containers decrease inoculum of *Fusarium* and *Cylindrocarpon* and increase Douglas-fir seedling growth. HortScience. 37: 943–947.

Dumroese, R.K.; Parkhurst, J.; Barnett, J.P. 2005. Controlled release fertilizer improves quality of container longleaf pine seedlings. In: Dumroese, R.K.; Riley, L.E.; Landis, T.D., tech. coords. National proceedings, forest and conservation nursery associations—2004. Proceedings RMRS-P-35. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 3–8.

Enebak, S.; Carey, B. 2002. Pest control for container-grown longleaf pine. 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: 43–46.

Hainds, M.J. 2002. Longleaf seedling trends. 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: 3–4.

Hainds, M.J. 2004. Establishing longleaf pine seedlings on agricultural fields and pastures. 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: 309–313.

Hainds, M.J.; Barnett, J.P. 2004. Container grown longleaf pine seedling quality. 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: 319–320.

Hainds, M.J.; Barnett, J.P. 2006. Container grown longleaf pine seedling quality. 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: 102–104.

Hinesley, L.E.; Maki, T.E. 1980. Fall fertilization helps longleaf pine nursery stock. Southern Journal of Applied Forestry. 4: 132–135.

Jackson, D.P. 2006. Relating morphology, nutrition, and bud development to liquid fertilizer application of containerized longleaf pine. Ruston, LA: Louisiana Tech University, College of Applied and Natural Resources. 62 p. M.S. thesis.

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 morphological and outplanting success of container longleaf pine seedlings. In: Stanturf, John A., ed. In: 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.

Jose, S.; Jokela, E.J.; Miller, D.L., eds. 2006. The longleaf pine ecosystem-ecology, silviculture, and restoration. New York: Springer. 438 p.

Landis, T.D.; Dumroese, R.K. 2006. Applying the target plant concept to nursery stock quality. In: MacLennan, L.; Fennessy, J., eds. Plant quality: a key to success in forest establishment. Proceedings of the COFORD conference. Dublin, Ireland: National Council for Forest Research and Development: 1–10.

Larson, D.R. 2002. Field planting containerized longleaf pine seedlings. 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: 62–63.

Little, E.L., Jr. 1979. Checklist of United States trees (native and naturalized). Agric. Handb. 541. Washington, DC: U.S. Department of Agriculture. 375 p.

Luoranen, J.; Rikala, R.; Smolander, H. 2004. Summer planting of hot-lifted silver birch container seedlings. In: Ciccarese, L.; Lucci, S.; Mattsson, A., eds. Nursery production and stand establishment of broadleaves to promote sustainable forest management. Rome, Italy: APAT (Italy's Agency for the Protection of the Environment and for Technical Services): 207–218. http://www.iufro.org/publications/proceedings/. [Date accessed: January 23, 2009].

Noss, R.F.; LaRoe, T.E., III; Scott, J.M. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Biol. Rep. 128. Washington, DC: U.S. Department of the Interior, National Biological Service. 58 p.

Outcalt, K.W. 2000. The longleaf pine ecosystem of the South. Native Plants Journal. 1: 42–44, 47–53.

Pessin, L.J. 1939. Density of stocking and character of ground cover as factors in longleaf pine reproduction. Journal of Forestry. 37: 255–258.

Pessin, L.J.; Chapman, R.A. 1944. The effect of living grass on the growth of longleaf pine seedlings in pots. Ecology. 25: 85–90.

Salonius, P.; Hallett, R.; Beaton, K.; French, C. 2002. Extended nursery rearing compromises field performance of container-reared conifer seedlings. Inf. Rep. M-X-214E. Fredericton, NB: Canadian Forest Service. 20 p.

South, D.B. 1998. Needle-clipping longleaf pine and top-pruning loblolly pine in bareroot nurseries. Southern Journal of Applied Forestry. 22: 235–240.

South, D.B. 2006. Freeze injury to southern pine seedlings. 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: 441–447.

South, D.B.; Harris, S.W.; Barnett, J.P. [and others]. 2005. Effect of container type and seedling size on survival and early height growth of *Pinus palustris* seedlings in Alabama, U.S.A. Forest Ecology and Management. 204: 385–398.

South, D.B.; Mitchell, R.G. 2006. A root-bound index for evaluating planting stock quality of container-grown pines. Southern African Forestry Journal. 207: 47–54.

Sung, S.S-J.; 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.

Tinus, R.W. 1974. Characteristics of seedlings with high survival potential. In: Proceedings of the North American containerized forest tree seedling symposium. Publ. 68. [Place of publication unknown]: Great Plains Agricultural Council: 276–282.

Tinus, R.W.; Sword, M.A.; Barnett, J.P. 2002. Prevention of cold damage to container-grown longleaf pine roots. 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: 35–42.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth and management. Washington, DC: Charles Lathrop Pack Foundation. 429 p. In cooperation with: U.S. Department of Agriculture Forest Service.

Wakeley, P.C. 1954. Planting the southern pines. Agric. Monogr. 18. Washington, DC: U.S. Department of Agriculture. 233 p.

White, J.B. 1981. The influence of seedling size and length of storage on longleaf pine survival. Tree Planters' Notes. 32(4): 3–4.

DEVELOPMENTAL DYNAMICS OF LONGLEAF PINE SEEDLING FLUSHES AND NEEDLES

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Abstract—Longleaf pine (*Pinus palustris* Mill.) seedlings were grown for 27 weeks in containers of three cavity sizes and two cavity types (with and without copper coating) and then outplanted in central Louisiana in November 2004. Three seedlings from each plot were assessed repeatedly for shoot flush and needle development in 2007 and 2008. Cavity type had no effect on seedling size or number of flushes. Cavity size did not affect number of flushes or needle length. However, seedlings grown in large cavities were taller than seedlings from medium and small cavities. Within each cavity size class, the first flush formed was the longest, and flushes formed thereafter were of similar lengths. Needles from flushes formed later in the year were shorter than those from the earlier flushes. Except for the first flush, it took needles twofold to threefold more time to complete elongation than the flushes.

INTRODUCTION

Disappearance of about 96 percent of the pre-European settlement longleaf pine (*Pinus palustris* Mill.) ecosystems in the South has been caused by extensive harvest of longleaf pine for timber and naval store products between late 1800s and early 1900s, conversion of lands supporting longleaf pine to agriculture farms or other fast-growing pine species, and exclusion of fire from the landscape (Brockway and Outcalt 1998, Landers and others 1995, Outcalt 2000). For the last two decades, many public, industrial, and private land managers and owners have been actively restoring longleaf ecosystems in the Southern United States (Barnett 2002, Boyer 1989, Landers and others 1995). In most artificial longleaf regeneration efforts, container-grown seedlings usually have had a higher survival rate than bare-root stock (South and others 2005 and references cited therein). However, one noted drawback of using container-grown stock for planting is that the established trees have experienced windthrow during strong wind events (South and others 2001). One of the attempted improvements in the morphological quality of container stock root systems was to coat the inside of the cavity with copper (Cu). Slow release of low concentration Cu stops seedling lateral roots from elongating once they reach the cavity wall (Ruehle 1985). In a root growth potential test, longleaf pine seedlings grown in Cu-coated cavities produced more new roots than those grown in non-Cu containers or bare-root seedlings (South and others 2005). Lodgepole pine (*P. contorta* Douglas ex. Louden) grown in Cu-coated cavities had fewer leaning seedlings 3 years after planting than those from cavities without a Cu coating (Krasowski 2003).

A study comparing the short- and long-term effects of different container cavity sizes and types on longleaf pine seedling growth, field performance, and tree stability was implemented in 2004 in central Louisiana. This report will focus on the effects of container cavity size and type on the seasonal developmental dynamics of individual flushes and their needles in 2007 and 2008 from that study.

MATERIALS AND METHODS

Seedling Culture and Stand Establishment

Seedlings were from a long-term study where the effects of container cavity size and cavity coating type on longleaf pine seedling growth, physiology, and root system architecture during greenhouse culture and subsequent field planting were being investigated. Details of the seedling culture and the plantation establishment were presented by Sung and others (2010). Briefly, longleaf pine seeds from a Florida seed orchard were sown in containers in April 2004. There were six container treatments—three cavity sizes and two cavity coating types. Cavity volume for the small (S), medium (M) and large (L) cavity sizes were 54, 93, and 170 ml, respectively. Styroblock® and Copperblock® containers (Beaver Plastics Ltd, Edmonton, Alberta, Canada) of the above mentioned cavity sizes were used for no coating (R) and Cu coating treatments, respectively. Cu oxychloride was the active ingredient in the coating. Protocols for growing longleaf pine by Barnett and McGilvray (2000) were adapted for this study with some modifications (Sung and others, 2010).

The field study 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 Paleudults). Mima mounds of Malbis fine sandy loam (fine loamy, siliceous, subactive, thermic, Plinthic Paleudults) are scattered across the study area. The field study is a randomized complete block factorial design with four replications. Blocking was by soil drainage. Twenty-four treatment plots of 0.0576 ha (24 by 24 m) each were established. Seedlings grown in the six cavity treatments (R-S, Cu-S, R-M, Cu-M, R-L, and Cu-L) were randomly assigned to a plot in each block. In early November 2004, 27-week-old container-grown longleaf pine seedlings were lifted and planted on the same day. Seedlings were planted at 2- by 2-m spacing. Treatment plots are 12 rows of 12 trees. All

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plots were prescribed burned in February 2006 (15 months postplanting) as part of the routine management of the site.

Field Measurements

In July 2006 (20 months postplanting), 3 seedlings from each of the 24 plots were randomly selected and tagged for repeated measurements. Flush development in these 72 seedlings was monitored in 2007 and 2008. The bud for the first flush of any given year was formed during the previous year. The starting day for the first flush was the same for all seedlings and was set at March 8 and February 8 when the field measurements began for 2007 and 2008, respectively. Flush length was measured periodically from mid-March through mid-December. In 2007, needle elongation was measured on needles within 5 cm above the nonneedle growing region at flush base. In 2008, specific needles were marked with lightweight paper clips so that the same needles were measured each time. Periodically, lengths of needles near the marked needles were measured to verify that the weight of the paper clip did not interfere with needle elongation. Needle development began with the protrusion of needles from their white fascicle sheaths.

Statistical Analysis

The randomized complete block analysis was used for all study variables that were associated with the entire seedling such as tree heights and number of flushes per seedling. The study was extended to a split-plot design by considering flush order (1 to 6) within each seedling as the split-plot factor. Analyses on variables associated with flushes or needles were analyzed according to this split-plot design. PROC MIXED (SAS Institute Inc. 2004) was used for all analyses and pairwise comparisons were performed using least squares means with a Bonferroni adjusted experimentwise significance level of 0.05. The logistic function was used to model the developmental pattern of flush length and needle length on an individual seedling basis (Sung and others 2004). Because of the less frequent monitoring of the shoot flush and needle development in 2007 than in 2008, only the 2008 shoot flush and needle lengths were modeled. The logistic equation was defined as:

$$\text{Length} = a / (1 + e^{b+c\text{Day}})$$

where

Length = flush or needle length (cm)

Day = days since the observation of a new bud or since the protrusion of needles from the fascicle sheath

a, b, c = parameters of the logistic function

Nonlinear regression was used to estimate the parameters using PROC NLIN (SAS Institute Inc. 2004). The instantaneous rate of flush or needle elongation at a given day was obtained by determining the slope of the specific logistic equation evaluated at that day:

$$\text{Slope} = (-ace^{b+c\text{Day}}) / (1 + e^{b+c\text{Day}})^2$$

The inflection point was where the instantaneous rate of flush or needle elongation reached its maximum and began to slow down:

$$\text{Inflection point day} = -b/c$$

RESULTS AND DISCUSSION

The flush development sequence of longleaf pine seedlings observed in this study can be classified into the following seven stages. Stage 1 is when a bud becomes visible to the unaided eye and still has tight scales (cataphylls) either white or light brown in color. At stage 2, the bud scales become loose. At stage 3, the lower portions of some bud scales turn green. At stage 4, intact white fascicle sheaths are visible. These fascicle sheaths usually start appearing at the lower portion of a flush. In some flushes, however, the fascicle sheaths appear from other portions of a developing flush, and their appearance is not necessarily synchronized around the circumference of a flush. At stage 5, green needle tips protrude out of the fascicle sheaths. At stage 6, the flush axis which has been elongating since stage 2 completes elongation when the flush elongation rate slows down to <0.5 cm over a 2-week period. At stage 7, needles complete elongation (that is, <1-cm increases over a 2-week period), and thus the entire flush development is considered complete for the purposes of this study.

Stage 1 was recorded as the beginning of a new flush except for the first flush. Buds that developed into the first flushes for any given year were formed in the previous year, staying dormant until the following spring. In most cases, stage 1 of the subsequent flush proceeded shortly before the end of stage 6 of the currently elongating flush. At the latter part of stage 6, the tip of the elongating flush became a tight bud which was recorded as the beginning (stage 1) for the subsequent flush. There was a narrow region below this newly formed bud that did not have needles. This bare region would later extend to 0.5 to 2.5 cm in length. Thus, it was easy to count how many flushes a longleaf pine seedling produces within a year. In recent yet fully extended needles, however, the chlorophyll contents and photosynthetic rates have yet to reach the highest levels.[2] Thus, these needles are not yet physiologically mature at the end of stage 7. In first-year northern red oak (*Quercus rubra* L.) seedlings, the fully expanded median leaf in the third flush did not photosynthesize as much as the fully expanded median leaves in the first two flushes at the end of flush development (Hanson and others 1986). These authors attributed it to the fact that physiological and anatomical development of oak leaves do not proceed at the same rate in all flushes.

Number of flushes in 2007 and 2008 were not affected by cavity type or size (data not shown). All seedlings grew at least three flushes in 2007 and 2008. In 2007, 92, 71, and 25 percent of seedlings grew at least four, five, and six flushes, respectively. In 2008, 97, 79, and 32 percent of seedlings grew at least four, five, and six flushes, respectively. Since less

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than one-third of seedlings assessed in this study produced six flushes, longleaf pine, like *Quercus* spp. (Hanson and others 1986), displays a semideterminate, recurrently flushing pattern of shoot growth. Sheffield and others (2003) reported that mature longleaf pine trees in south Georgia exhibited a determinant pattern (one flush per year) of shoot flush and needle growth in some years and an indeterminate pattern (multiple flushes per year) in other years.

In 2008, cavity type did not affect flush length, needle length, flush elongation duration, needle elongation duration, flush inflection point day and its slope, and needle inflection point day and its slope (table 1). Nor did cavity type affect flush length in 2007 (data not shown). Cavity size affected flush length and inflection day slope significantly in 2008 (table 1). Flush order had the most significant effects on the assessed parameters in 2007 (data not shown) and 2008 (table 1). Cavity size had significant effects on seedling heights in the first 4 years after planting. Four years after planting, the L seedlings were 29 and 55 percent taller than the M and the S seedlings (fig. 1). There were no height differences between the M and the S seedlings.

When lengths of individual flushes were analyzed, L seedlings always had longer flushes than those from the S seedlings for each of the first four flushes formed in 2007 (table 2). Within each cavity size class, the first flush of 2007 was always the longest among all the flushes formed. Flushes formed after the first one were similar in length. In 2008, there were no interactions between flush and cavity size effects on flush lengths. Mean flush lengths for the L seedlings were significantly greater than that of the S seedlings (table 2). The first flush of 2008 was the longest among all flushes (table 2, fig. 2). The fifth and sixth flushes which were formed later in the year were the shortest. This is quite different from the flush size reported for oaks. In the first-year seedlings of several oak species, all flushes except for the last flush of the season were longer than their proceeding flushes (Sung and others 2004).

Needle lengths were affected by cavity type and size in 2008 through their significant interaction. Needles from the earlier formed flushes in a year, such as the first three flushes, were significantly longer than those from the later flushes, such as the fifth and the sixth flushes (table 3, fig. 2). When needles of the fifth and sixth flushes started developing, it was late in the growing season (see below) and these needles did not have as many days for elongation before the short day length and cool temperatures stopped them from elongating. These shorter needles did not resume elongation the following spring. Cavity type affected needle lengths in 2007, and there were three- way interactions among type, size, and flush (data

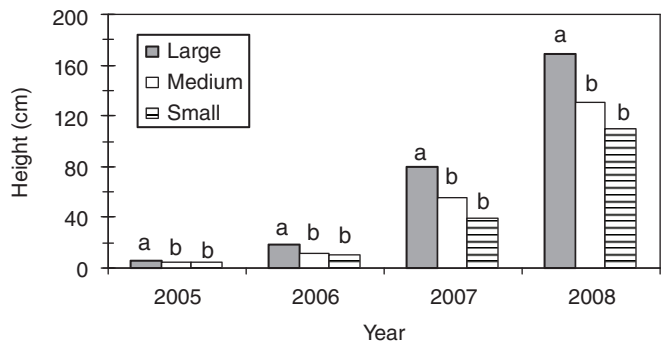

Figure 1—Effects of container cavity size on longleaf pine seedling height growth in the field. Seedlings were grown in containers for 27 weeks in a greenhouse and planted in central Louisiana in November 2004. Least square means with the same letter for each year were not significantly different at the Bonferroni adjusted 0.05 level.

Table 1—Probabilities of a greater *F*-value for the fourth-year longleaf pine seedling flush length, flush elongation duration, flush logistics inflection day, flush slope at inflection, needle length, needle elongation duration, needle logistics inflection day, and needle slope at inflection in response to container cavity size (small, medium, and large), coating type (with and without copper), and flush order in 2008

Source of variation	FL	FED	FI	FIS	NL	NED	NI	NIS
Type	0.6218	0.6674	0.6435	0.4250	0.3553	0.9012	0.3420	0.9051
Size	0.0196	0.3226	0.9303	0.0246	0.4219	0.9261	0.6288	0.0814
T × S	0.2062	0.7281	0.5324	0.0629	0.2039	0.2223	0.0748	0.0577
Flush	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
T × F	0.9370	0.9681	0.7522	0.8657	0.5355	0.9866	0.6352	0.7441
S × F	0.1370	0.1908	0.8796	0.2773	0.0329	0.4490	0.1280	0.4993
T × S × F	0.8709	0.3888	0.0129	0.5601	0.5696	0.9445	0.6760	0.5048

FL = flush length; FED = flush elongation duration; FI = flush logistics inflection day; FIS = flush slope at inflection; NL = needle length; NED = needle elongation duration; NI = needle logistics inflection day; NIS = needle slope at inflection.

Table 2—Lengths of the third-year (2007) and the fourth-year (2008) longleaf pine seedling flushes. The third year had a significant size x flush interaction while the fourth year did not.

Container size	First	Second	Third	Fourth	Fifth	Sixth
	---------- 2007 individual flush length (*cm*) ----------					
Large	23.8[a] a A	9.2 a B	10.0 a B	8.8 a B	8.3 a B	6.7 a B
Medium	15.1 b A	6.6 ab B	7.2 ab B	6.6 ab B	5.5 a B	5.3 a B
Small	10.4 c A	4.1 b B	4.6 b B	5.1 b B	4.5 a B	3.4 a B
	---------- 2008 all flush length (*cm*) ----------					
Large	15.7 a					
Medium	13.7 ab					
Small	12.6 b					
	---------- 2008 individual flush length (*cm*) ----------					
All sizes	34.0 A	13.4 B	13.0 B	10.8 B	7.4 C	5.3 C

[a] Least square means followed by the same letter within a column (lower case) or a row (upper case) were not significantly different at the Bonferroni adjusted 0.05 level.

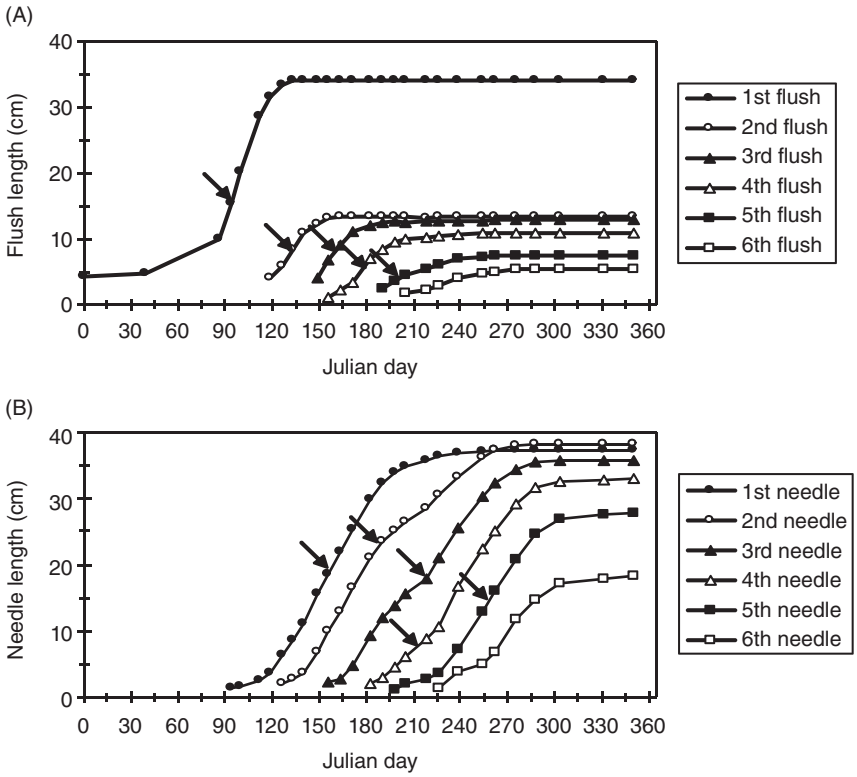

Figure 2—Temporal patterns of the fourth-year longleaf pine seedling (A) flush and (B) needle development in 2008. Arrows indicate the inflection point day for each curve.

Table 3—Lengths of needles from different flushes of the fourth-year longleaf pine seedlings in 2008

Container size	First	Second	Third	Fourth	Fifth	Sixth
	---------- *cm* ----------					
Large	37.3[a] A	38.8 A	36.4 AB	32.9 B	28.0 C	16.2 D
Medium	38.1 A	37.9 A	34.6 AB	32.0 BC	29.4 BC	22.9 D
Small	36.6 AB	38.0 A	36.2 AB	33.7 B	26.1 C	16.5 D

[a] Least square means within the same row followed by the same letter were not significantly different at the Bonferroni adjusted 0.05 level.

not shown). Nevertheless, as in 2008, needles from the sixth flush were shorter compared to needles from earlier flushes in 2007 (mean value of 25 cm for the sixth flush and mean value of 38 cm for the first five flushes).

Cavity type or size did not affect the starting date for all the flushes formed after the first flush in 2007 or 2008 (data not shown). Nor did the container type or size affect the duration of flush and its needle elongation for all six flushes in 2007 (data not shown) or 2008 (table 1). However, the order of flush had significant effects on the duration of shoot flush and needle extension in 2008 (table 4). Figure 2 presents the temporal patterns of shoot flush and needle development in 2008. When the first flush bud started to elongate in late February and early March, temperatures were still cool at night and at times during the day. When the second and third flush buds appeared in late April and May, respectively, temperature and soil water content at this site were favorable for plant growth. Therefore, duration of flush elongation was the shortest for these two flushes. Flushes that started between late June and July were affected by the drought and high temperatures in July and August in this area and thus had longer elongation duration than those formed in spring. Sword and others (1996) reported similar results of limited resource affecting flush elongation in loblolly pine (*P. taeda* L.) trees grown in this area. Although the flush elongation duration was long for the first flush, the instantaneous rate (cm per day) of flush elongation at the inflection point was still the highest for this flush (table 5). Except for the sixth flush, mean daily elongation rate for the first through the fifth flushes (flush length divided by flush duration, cm per day) were twofold to fourfold more than the reported values for shoot elongation in mature longleaf pine trees (Sheffield and others 2003). The duration of first flush elongation was greater than the values (20 to 69 days) reported for mature trees (Sheffield and others 2003).

During the development of the first flush, some of the current photosynthate produced by needles of the previous 1 or 2 years are used to grow the first flush stem and its needles. By the time the first flush needles finished elongation in mid-August, the seedlings had grown three or four more flushes.

Table 4—Elongation duration of flush and its needles in different flushes of fourth-year longleaf pine seedlings in 2008

Flush	Flush	Needle
	---------- *days* ----------	
First	86[a] a	117 b
Second	32 d	125 a
Third	30 d	116 b
Fourth	40 c	104 c
Fifth	44 bc	99 c
Sixth	52 b	85 d

[a] Least square means within the same column followed by the same letter were not significantly different at the Bonferroni adjusted 0.05 level.

Table 5—Slopes of flush and needle elongation at the inflection point day in different flushes of the fourth-year longleaf pine seedlings in 2008

Flush	Flush	Needle
First	0.91[a] a	0.52 a
Second	0.44 bc	0.43 b
Third	0.51 b	0.42 b
Fourth	0.38 c	0.49 a
Fifth[b]	0.23 d	0.49 a

[a] Least square means within the same column followed by the same letter were not significantly different at the Bonferroni adjusted 0.05 level.

[b] No logistics were analyzed for the sixth flush or needles because less than one-third of seedlings grew the sixth flushes.

From May through August, longleaf pine seedlings and trees in the Southern United States are active in stem height and diameter growth, fine root elongation, and storing starch reserves (Sung and others 2004, Sword-Sayer and Haywood 2006). Therefore, competition for current photosynthate produced by previous years' needles and the first flush needles of the current year within a seedling or tree is heavy. This could explain why the slopes of the second and third flushes were about 50 percent of the first flush slope (table 5) even though the environment is better for them to elongate than for the first flush. There generally was no lag period between flush elongation completion and the appearance of the subsequent flush bud. This developmental pattern is different from those of oak species where a 10-day-to-2-week lag period was reported between flushes (Hanson and others 1986, Sung and others 2004).

Duration of needle elongation lasted more than 3 months for all flushes except for the last flush (table 4). Sheffield and others (2003) also reported this unique developmental pattern of unsynchronized elongation between flush and its needles in mature longleaf pine trees. In loblolly pine and slash pine (*P. elliottii* Engelm), elongation of flush and needle occur simultaneously (Dougherty and others 1994). However, shoot flush and needle development of young loblolly pine trees in central Louisiana was also unsynchronized (Tang and others 1999). 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 seedlings started elongation before flush elongation was almost complete (fig. 2). Needles from the first three flushes had longer elongation duration than that from the fourth and the fifth flushes. The short elongation duration for the sixth flush needle was caused by the onset of short day length and cold temperatures. Slopes for needle elongation were dissimilar for flushes one through five (table 5). Slopes for the first, fourth, and fifth flush needles were greater than those of the second and third flush needles. Mean daily elongation rate for needles from the first through the fifth flushes (flush length divided by needle duration, cm per day) were 30 percent more than the reported values for needle elongation in mature longleaf pine trees (Sheffield and others 2003). We found the duration of first flush needle elongation was similar to the reported values from mature trees (Sheffield and others 2003).

The developmental patterns of flush and needles are only affected by the order of the flush and not by container treatments. The delineated developmental dynamics of flush and needle in longleaf pine should offer researchers a way to use flushes and needles of similar morphological attributes to evaluate treatment effects on some of the physiological parameters such as photosynthetic rate, chlorophyll contents, and carbohydrate pools. For example, one should note that even when needles have fully extended, it does not mean that these needles have reached physiological maturity.

FURTHER RESEARCH QUESTIONS

In longleaf pine plantation management, prescribed burns are usually implemented every 2 to 3 years and alternated between dormant season and growing season. This study site is scheduled to be burned in spring (April or May) of 2009. Shoot flush and needle development will be monitored in 2009 to assess how prescribed fire affects the developmental dynamics of flushes and needles. If the previous years' needles and the current year first and second flush needles are scorched by the fire, one would speculate that growth of the third and fourth flushes and needles has to come from the stored reserves in stems and taproots, and thus probably would not be sizable. How will the dormant season burn impact the developmental pattern of the next year's first flush and needles? Can one use the shoot flush and needle developmental dynamics to justify burning in one season versus the other?

LITERATURE CITED

Barnett, J.P. 2002. Longleaf pine: why plant it? Why use containers? In: Barnett, J.P.; Dumroese, R.K.; Moorhead, D.J., eds. Proceedings of workshops on growing longleaf pine in containers—1999 and 2001. Gen. Tech. Rep. SRS-56. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 5–7.

Barnett, J.P.; McGilvray, J.M. 2000. Growing longleaf pine seedlings in containers. Native Plants Journal. 1(1): 54–58.

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 fifth biennial southern silvicultural conference. Gen. Tech. Rep. SO-74. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 165–168.

Brockway, D.G.; Outcalt, K.W. 1998. Gap-phase regeneration in longleaf pine wiregrass ecosystems. Forest Ecology and Management. 106: 125–139.

Dougherty, P.M.; Whitehead, D.; Vose, J.M. 1994. Environmental influences on the phenology of pine. Ecological Bulletin. 43: 92–101.

Hanson, P.J.; Dickson, R.E.; Isebrands, J.G. [and others]. 1986. A morphological index of *Quercus* seedling ontogeny for use in studies of physiology and growth. Tree Physiology. 2: 273–281.

Krasowski, M.J. 2003. Root system modifications by nursery culture reflect on post-planting growth and development of coniferous seedlings. The Forestry Chronicle. 79: 882–891.

Landers, J.L.; Van Lear, D.H.; Boyer, W.D. 1995. The longleaf pine forests of the Southeast: requiem or renaissance? Journal of Forestry. 93(11): 39–44.

Outcalt, K.W. 2000. The longleaf pine ecosystem of the South. Native Plants Journal. 1(1): 42–53.

Ruehle, J.L. 1985. The effect of cupric carbonate on root morphology of containerized mycorrhizal pine seedlings. Canadian Journal of Forest Research. 15: 586–692.

SAS Institute Inc. 2004. SAS/STAT® 9.1 user's guide. Cary, NC: SAS Institute Inc. 5,121 p.

Sheffield, M.C.P.; Gagnon, J.L.; Jack, S.B.; McConville, D.J. 2003. Phenological patterns of mature longleaf pine (*Pinus palustris* Miller) under two different soil moisture regimes. Forest Ecology and Management. 179: 157–167.

South, D.B.; Harris, S.W.; Barnett, J.P. [and others]. 2005. Effect of container type and seedling size on survival and early height growth of *Pinus palustris* seedlings in Alabama, U.S.A. Forest Ecology and Management. 204: 385–398.

South, D.B.; Shelton, J.; Enebak, S.A. 2001. Geotropic lateral roots of container-grown longleaf pine seedlings. Native Plants Journal. 2: 126–130.

Sung, S.S.; Haywood, J.D.; Sword-Sayer, M.A. [and others]. 2010. Effects of container cavity size and copper coating on field performance of container-grown longleaf pine seedlings. In: Stanturf, John 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.; Kormanik, P.P.; Zarnoch, S.J. 2004. Flush development dynamics in first-year nursery-grown seedlings of eight oak species. 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: 542–546.

Sung, S.S.; Otrosina, W.J.; Zarnoch, S.J. 2004. Seasonal sucrose metabolism in longleaf pine tree stem cambial tissues. 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: 492–495.

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, M.A.; Gravatt, D.A.; Faulkner, P.L.; Chambers, J.L. 1996. Seasonal branch and fine root growth of juvenile loblolly pine five growing seasons after fertilization. Tree Physiology. 16: 899–904.

Tang, Z.; Chambers, J.L.; Guddanti, S. [and others]. 1999. Seasonal shoot and needle growth of loblolly pine responds to thinning, fertilization, and crown position. Forest Ecology and Management. 120: 117–130.

EFFECTS OF CANOPY TREATMENTS ON EARLY GROWTH OF PLANTED LONGLEAF PINE SEEDLINGS AND GROUND VEGETATION IN NORTH CAROLINA: A PRELIMINARY STUDY

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Abstract—We installed a field experiment to support the development of protocols to restore longleaf pine (*Pinus palustris* Mill.) to existing mature loblolly pine (*P. taeda* L.) stands at Camp Lejeune, NC. Seven canopy treatments included four uniform and three gap treatments. The four uniform treatments were defined by target residual basal area (BA) [control (uncut), BA9, BA4.5, and BA0 (clearcut) m^2/ha] and three gap treatments: designated large-gap (5027 m^2), medium-gap (2827 m^2), and small-gap (1257 m^2). We quantified treatment effects on planted longleaf pine seedlings and ground vegetation after the first-growing season. Canopy treatments significantly affected the growth of planted longleaf pine seedlings, with the largest root-collar diameter on BA0 (12.4 mm) and the smallest on control (10.6 mm). Total vegetation cover and total herbaceous cover, measured in both May and September 2008, were also significantly influenced by canopy treatments.

INTRODUCTION

Throughout the Southeastern United States, fire suppression and exclusion has resulted in the replacement of historically dominant longleaf pine (*Pinus palustris* Mill.) (LLP) with faster growing, less fire-tolerant species, especially loblolly pine (*P. taeda* L.) (LBP). Compared to LBP, LLP is longer lived, less susceptible to a variety of pests and diseases, and an ideal habitat for the red-cockaded woodpecker (*Picoides borealis*) (RCW) and other rare animals and plants (Walker 1993). Once established, LLP stands are conducive to management with prescribed fire. LLP restoration could be accomplished by clearcutting the existing canopy trees and planting LLP seedlings, because LLP is widely recognized as intolerant of competition for light, moisture, and nutrients (Boyer 1990). However, because the widespread loss of LLP forests has resulted in existing RCW populations using LBP stands for nesting and foraging habitat in recent decades, clearcutting is not desirable (U.S. Fish and Wildlife Service 2003).

The structure of naturally regenerated LLP stands and the results of recent studies that examined the response of naturally and artificially established LLP seedlings in canopy gaps within LLP overstories (Brockway and Outcalt 1998; Gagnon and others 2003; McGuire and others 2001; Palik and others 1997, 2003; Rodriguez-Trejo and others 2003) suggest that LLP could be restored with partial canopy retention, a strategy that would retain RCW habitat values. However, protocols for restoring LLP in LBP stands while retaining a LBP canopy sufficient for RCW use are not currently available. Our study was designed to answer the question: What are optimal silvicultural practices for restoring LLP to LBP stands while retaining mature trees and enhancing the herbaceous ground layer? In this preliminary report, we quantified the effects of canopy treatments on early growth of planted LLP seedlings and ground-layer vegetation based on data collected in 2007 and 2008.

METHODS AND MATERIALS

Study Site

This study was conducted on Marine Corps Base Camp Lejeune, in Onslow County, NC. The area is located within the Atlantic Coastal Flatlands Section of the Outer Coastal Plain Mixed Forest Province (Bailey 1995). The climate is classified as warm-humid temperate with hot, humid summers and mild winters. Mean annual temperature is 16 °C, and annual precipitation averages 1420 mm, which is evenly distributed throughout the year (National Climatic Data Center, Asheville, NC). The sites are on well-drained soils with low-to-moderate available water-holding capacity, including the Norfolk loamy fine sand, Wando fine sand, and Baymeade fine sand soil series (Barnhill 1992).

We used information from land managers at Camp Lejeune to identify two types of LBP stands in greatest need of conversion to LLP. The first condition included extensive, 35-year-old LBP plantations established on sites that are better suited for LLP, and the second stand type was dominated by 60-year-old LBP canopies, composed of large trees at irregular spacing.

Experimental Design

The study used a randomized complete block design, with location as the blocking factor, and consisted of seven treatments replicated on eight blocks. The study area was harvested from February to May 2007, by a local logging crew frequently used at Camp Lejeune. We were unable to apply the large-gap (LG) treatment to one of the blocks due to spatial constraints within the forest.

Seven canopy treatments (described in table 1) were applied in each block, with each main plot receiving a randomly assigned canopy treatment. Prior to planting LLP seedlings,

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Table 1—Summary of overstory treatments implemented in the study

Treatment	Silvicultural practice	Plot size	Number
Control	No cutting	100 by 100 m	8
BA9	Single tree selection to create uniform canopy with target basal area of 9 m^2/ha	100 by 100 m	8
BA4.5	Single tree selection to create uniform canopy with target basal area of 4.5 m^2/ha	100 by 100 m	8
BA0	Clearcut with basal area of zero m^2/ha	141 by 141 m	8
LG	Group selection to create circular "large" canopy gap with radius = 40 m	120 by 120 m	7
MG	Group selection to create circular "medium" canopy gap with radius = 30 m	100 by 100 m	8
SG	Group selection to create circular "small" canopy gap with radius = 20 m	100 by 100 m	8

our study sites were prepared with mechanical mowing in the late summer of 2007 and prescribed burning in fall 2007. Container-grown LLP seedlings were handplanted in December 2007 at a spacing of 1.8 by 3 m by contracted crews. Root-collar diameter (RCD) of planted LLP seedlings was measured in March 2008 and averaged 8.62 mm with a standard deviation of 1.49 mm.

Data Collection

Postharvest Stand Structure—In the summer of 2007, we measured characteristics of stand structure to describe postharvest conditions and to assess treatment uniformity among the blocks. Within each uniform treatment (control, BA9, BA4.5, and BA0), we permanently marked all overstory trees [diameter at breast height (d.b.h.) ≥ 10 cm] with aluminum tags and recorded species and d.b.h. Tree height was measured on a subsample of canopy LBP ($n \geq 30$) with a Laser Technology Impulse 200 Laser rangefinder. D.b.h. measurements were converted to basal area (BA) (m^2/ha) at the plot level.

Within each gap treatment—large-gap (LG), medium-gap (MG), and small-gap (SG)—we marked all overstory trees extending 20 m from the gap edge into the forest and recorded species and d.b.h. BA within the matrix surrounding each gap was then calculated from d.b.h. measures. Distance and azimuth from gap center to each tree within 10 m of the gap edge were recorded to determine the spatial distribution of trees surrounding each gap. Height of each LBP tree extending 10 m from gap edge into the forest was measured with a Laser Technology Impulse 200 Laser rangefinder.

LLP Seedlings Survival and Growth—In the summer of 2008, within each uniform treatment plot (control, BA9, BA4.5, and BA0), a sample of 120 seedlings was randomly selected and permanently marked for repeated measurements. Within each gap plot (LG, MG, and SG), we selected four rows of seedlings spaced at equal intervals and permanently marked each seedling for repeated measurements. Distance from the center of the measurement row to each tagged seedling was also recorded in the gap plots. RCD was measured for each marked seedling using digital calipers in October 2008.

Vegetation—We surveyed the amount (cover) of ground-layer vegetation in the beginning (May/June) and end (September/October) of the 2008 growing season. Within each uniform treatment (control, BA9, BA4.5, and BA0), we established eight parallel, 20-m transects across each plot, with the position of each transect located randomly. Along each transect, we randomly located ten 1- by 1-m sampling quadrats, for a total of 80 quadrats per plot. Within each gap treatment (LG, MG, and SG), we established eight transects along the four rows selected for seedling measurements, with one transect running north and one transect running south from each row center. Along each transect, we established ten 1- by 1-m sampling quadrats at equal intervals, covering the gradient of conditions from gap center to forest edge. All transects were permanently marked with nails and flags, and quadrat locations were recorded for future measurement.

Within each 1-m^2 sampling quadrat, we recorded ocular estimates of the percentage of the quadrat covered by vegetation <1 m tall. The vegetation cover was recorded by functional group (bunchgrasses, other graminoids, ferns, forbs, woody shrubs/trees, and woody vines), species groups of interest (legumes, invasive species), and other (species of interest, pine straw, coarse woody debris, bare mineral soil, and disturbance). Cover was recorded using the following cover classes: 1 = trace, 2 = 0 to 1 percent, 3 = 1 to 2 percent, 4 = 2 to 5 percent, 5 = 5 to 10 percent, 6 = 10 to 25 percent, 7 = 25 to 50 percent, 8 = 50 to 75 percent, 9 = 75 to 95 percent, and 10 = 95 to 100 percent.

Data Analysis

LLP Seedlings Growth—We tested effects of canopy treatments on RCD of LLP seedlings with analysis of variance using PROC GLM in SAS (SAS Institute Inc 2004). Log transformations were used to normalize data.

Vegetation—We analyzed the cover data to detect canopy treatment effects on total vegetation cover, total herbaceous cover, and total woody cover. Plot means were calculated for the analysis using midpoints of the cover classes recorded in the field. Data were analyzed using PROC GLM in SAS (SAS Institute Inc 2004).

RESULTS AND DISCUSSION

Postharvest Stand Structure

BA of control averaged 16.4 m^2/ha with a standard error (SE) of 1.14 m^2/ha (fig. 1). BA of BA9 was slightly smaller than our target BA, with an average of 8.48 m^2/ha (0.53 SE). However, BA of BA4.5 was almost 40 percent larger than our target BA with a mean of 6.17 m^2/ha (0.18 SE). The residual matrix of overstory trees surrounding gaps had a wider range of BAs, with standard errors of 1.78 m^2/ha in LG, 1.45 m^2/ha in MG, and 1.76 m^2/ha in SG. In the majority of gap plots, the residual canopy had BAs that were similar to control, with the averages of 14.4 m^2/ha in LG, 16.5 m^2/ha in MG, and 17.9 m^2/ha in SG.

Mean d.b.h. and tree height of residual LBP trees at the block level are shown in figure 2. Mean d.b.h. ranged from 26.5 cm in block 3 to 45.4 cm in block 8. Mean tree height varied from 18.3 m in block 4 to 27.9 m in block 8. The size of residual LBP trees reflected the two stand conditions at Camp Lejeune. Blocks 1 through 4, established in 35-year-old LBP plantations, had smaller mean d.b.h. (28.8 cm) and tree height (20.5 m) than blocks 5 through 8, which were established in 60-year-old natural LBP stands (mean d.b.h. of 42.6 cm and tree height of 26.5 m).

Root-Collar Diameter Growth

After the first-growing season, RCD was significantly affected by canopy treatment (F = 4.52, P = 0.0013) (fig. 3) with RCD largest on BA0 (12.4 mm) and smallest on control (10.6 mm).

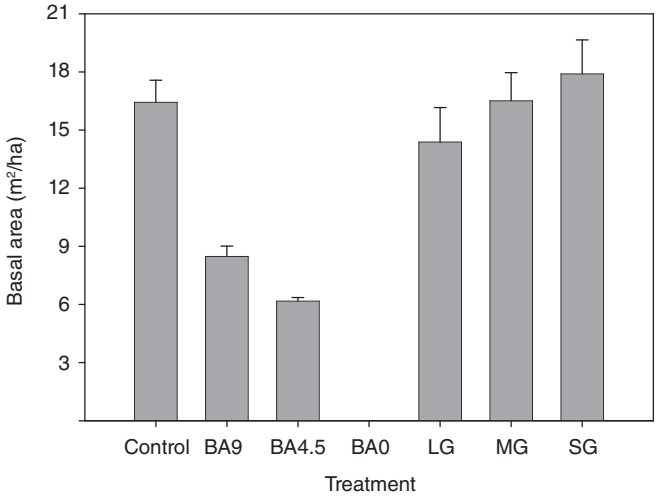

Figure 1—Stand basal area (mean ± 1 standard error) by canopy treatments.

Across the four uniform treatments (control, BA9, BA4.5, and BA0), there was a consistent increasing trend in RCD as residual overstory LBP BA decreased (mean RCD of 10.6 mm on control, 11.6 mm on BA9, 12.0 mm on BA4.5. 12.4 mm on BA0). Similarly, Palik and others (1997) found that the growth of container-grown LLP seedlings (measured as above- and belowground biomass) 1 year after planting increased with decreasing BA of overstory LLP at the Joseph W. Jones Ecological Research Center in southwestern Georgia, U.S.A.

We found that gap size positively affected RCD of planted LLP seedlings, although differences in RCD among gap treatments were not significant (11.0 mm, 11.5 mm, and 11.6 mm for SG, MG, and LG, respectively). Our results agreed with the study of McGuire and others (2001), conducted in 60- to 90-year-old, second-growth LLP stands in southwestern Georgia, who found that average RCD of planted LLP seedlings was similar among three gap sizes (0.11 ha, 0.41 ha, and 1.63 ha) after the second growing season, although average RCD of planted LLP seedlings was larger within gap openings than under intact LLP canopies (12 mm vs. 9 mm, respectively). Our results and those of McGuire and others (2001) suggest that early growth of planted LLP seedlings within gaps may not be strongly affected by gap size.

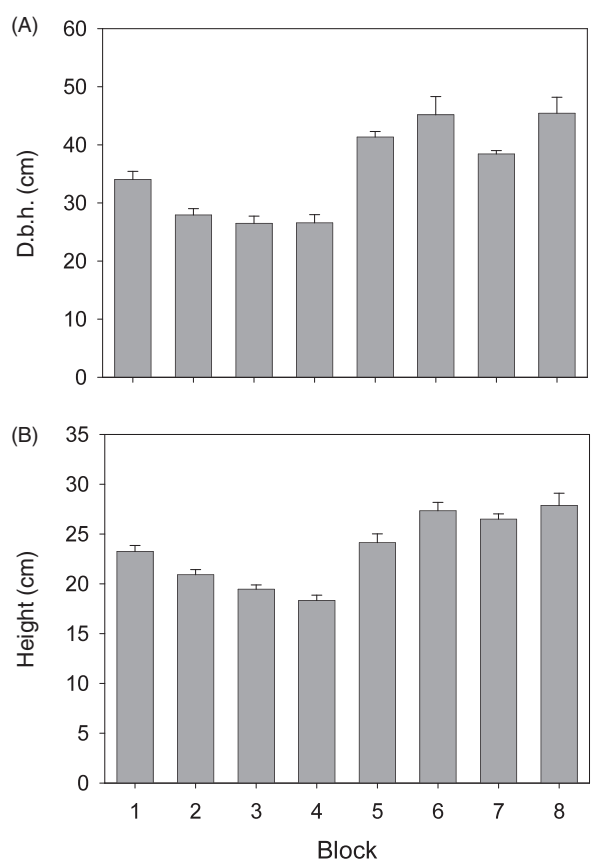

Figure 2— (A) D.b.h. (mean ± 1 standard error) and (B) height (mean ± 1 standard error) of loblolly pine by block.

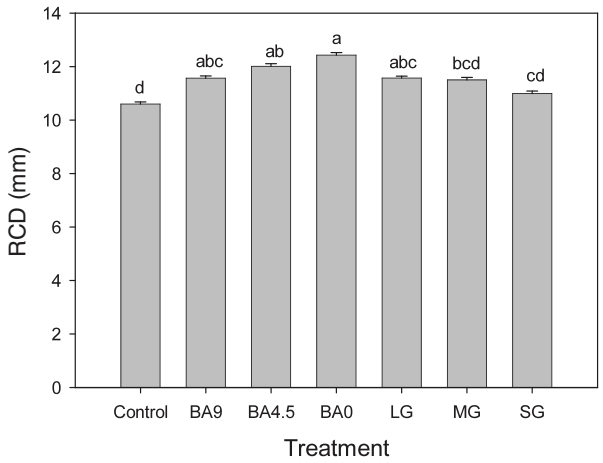

Figure 3—Root-collar diameter (RCD) (mean ± 1 standard error) of first year longleaf pine seedlings by canopy treatments in October 2008. The same letters indicate no significant difference at α <0.05.

Vegetation

Total vegetation cover was significantly influenced by canopy treatments in May (F = 4.22, P = 0.0021) and September (F = 5.54, P = 0.0003) (fig. 4A). In May, LG and MG had the largest total vegetation cover; BA9, BA0, SG, and BA4.5 were intermediate; and control had the least. Similarly, in September, all of thinning treatments (MG, LG, BA0, BA4.5, BA9, and SG) had significantly greater total vegetation cover than control. Total herbaceous cover (the sum of bunchgrass, other graminoid, fern, and forb cover) was also significantly influenced by canopy treatment in May (F = 3.28, P = 0.0099) and September (F = 3.40, P = 0.0081) (fig. 4B). In May, LG had significantly greater total herbaceous cover than control (34.5 percent vs. 14.4 percent, respectively). In September, only MG had significantly greater total herbaceous cover than control (45.1 percent vs. 19.9 percent, respectively). Although not statistically significant, we found that total herbaceous cover on control was less than any other treatment. Our results were similar to a study conducted in young and old Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco] stands in western Oregon (Bailey and others 1998), where they found that total herbaceous cover was greater in thinned stands relative to unthinned or old-growth stands, although commercial thinning had occurred 10 to 24 years previously.

Total woody cover (the sum of woody shrub/tree and woody vine cover) did not show significant differences among canopy treatments in either May or September (F = 0.93, P = 0.485 in May; F = 1.50, P = 0.202 in September; fig. 4C). Jack and others (2006) reported an increase in biomass of woody stems following harvest treatments that was likely associated with resource availability. In our study, the mechanical mowing used for site preparation removed standing vegetation and resulted in vigorous resprouting of woody stems during the first growing season. It is likely that the resprouting response was similar among treatments, regardless of the overstory condition.

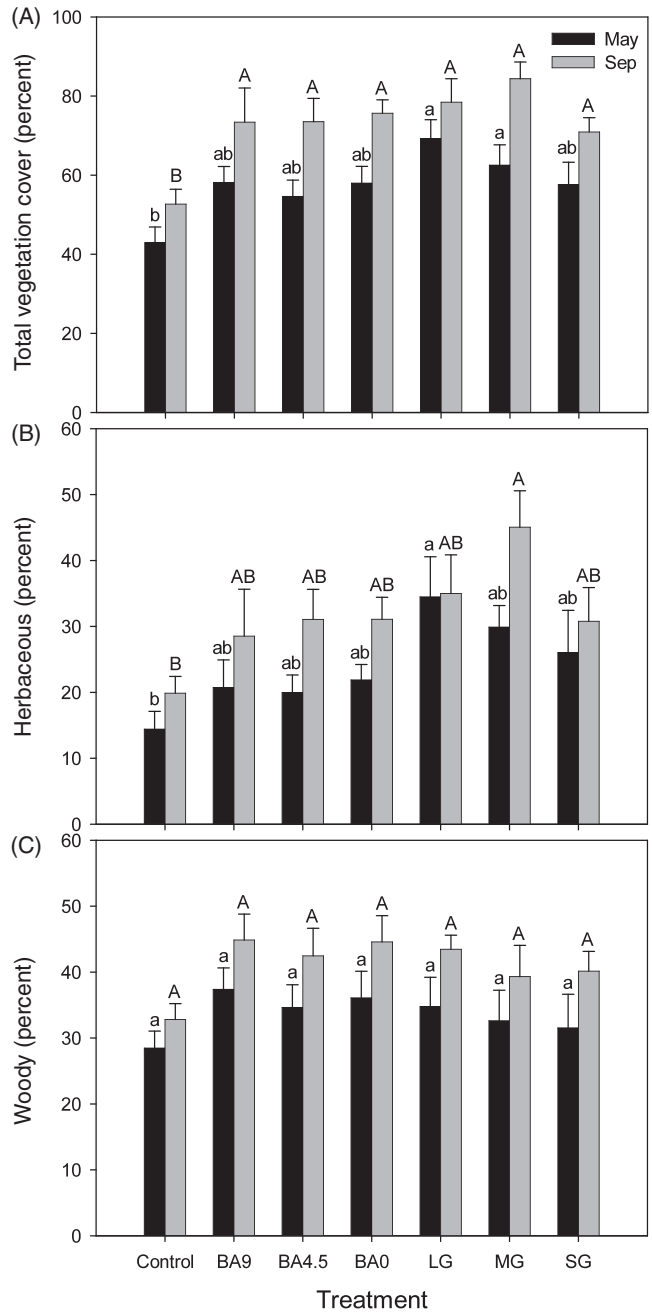

Figure 4—(A) Total vegetation cover (mean ± 1 standard error), (B) total herbaceous cover (mean ± 1 standard error), and (C) woody cover (mean ± 1 standard error) by canopy treatments in May and September 2008. The same letters within each time period indicate no significant difference at α <0.05.

CONCLUSIONS

It is well understood that LLP can be established following removal of canopy trees, but less is known about restoring LLP while retaining canopy trees for RCW habitat. After one growing season, we found RCD of planted LLP seedlings was larger on uniform thinning and gap treatments compared

to control. Canopy gap size did not affect RCD and may be a useful restoration option for regenerating pockets of LLP within a matrix of RCW habitats, as suggested by Palik and others (1997). As expected, all uniform thinning and gap treatments had greater total vegetation cover by the end of the growing season than control. Total herbaceous cover, important for good quality RCW habitat, was also greater in uniform thinning or gap treatments than control, although some of those differences were not statistically significant. However, total woody cover was not different among the treatments, a likely result of vigorous resprouting on all treatments following site preparation.

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LITERATURE CITED

Bailey, J.D.; Mayrsohn, C.; Doescher, P.S. [and others]. 1998. Understory vegetation in old and young Douglas-fir forests of western Oregon. Forest Ecology and Management. 112: 289–302.

Bailey, R.G. 1995. Description of the ecoregions of the United States. 2d ed., rev. Misc. Publ. 1391. Washington, DC: U.S. Department of Agriculture. 108 p. + map.

Barnhill, W.L. 1992. Soil survey of Onslow County, North Carolina. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 139 p. + map.

Boyer, W.D. 1990. Longleaf pine. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: 1. Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service: 405–412.

Brockway, D.G.; Outcalt, K.W. 1998. Gap-phase regeneration in longleaf pine wiregrass ecosystems. Forest Ecology and Management. 106: 125–139.

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.

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

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 response in a longleaf pine ecosystem. Canadian Journal of Forest Research. 27: 1458–1464.

Palik, B.J.; Mitchell, R.J.; Pecot, S. [and others]. 2003. Spatial distribution of overstory retention influences resources and growth of longleaf pine seedlings. Ecological Applications. 13: 674–686.

Rodriguez-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.

SAS Institute Inc. 2004. SAS user's guide: statistics. 9.1th ed. Cary, NC: SAS Institute Inc. 816 p.

U.S. Fish and Wildlife Service. 2003. Recovery plan for the red-cockaded woodpecker (*Picoides borealis*). 2d rev. Atlanta. 296 p.

Walker, J.L. 1993. Rare plants associated with longleaf pine: patterns in life history and ecology. In: Hermann, S.M., ed. Proceedings of the Tall Timbers fire ecology conference. The longleaf pine ecosystem: ecology, restoration, and management. Tallahassee FL: Tall Timbers Research Station. Timbers Fire Ecology Conference Proceedings. 18: 105–125.

SIXTY YEARS OF MANAGEMENT ON A SMALL LONGLEAF PINE FOREST

Rebecca J. Barlow, John S. Kush, and William D. Boyer[1]

Abstract—A management demonstration in a 40-acre tract of second-growth longleaf pine (*Pinus palustris* Mill.) had its 60th anniversary in 2008. A demonstration was initiated by the U.S. Forest Service in 1948 on the Escambia Experimental Forest in south Alabama. At the time, the management goal for this Farm Forty was to produce high-quality poles and logs on a 60-year rotation. The goal was to be accomplished entirely through management of the existing natural forest with little to no capital investment other than the cost for prescribed burning, marking trees for cut, and limited control of cull hardwoods. Since that time, management has continued making the stand an excellent demonstration of small-scale longleaf pine management. This paper celebrates the 60th anniversary of the Escambia Farm Forty with discussions of standing volume of merchantable pine timber at selected inventories plus volumes harvested between inventories.

INTRODUCTION

Some of the earliest observations about forested systems in the United States were written about longleaf pine (*Pinus palustris* Mill.). In 1840 after traveling through the northern part of the Mississippi Piney Woods, historian John F.H. Claiborne (1906) wrote "The growth of giant pines is unbroken for a hundred miles or so, save where river or large water courses intervene. . . . Much of it is covered exclusively with the longleaf pine; not broken, but rolling like the waves in the middle of the great ocean. The grass grows three feet high and hill and valley are studded all over with flowers of every hue. The flora of this section of the state and thence down to the sea border is rich beyond description."

The first European settlers in what is now the Southeastern United States were confronted with an upland forest that was dominated by a single species of tree—longleaf pine. Stretching from the Coastal Plain of southern Virginia across a broad belt of the South Atlantic and Gulf Coasts into eastern Texas, longleaf occurred on nearly 90 million acres and on over 60 million of those acres, longleaf alone dominated the overstory. From Virginia to Texas, it dominated the Coastal Plain and sandhills and extended into the Cumberland Plateau, Valley and Ridge, Blue Ridge, and Piedmont physiographic provinces.

Today, only about 3 million of the original 90 million acres still support longleaf pine—a loss of over 96 percent of the original longleaf forest acreage. Sadly, even fewer acres retain examples of an intact, functioning longleaf ecosystem with all associated plants and wildlife. Longleaf pine forests have been listed as one of the rarest ecosystems in the United States. The rangewide, large-scale reduction of this ecosystem began with the cutting of the original forests at the turn of the 20th century with little to no regard for regenerating stands after they were cut. This coincided with a major effort to exclude and/or suppress all fires including the frequent, low-intensity fires which are critical for longleaf pine regeneration. Other reasons for the severe decline include conversion to non-longleaf pine plantations, agriculture, urbanization, and development.

With the longleaf pine resource declining, the Forest Service, U.S. Department of Agriculture established the Escambia Experimental Forest (EEF) in 1947 with the hope that Forest Service scientists could find ways to provide for longleaf's restoration, with special emphasis on its regeneration. The EEF is a 3,000-acre tract near Brewton, AL, on private land owned by T.R. Miller Mill Company. The company, interested in the higher prices longleaf timber commanded, leased the property to the Forest Service for 99 years.

At the time the EEF was established, about half of the forest land over most of the South was in small ownerships. Many of these tracts had been heavily cutover in the past, and returns were low from any forestry activity. As a result, many owners were often uncertain as to their land's best use. What were costs and returns when such lands were intensively managed and the best-known practices applied? It was in response to questions such as this that researchers established the Farm Forty in 1948 as a demonstration of small woodlot management.

The Farm Forty

The Farm Forty was established on the EEF as a demonstration of longleaf pine forest management for the small-scale private forest landowner. At the time of establishment, the Farm Forty supported an understocked, 35- to 45-year-old, second-growth longleaf pine forest that was common on many farm forests in the Coastal Plains of the Gulf South (Boyer and Farrar 1981). The tract is predominantly longleaf pine (31 acres), with the remaining 9 acres comprised of mostly slash pine (*P. elliottii* Engelm.) in the creek bottoms and flats. Site index for longleaf on the Farm Forty at the time it was established was average—70 feet at 50 years. The results of the first 30 years of management and demonstration on the Farm Forty were reported by Boyer and Farrar (1981).

MANAGEMENT

The long-term management goal of the forest was to grow high-quality sawtimber and poles on a 60-year rotation (Boyer and Farrar 1981). With the private landowner in mind, this goal was to be accomplished through management of

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the existing forest with little to no additional investment or expense. Primarily expenses were from cost of prescribed burning and control of hardwoods. Regeneration and intermediate cuts provided regular income from the forest; however, removals were not to exceed growth until full stocking, and a balanced distribution of age classes was achieved.

Beginning in 1948, initial harvests were conducted annually. These harvests removed the poorest quality trees through thinnings and improvement cuts to gradually improve growing stock. Shelterwood cuts were used to recruit younger age classes and promote natural regeneration on the forest starting in 1954. Harvest areas were established on the forest so that small areas, 2 to 4 acres in size, were harvested leaving the highest quality seed bearers in the overstory. Seed from these remaining overstory trees was used to regenerate the stand. Once the new stand of young trees was established sawtimber-sized trees were removed from the overstory, often through a series of thinnings. Through time, a number of age classes were developed on the Farm Forty with the goal of always having mature stands to be harvested so there could be continuous revenue from the forest.

Other than periodic harvests, the primary cultural treatment and expense during the first 30 years of management on the Farm Forty was prescribed fire. Through the use of winter burns, the entire Farm Forty was burned seven times for brush control, hazard reduction, and seedbed preparation. Management costs were intentionally kept low to demonstrate to landowners how they could manage their property with limited resources. The only additional costs beyond burning expenses were marking trees for harvest and control of cull hardwoods.

Management over the second 30 years continued with periodic shelterwood harvests to promote regeneration, and tracts were burned on a 2- to 3-year cycle (Boyer and Farrar 1981). Season of burn was shifted from winter to spring burns during this time to improve hardwood control. Additional benefits from the spring burns included better control of hardwoods in low lying areas, and reestablishment of native cane breaks [*Arundinaria gigantea* (Walt.) Muhl.] and pitcher plants (*Sarracenia* spp.) in the flats.

GROWTH AND HARVEST

Growth and removal of pine volumes on the Farm Forty from 1977 to 2007 are summarized in table 1. During the period

Table 1—Total stand per-acre volumes in cubic feet, sawtimber stand per-acre volumes in cubic feet, and International 1/4 and Doyle log rules for the Escambia Experimental Forest Farm Forty from 1977 to 2007

Farm Forty 1977 to 2007	Total stand per acre (>3.5 inches d.b.h.)	Sawtimber stand per acre (>9.5 inches d.b.h.)		
	cubic feet	*cubic feet (stem only)*	*board feet International 1/4-inch*	*board feet (Doyle)*
Inventory 1977	1,194	855	5,408	3,268
Increase 1962 to 1977	261	133	959	794
Cut 1963 to 1977	275	219	1,351	768
Growth 1963 to 1977	536	352	2,310	1,562
Inventory 1987	1,392	1,003	6,417	4,022
Increase 1978 to 1987	198	148	1,009	754
Cut 1978	159	89	537	281
Growth 1978 to 1987	357	237	1,546	1,035
Inventory 1992	1,426	1,010	6,455	4,072
Increase 1988 to 1992	34	7	38	50
Cut 1988	140	119	748	446
Growth 1988 to 1992	174	126	786	496
Inventory 1997	1,533	1,084	6,914	4,334
Increase 1993 to 1997	107	74	459	262
Cut 1993	47	38	211	112
Growth 1993 to 1997	154	112	670	374
Inventory 2002	1,593	1,137	7,226	4,491
Increase 1998 to 2002	60	53	312	157
Cut 1998	261	184	—	—
Growth 1998 to 2002	321	237	—	—
Inventory 2007	1,457	1,034	6,529	3,979

from 1977 to 2007, overall volume per acre increased 22 percent, and sawtimber volume increased 15 percent to 1,084 cubic feet per acre from 1977 to 1997 (table 1). Gains in total volume per acre slowed through the 2002 inventory period, with a <5-percent increase to 1,593 cubic feet per acre.

September 15, 2004, Hurricane Ivan made landfall and impacted much of the EEF, including 10 acres of timber in shelterwood systems on the Farm Forty along with many other acres on the EEF. Salvage harvests were conducted over the following year; however, timber volumes harvested during these operations were not separated for the Farm Forty alone making it impossible to know exactly how much volume was lost as a result of Hurricane Ivan. Inventories conducted in 2007 show that there was a 9-percent volume decrease in total standing timber volume from the 2002 inventory (table 1). Decreases occurred in both sawtimber and pulpwood as standing volumes were reduced to 1,034 and 422 cubic feet per acre, respectively.

DISCUSSION

Using the shelterwood method, stands were thinned to approximately 30 square feet of basal area and naturally regenerated to mimic natural processes on a small scale. Eventually, older stands were removed as the new stand matured. Over time, a number of age classes were developed within the Farm Forty so there were always mature stands to be harvested providing periodic revenue from the forest with minimal cost to the landowner. An additional benefit of this method is that while a stand reestablishes itself, high-value wood can be grown on the remaining large, seed-bearing trees.

CONCLUSIONS

Beginning in 1948, the Farm Forty was set up as a demonstration forest on the EEF for private landowners to help them effectively manage their tracts with limited monetary input. Over 60 years of research on the Farm Forty has provided information vital to the southern landowner and timber manager.

Forest researchers continue to use the EEF to delve into many longleaf management concerns and problems. Research topics over the past 60 years have included natural regeneration, stand management and growth, site quality and soils, fire ecology, and woods grazing. The EEF is a much used outdoor demonstration area for the education and enjoyment of a host of visitors including school children and forestry students. It is a real-world, living demonstration of proven techniques that can be used to produce both tangible products and aesthetic values. As the timber grows and responds to forces such as hurricanes, and as interest in new forest products develops, the Farm Forty will continue to be managed with the perspective of the private landowner in mind.

A 15-minute video, “Sixty Years on the Farm Forty: Longleaf Pine Management for the Private Landowner,” was recently produced, which highlights the Farm Forty and its 60th anniversary. In addition, a Farm Forestry Field Day has been scheduled for May 2009. The authors may be contacted for more information on this and other activities on the Farm Forty.

LITERATURE CITED

Boyer, W.D.; Farrar, R.M. 1981. Thirty years of management on a small longleaf pine forest. Southern Journal of Applied Forestry. 5(2): 73–77.

Claiborne, J.F.H. 1906. A trip through the Piney Woods. Publications of the Mississippi Historical Society. 9: 487–538.

QUANTITATIVE SILVICULTURE AND ECONOMICS

Growth rings visible on a stump harvested from the Poor Farm Forestry Forty uneven-aged demonstration stand at the Crossett Experimental Forest, Arkansas. The demonstration began 69 years ago, prior to the germination date of this tree. (Photo by James M. Guldin)

THE ESTATE OWNER'S APPROACH TO FOREST ECONOMICS

David B. South and David N. Laband[1]

Abstract—Faustmann's formula is often used by forest economists but some landowners ignore the time value of money and rely primarily on some type of cashflow analysis. A cashflow method was used by "estate owners" like kings, land barons, and governments. We coin the term "estate owner's method" to describe one version of cashflow analysis. This approach can be used for either a "fully regulated forest" or for a forest that is harvested using the single-tree method. When calculating a return on investment, all past investments are "sunk" and all future returns and investments are ignored. Only costs and revenue realized during the current year are considered. This method is occasionally used by some large landowners to justify long rotations. Long rotations are easier to justify when time value of money is ignored or when the interest rate is close to zero.

INTRODUCTION

Prior to 1849, the nobility (who owned large tracts of forests) kept a close eye on their forest management accounts. Some estate owners hired foresters with graduate degrees from universities in France and Germany. Trained foresters were more likely to produce a continuous source of income from the forested estate than untrained managers who simply harvested stands with no concern for regeneration success. However, regardless of who was in charge of wildlife management and timber harvesting, the estate owner's approach to cashflow analysis was employed. This method does not involve calculating a "land expectation value" (LEV) and does not involve discounting.

In current times, some large estates are managed by foresters who use the single-tree selection method of harvesting (McIntyre and others 2008). Others' estates might involve establishing a "fully regulated" forest, a.k.a. normal forest, where the forest contained an even distribution of age classes, so that it would be capable of yielding the same volume of timber every year in perpetuity (Helms 1998, Tahvonen and Viitala 2006, Viitala 2006). Regardless of the harvesting method, the estate bookkeeper examines the income and expenses on an annual basis. If income from harvests and processing exceeded expenses, the estate owner is usually satisfied. However, if expenses exceeded income for the year, the landowner would ask the forester why the expenses were so high or the harvests so low. The next year, the process would begin anew with an annual accounting of costs and income. We define this method as the "estate owner's method" (EOM) of forest economics. This approach to analyzing forest costs and returns has been used by the South African Timber Growers' Association (1993) in South Africa.

In 1849, Faustmann wrote his now famous (among academics) article on calculating forest land value (Faustmann 1849). This formula has been adapted and is taught in most forestry schools in North America. This formula allows the user to determine an LEV which could be used to determine an inherent value of the land (assuming certain management practices were used in perpetuity). However, today some forest economists select the goal of "maximizing LEV" in order to determine the optimum site preparation methods (Busby and others 1998) or the optimum rotation age for even-age management (Caufield and others 1992, Huang and others 2005).

Although developed in Germany, Faustmann's formula is seldom used in the public sector in Germany (Ince 1999). It also is given low priority when managing plantations managed by the U.S. Forest Service. Government foresters in France manage publicly owned forests using a zero interest rate. In some circles, discount rates of 4 percent are viewed as socially unethical (Sukhdev 2008). Typically, the rotation age used for public forests is much longer than that based on maximizing LEV at discount rates of 3 percent or more. For example, in the Ukraine, the official rotation age for spruce is 81 to 100 years while LEV rotation (at 4 percent) would only be 32 years (Nijnik 2004). We wonder, how often is the LEV formula used by private landowners who own large estates? Do they tend to manage on long rotations and accept low discount rates, or do they tend to manage their estates under the objective of maximizing LEV?

Is Faustmann's Formula Used in the Real World?

Faustmann's LEV formula is likely used by forest industry economists and by those of us in academia. However, it seems that landowners—either owners of single-aged plantations or owners of estates—often have objectives that do not involve calculating an LEV (e.g., McIntyre and others 2008). Some landowners may desire a plantation that offers a "sustained cashflow." Others may want a forest that "maximizes return on assets." Some may want a forest with a high benefit/cost ratio. Some may want to own forests in order to diversify their investments. Some view the timber as an "insurance policy" while others may be reluctant to pay capital gains taxes on old stands (Haney and Siegel 1993). Some may prefer a management regime that results in asset appreciation rather than maximizing new present value (NPV) or internal rate of return (IRR) (Anonymous 2002). Some may simply want the forest to provide sufficient revenues to support operations well into the future (McIntyre and others 2008). In addition to simply ignoring the power of discounting (Bilek 1994, Henry 1994), these objectives can lead to adopting rotation lengths that extend well beyond a stand's economic maturity as determined by LEV.

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Paper Objectives

The primary objective of this paper is to outline an approach to forest accounting that is sometimes used by some estate owners. This approach does not involve discounting and is referred to as the EOM. This method does not account for any costs incurred in previous years.

The secondary objective of this paper is to illustrate how seductive the EOM may be when comparing the outcome of "fully regulated" plantations. With the EOM, the inherent value and risk of the standing (uncut) timber is not taken into account when calculating the benefit/cost ratio or IRR. As a result, the benefit/cost ratio is inflated when using the EOM method. Since the EOM does not take into account the foregone benefits and further risks associated with developing the "wood factory" the EOM method favors longer rotations while the LEV method favors shorter rotations (when the interest rate is sufficiently low).

MATERIALS AND METHODS

Objective 1

We follow the example of Haney and Siegel (1993) who provide examples of forest owners who own large estates. They provide and compare specific examples of forest management plans that are designed for large estates. To illustrate the EOM, we provide two examples of estates that contain fully regulated plantations.

The first example involves the landowner, Larry, who recently inherited an estate that contained 5000 ha of loblolly pine (*Pinus taeda* L.) plantations. The estate is fully regulated, i.e., 100 ha are harvested each year and 100 ha are planted each year. The estate is valued at $36 million ($15 million for the land and about $21 million for the trees). Larry has an experienced forester who manages the forest using a 50-year rotation (note: his annual salary is $50,000).

The second case involves Terry, who also inherited a 5000-ha estate consisting of "fully regulated" loblolly pine plantations. The estate is managed by a forester (salary = $50,000) on a 25-year rotation, i.e., 200 ha are harvested each year and 200 ha are planted each year. The estate is valued at $21 million ($15 million for the land and about $6 million for the trees).

Each year, both estate managers submit financial reports to show the costs, revenue, and net profits for the year (table 1). The EOM report only considers the current year's income and expenditures. All previous investments and past revenues (spent by the original estate owner) are ignored, a.k.a. sunk. The information in this financial report is used when filling out tax forms.

Objective 2

Each year, Terry prepares a financial report that compares the estimated LEVs for both estates (table 2). When calculating the LEV, a 6-percent discount rate is used and future stumpage values are assumed to be fixed at $7, $17, $35, and $60 per green ton for pulpwood, chip-n-saw, sawtimber, and poles (all poles are older than 28 years). An adjacent neighbor recently paid $3,000/ha for a large tract of cutover land (the forest had been harvested prior to selling the property).

RESULTS

According to the EOM, the annual cashflow was approximately $0.4 million higher for the 50-year rotation than for the 25-year rotation (table 1). The annual income per ha was about 39 percent greater for the longer rotation ($293 vs. $211/ha/year). The $36 million estate generates about $1.4 million in profits each year, i.e., 9.1 percent IRR, while the $21 million estate generates about $1 million in profits, i.e., 10.3 percent IRR.

Note, however, that the LEV for the 25-year rotation was slightly higher than the LEV for the 50-year rotation. In neither case did the LEV equal $3,000/ha. The EOM and LEV methods produced different benefit/cost ratios. Due to no discounting, the benefit/cost ratio was higher for the EOM (table 2).

DISCUSSION

A major difference between the LEV and EOM is the time value of money. With LEV, establishment costs are compounded over time and with long rotations, this reduces both the LEV and the benefit/cost ratio. In contrast, costs incurred in the past are "sunk" with the EOM and therefore these costs do not lower the benefit/cost ratio. Only costs during the current calendar year are included in the calculations. Therefore, if a reforestation check is written in December 2009, the value is "sunk" and does not enter into the EOM benefit/cost ratio for 2010.

Using the LEV method, the benefit/cost ratio is about same for both Terry's and Larry's estates, i.e., 2.5. In contrast, the EOM produced a benefit/cost ratio of 8 to 15.6 (table 2). The much higher ratio for Larry's estate might persuade some landowners to say the economics of long-rotation loblolly pine stands is attractive. However, we cannot compare "apples with oranges" and therefore any benefit/cost ratio calculated with no discounting should have a footnote to indicate that no discounting was used. We suspect that most forest economists would agree that discounting should be used when comparing forestry investments.

Larry's estate is worth about $14 million more than Terry's estate because of the stock of old-growth timber. All other things equal, when a 50-year-old stand is harvested, the wood is worth about $9,200/ha while a 25-year-old stand is worth about $4,250/ha. When discounted at 6 percent, these values are roughly $500 for the 50-year-old stand and $1,210 for the 25-year-old stand. Therefore, the high value of the standing trees looks more attractive to an estate owner when discounting is ignored. Discounting is not used with the EOM (table 3).

Should LEV Be Used to Compare Fully Regulated Forests?

Terry and Larry sat down to discuss the management regimes of their "normal forests." Terry said the 25-year

Table 1—The estate owner's method of cashflow analysis. Each estate contains 5000 ha (adjacent cutover land valued at $3,000/ha) and the following expenses are only for the year 2000. Each estate is fully regulated; Terry's estate is managed on a 25-year rotation while Larry's estate is managed on a 50-year rotation.

Estate	Treated each year	Per ha (cost) or revenue	EOM Annual cashflow
	ha	*dollars*	
Terry's estate			
Site preparation and planting	200	($500)	($100,000)
Forester costs	5000	($10)	($50,000)
Total costs			($150,000)
Thinning revenue (age15 years)	200	$715	$143,000
Harvest revenue (age 25 years)	200	$5310	$1,062,000
Total revenue		$6025	$1,205,000
Net profit for 2000			$1,055,000
Larry's estate			
Site preparation and planting	100	($500)	($50,000)
Forester costs	5000	($10)	($50,000)
Total costs			($100,000)
Thinning revenue (age15 years)	100	$715	$71,500
Thinning revenue (age 30 years)	100	$3,585	$358,500
Thinning revenue (age 40 years)	100	$2,017	$201,700
Harvest revenue (age 50 years)	100	$9,584	$958,400
Total revenue		$15,901	$1,565,000
Net profit for 2000			$1,466,000

EOM = estate owner's method.

rotation was the best option since it produced the highest LEV while Larry said the 50-year rotation was best since it produced a higher cashflow with less harvesting, i.e., 100 ha/year, and less capital expenditures. Larry said that he did not care if the LEV was 16 percent higher because his objective was to optimize cashflow. He did not care about a value that is based on the assumption that future stumpage prices and future management regimes would not change. Besides, the LEV is used to determine how much one could afford to pay for land, but Larry already owned the land. He had no intention of stopping forest management because the LEV was <$3,000/ha.

Which Option Would You Choose?

Larry said the question is not which regime has a higher LEV, but which regime is more profitable? Therefore, should Larry manage his estate? Should he continue to harvest on a 50-year cycle on a fully regulated basis, or should he convert the estate to a 25-year cycle? Terry thought for a while, ran several scenarios on her computer, and came up with several hypothetical "tipping point" alternatives (each with an annual cashflow of about $1.4 million).

A. The first case involved selling the entire estate for $36 million and investing the capital in a 3.9-percent Certificate of Deposit (CD). This would bring in an annual cashflow of $1.4 million.

B. This option keeps the land in the estate, but all the old timber is sold and the receipts are put into a 5-percent CD. All the 26- to 50-year-old timber, i.e., 2500 ha, is sold for about $18 million, i.e., $7,200/ha. The clearcut half would be allowed to regenerate naturally and the remaining stands (containing 2500 ha) would be managed on a 25-year rotation. The profit from annual timber harvests ($427,500 per year) could be combined

Table 2—A comparison of the LEV approach with the "estate owner's" approach to forest economics. Each estate is fully regulated, i.e., each year an equal amount of land is harvested. Terry's estate is on a 25-year rotation (clearcut of 200 ha/year) while Larry's estate is managed on a 50-year rotation (clearcut 100 ha/year). Cashflow is higher on Larry's estate while Terry's estate has a higher LEV.

	Terry's estate		Larry's estate	
Comparison items	Discounted value/ha at 6 percent	Annual cashflow	Discounted value/ha at 6 percent	Annual cashflow
Harvest revenue	$1,534		$1,638	
Total costs	($618)		($647)	
Net present value	$917		$991	
Equal annual equivalent	$72		$62	
Land expectation value	$1,196		$1,030	
Internal rate of return	10.3		9.1	
Annual revenue		$1,205,000		$1,566,000
Annual costs		($150,000)		($100,000)
Net profit for 2025		$1,055,000		$1,466,000
Profit per ha/year		$211		$293
Benefit/cost ratio	2.5	8.0	2.5	15.6
Land value		$15,000,000		$15,000,000
Value of standing timber		$6,600,000		$21,000,000
Asset value		$21,600,000		$36,000,000
Return on assets		4.9 percent		4.1 percent

LEV = land expectation value.

from the interest from the CD ($972,500 per year) to equal an annual cashflow of $1.4 million.

C. The third option is similar to the second, except that instead of relying on natural regeneration, 2500 ha are converted to a single plantation to be harvested in 25 years. All 2500 ha are replanted in year 2011 at a cost of $500,000. Each year, a loan (to be paid back in 2035) is obtained from the bank at 6 percent real to produce an income of $276,500 per year. All the 26- to 50-year-old timber, i.e., 2500 ha, is sold for $18 million. After paying the $0.5 million bill, the remaining $17.5 million is invested in a 4-percent CD (to yield $696,000 per year). This, plus the $276,500-per-year loan plus the $427,500 per year results in an annual cashflow of $1.4 million.

Therefore, if a cashflow of $1.4 million per year is desired, it could be achieved either by: (1) investing $36 million at 3.9 percent (real); (2) investing $18 million at 5 percent (real) and managing half of the estate on a fully regulated 25-year rotation; (3) investing $17.5 million at 4 percent (real), borrowing $204,000 per year, converting half the entire estate to a single stand (25-year rotation) and leaving the remaining half on a fully regulated basis; or (4) managing a 5000-ha, fully regulated estate on a 50-year rotation, i.e., no change in management.

Terry said that when banks are only offering 3-percent real interest rates for CDs, then Larry's current management (option 4) would be favored over option 1. In contrast, if banks are offering a 5-percent real interest rate CDs, then option 1 would be favored over option 4.

Larry said he has no intention of selling his property. Although the risk might be less if he had $36 million in the bank, he chooses to accept the 0.5-percent-per-year risk of losing his timber in a wildfire. Larry also said the tax implications would likely favor option 4 over 1, especially in cases when the capital gains taxes were lower than the personal income tax rate. Since option 1 is not on the table, why would Larry choose option 2 or 3 just because the LEV/ha is higher? With option 3, he is taking the risk that a hurricane might destroy much of his stand prior to paying off the bank loan.

Is the Estate Owner's Method Similar to Using a Discount Rate Near Zero?

When the discount rate selected is 0.00001 percent, then, in some cases, the classical method of forest economics and

Table 3—A comparison of the land expectation value method of forest economics with the estate owner's method of cashflow analysis.

Comparison items	LEV	EOM
Harvest age required?	Yes	No
Useful for optimum afforestation evaluation?	Yes	No
Useful for thinning regime comparison?	Yes	No
Useful for a single stand?	Yes	No
Used to determine optimum LEV?	Yes	No
Assumes no change in future management?	Yes	No
Assumes fixed stumpage prices?	Yes	No
Discounting used?	Yes	No
Fully regulated forest required?	No	Desired
Considers only current year's costs?	No	Yes

LEV = land expectation value; EOM = estate owner's method.

the EOM will produce similar results. However, since the EOM "sinks" costs for previous years, the two methods are not identical when the discount rate is zero. The output from the EOM will fluctuate from year to year (depending on the costs and returns for that year), while results from using the Faustmann LEV formula are more stable (since users of the formula assume constant stumpage prices, constant labor prices, and a constant interest rate).

Do All Estate Owners Use the Estate Owner's Method?

Some estate owners do not use the EOM spreadsheet. In some cases, cashflow methods include inflating stumpage values 50 years into the future (McIntyre and others 2008). Occasionally, foresters compare management records over a 17-year period (Handley and Dickinson 2008). When cashflow methods involve more than one calendar year or predict future stumpage values, they do not qualify as an EOM.

The Estate Owner's Method Is Not a Method Used to Determine the Optimal Rotation

The EOM is an inappropriate method of comparing various management options since previous stand management costs are "sunk." Since the EOM uses actual stumpage values (that vary with year and season of harvest), the benefit/cost ratio will vary from year to year. In contrast, with the LEV method, the value of products at harvest is assumed to be constant and therefore the theoretical LEV for 1 year will be the same as for the next. In some cases, the "sinking of past establishment costs" might mislead landowners into thinking that planting trees for a 46-year rotation would be more profitable than for a 23-year rotation. This is because the EOM produces a benefit/cost ratio that is higher for the longer rotation (due to comparing costs and benefits for 1 year). The net cashflow for the 46-year-old pine plantation is higher than for the 23-year plantation (due in part to planting twice as many ha/year). When considering tree planting on an old field, the LEV is the appropriate method to apply when answering the question: What rotation length is optimum when the objective is to maximize the internal rate of return?

Risk

Risk can be included in both EOM and LEV comparisons. However, there are two schools of thought when it comes to assessing risks associated with long rotations. Some believe 50-year rotations of loblolly pine carry more risk while others say that 25-year rotations carry more risk. One school believes that old loblolly pines are more susceptible to beetles than young stands, especially when the basal area is high (in most years after age 25 years). Some say the risk of damage and windthrow from hurricanes is greater soon after thinning and long rotations may average more thinnings per century. Loblolly pine decline is more likely when thinning and burning cause stress to develop in loblolly pine plantations (Menard 2007).

In contrast, the other school believes risks are lower with long rotations. When consideration is given to fluctuations in stumpage prices, price appreciation, and discount rates, management for high-value products like poles may indicate a reduced risk (Anonymous 2002). In addition, historical performance has proven that favorable rates of return can be achieved from holding properties with higher near-term cashflows (MacKay 2001).

Quantifying risk is difficult which explains why risk is often ignored when comparing various scenarios. When a landowner inherits a long-rotation estate, e.g., Larry's estate, the decision to shorten the rotation might have more to do with the perceived risks than with the perceived LEV.

CONCLUSIONS

Many landowners pay taxes on their forestry investments and, therefore, each year they examine the costs and revenues from their forest land. The EOM can easily be applied each year at tax time. In contrast, the LEV method is more complicated and is typically not required for most landowners' objectives. For example, some landowners are more concerned with maximizing cashflow than they are with maximizing LEV. In general, maximizing cashflow will favor longer rotations, i.e., 35 to 50 years for southern pines, while maximizing LEV will favor shorter rotations, i.e., 20 to 25 years if a reasonable interest rate is selected.

Many landowners have multiple objectives when managing their forest land. Some do not use an LEV calculation to determine if they should own or sell their forest land. Some estate owners use a cashflow method of analysis to support their decision to manage pine stands on a long rotation. In some cases, the method of cashflow analysis involves the EOM where previous investments are "sunk" and when the time value of money is ignored.

LITERATURE CITED

Anonymous. 2002. Managing longleaf pine for sustained profits. Mobile, AL: Larson & McGowin Inc. 54 p.

Bilek, E.M. 1994. Forest valuation. New Zealand Journal of Forestry. 39(2): 12–13.

Busby, R.L.; Miller, J.H.; Edwards, M.B. 1998. Economics of site preparation and release treatments using herbicides in central Georgia. Southern Journal of Applied Forestry. 22: 56–162.

Caulfield, J.P.; South, D.B.; Somers, G.L. 1992. The price-size curve and planting density decisions. Southern Journal of Applied Forestry. 16(1): 24–29.

Faustmann, M. 1849. [Linnard (trans.) and Gane (ed.) 1968]. On the determination of the value which forest land and immature stands possess for forestry. English translation: Faustmann, Martin. 1968. The evolution of discounted cash flow. [Translated by W. Linnard; with editing and introduction by M. Gane.] Commonw. For. Inst. Pap. 42. Oxford, England: University of Oxford. [Translation republished with permission from Commonwealth Forestry Association in Journal of Forest Economics. 1:1 (1995).]

Handley, D.M.; Dickinson, J.C. XXXX. Uneven-aged management of southern yellow pine for improved income and ecosystem services. In: Program and abstracts of the fifteenth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-XXX. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 27–28.

Haney, H.L.; Siegel, W.C. 1993. Estate planning for forest landowners: what will become of your timberland? Gen. Tech. Rep. SO-97. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 186 p.

Helms, John A., ed. 1998. The dictionary of forestry. Bethesda, MD: The Society of American Foresters. 210 p.

Henry, J.E. 1994. Forest valuation. New Zealand Journal of Forestry. 39(1): 14–15.

Huang, C.-H.; Kronrad, G.D.; Morton, J.D. 2005. The financially optimal loblolly pine planting density and management regime for nonindustrial private forestland in east Texas. Southern Journal of Applied Forestry. 29(1): 16–21.

Ince, P.J. 1999. Faustmann and the forestry tradition of outcome-based performance measures. In: Chang, S.J., ed. Proceedings of the international symposium 150 years of the Faustmann formula: its consequences for forestry and economics in the past, present, and future. Darmstadt, Germany: IUFRO. http://www.fpl.fs.fed.us/documnts/pdf1999/ince99d.pdf. [Date accessed: June 26, 2011].

MacKay, D.G. 2001. Relating cash flow and total return: do properties with lower near-term cash flows produce higher total returns? Hancock Timber Res. Rep. R-01-3. Hancock Timber Resource Group. 5 p.

McIntyre, R.K.; Jack, S.B.; Mitchell, R.J. [and others]. 2008. Multiple value management: the Stoddard-Neel approach to ecological forestry in longleaf pine grasslands. Newton, GA: Joseph W. Jones Ecological Research Center. 36 p. http://www.jonesctr.org/education_and_outreach/publications/EFpub.pdf. [Date accessed: June 26, 2011].

Menard, R.D. 2007. An assessment of loblolly pine decline risk mapping system for managing loblolly pine decline sites within red cockaded woodpecker (RCW) habitat. Baton Rouge, LA: Louisiana State University. [Number of pages unknown]. MS thesis.

Nijnik, M. 2004. To an economist's perception on sustainability in forestry-in-transition. Forest Policy and Economics. 6: 403–413.

South African Timber Growers' Association. 1993. Annual report and financial statements for the year ended 31 March 1993. [Place of publication unknown]: [Publisher unknown]. 36 p.

Sukhdev, P. 2008. The economics of ecosystems and biodiversity: an interim report. Wesseling, Germany: Welzel+Hardt. [Number of pages unknown].

Tahvonen, O.; Viitala, E.-J. 2006. Does Faustmann rotation apply to fully regulated forests? Forest Science. 52(1): 23–30.

Viitala, E.-J. 2006. An early contribution of Martin Faustmann to natural resource economics. Journal of Forest Economics. 12(2): 131–144.

COST EFFECTIVENESS OF THREE DIFFERENT RELEASE TREATMENTS OF TABLE MOUNTAIN PINE IN A SEVERELY OVERSTOCKED AND PURE STAND

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Abstract—Table Mountain pine (*Pinus pungens* Lamb.) (TMP) is a threatened species, endemic to the Southern Appalachian Mountains. This study focuses on the release of TMP stems in an overstocked and pure TMP stand on the Cherokee National Forest in eastern Tennessee. The objective of the case study was to produce a cost analysis/comparison of releasing young TMP that are in the stem exclusion stage of stand development by several silvicultural methods: strip thinning, crop-tree release, and prescribed burning. Initial cost effectiveness of release treatments was analyzed. Regardless of treatment, costs ranged from $18 to $45 per acre. In this study, prescribed burning, generally considered more cost effective than mechanical treatments, was most expensive because of the small tract size and the labor involved to monitor the burn. The crop-tree release treatment had the least cost because small trees were cut and cost of equipment was minimal.

INTRODUCTION

Table Mountain pine (*Pinus pungens* Lamb.) (TMP) is a shade-intolerant, fire-adapted species, endemic to the Southern Appalachian Mountains. The acreage of TMP has declined because of fire exclusion policies (Waldrop and others 2006). TMP stands often originate from stand replacement fires which lead to severely overstocked conditions without silvicultural treatment or other disturbances including a fire regime. On the Cherokee National Forest (CNF), several TMP stands have developed under these conditions. The stands are overstocked (averaging 5,600 stems per acre, 1.6 inches d.b.h. after 27 years for the stand sampled in this study) with small crowns and poor vigor. These stands are in danger of stagnating and perhaps eventually succumbing before they reach optimal seed-bearing conditions. These unhealthy, overstocked stand conditions also make the stand more vulnerable to southern pine beetle (*Dendroctonus frontalis*) (SPB) infestations.

OBJECTIVES

The objective of the case study was to produce a cost analysis/comparison of releasing young TMP that are in the stem exclusion stage of stand development by several silvicultural methods: strip thinning, crop tree release, and prescribed burning. The associated economic analysis will provide land managers with baseline information on the cost effectiveness of the various release treatments in TMP.

STUDY AREA

The study was conducted in a 30-acre TMP stand at Horsehitch Gap in the southern portion of Greene County, TN, on the Nolichucky/Unaka District of the CNF. The study area is on Short Mountain, part of the Unaka Mountains located on the northeastern end of the mountain between Woolsey Gap and Horsehitch Gap (36°2'15" N, 82°46'30" W), on the Davy Crockett Lake, TN-NC quadrangle map.

Horsehitch Gap burned completely in April 1941 as a result of a brush pile fire on Paint Creek. The fire consumed approximately 2,965 acres (Sanders 1992). In 1981, a portion of this area burned once again in a stand-replacing fire, thus, creating two distinct stands—the 1941 cohort stand and the 1981 cohort stand. The 1981 fire burned a total of approximately 1,976 acres (Sanders 1992). In 2000, an SPB outbreak killed most of the TMP in the 1941 cohort and portions of the 1981 cohort. Sanders (1982) reported that the 1981 stand was beginning to show signs that it was approaching a stagnant condition in 1992, when the stand was only 11 years old. In 2001, approximately 25 acres within the 1981 cohort were killed when a fire occurred in the stand.

METHODS

A 30-acre overstocked TMP stand at Horsehitch Gap was selected to implement and investigate various release treatments. The stand was selected because of its overstocked condition, was easily accessible by road, had an existing trail at the base of the stand, and had a known recent fire history. The release treatments were a prescribed burn, strip thin, crop tree release, as well as an uncut control. A small plot size of one two-hundredth of an acre was chosen for the inventory because the stand was fairly uniform and consistent with many small diameter trees of relatively the same size (1 to 3 inches d.b.h.). Diameter, height, and number of cones for each tree on the permanent plot were recorded.

The prescribed burn treatment block was 7.3 acres and was located on the upper portion of the southern facing slope. In this block, 16 one two-hundredth-acre permanent plots were installed along transects. The prescribed burn occurred on April 22, 2008. The backing fire was ignited at 1:15 at the top of the burn block with drip torches.

The strip-thin treatment block was 4.4 acres in size and located on the westerly side of the stand. Four strips were installed downhill on the southern facing slope on November 27, 2007. The first strip was placed 32 feet east of the western boundary and was on a bearing of zero degrees. The strip was 8 feet wide. The strips were approximately 32 feet apart. Forty-eight permanent measurement plots adjacent to the thinned strips were installed in January 2008. Thirty-two plots were installed on the edge of the strips and 16 plots in the center of the strips of trees.

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The control treatment block was 3.5 acres in size and was located between the strip-thin and the crop-tree release blocks. Eight permanent plots were installed in January 2008.

The crop-tree release treatment block was 4.8 acres and was located on the eastern side of the stand. The spacing for the crop trees was approximately 45 by 45 feet, or 20 crop trees per acre. The crop trees were selected along a transect at approximately 45-foot intervals by identifying the largest diameter tree with the best overall form with a full, dominant, and symmetrical crown. The diameters (inches) and heights (feet) of each crop tree were measured. The trees were released using a crown-touching method. The diameters and heights of the cut trees were recorded. The trees were released on November 27, 2007.

Horsehitch Gap data were compiled with basal area, trees per acre, importance values of each species, and mean diameters and heights determined for each treatment. Importance values were used to compare uniformity of TMP (basal area and density) between treatment units in the stand. Analysis of variance (ANOVA) was analyzed using the importance values (TMP and hardwood) as the dependent variable and treatments as the independent variables. All of the species importance values failed to meet the equal variance assumptions, thus, the data were transformed using the natural log. This did not correct the variance issue so a rank transformation was used. The ANOVA was analyzed using the ranked values.

The cost analysis of the treatments at Horsehitch Gap was completed using methods of Miyata (1980) and Brinker and others (2002). Miyata (1980) outlines the methods for calculating operating costs. Brinker and others (2002) developed a worksheet that contains these formulas in a user-friendly layout.

At Horsehitch Gap, the strips were installed using a John Deere 450H LT dozer [http://www.deere.com/en_US/cfd/construction/deere_const/media/non_current_pdfs/noncurrent_dozers/DKA450H0209.pdf (Date accessed: August 2008)]. The time to install each strip was recorded. Purchase prices were obtained from local dealers. Percentages for fringe benefits followed Christman (2002). The life estimate, salvage value, utilization rate, repair and maintenance, interest rate, and lubrication costs were obtained from Brinker and others (2002). The tax rate also followed Brinker and others (2002). No tax costs were calculated for in-woods equipment because this equipment is not usually subject to tax collection (Brinker and others 2002). Ratings for fuel usage per hour were obtained from John Deere [http://www.deere.com/en_US/cfd/construction/deere_const/media/non_current_pdfs/noncurrent_dozers/DKA450H0209.pdf (Date accessed: August 2008)]. Off-highway diesel costs were obtained by using the U.S. Department of Energy (2008a) rate for on-highway diesel and subtracting the State and Federal taxes (University of Tennessee 2007). Wage rates were obtained from the U.S. Department of Labor (2007). Scheduled machine hours followed the methods of Miyata (1980).

Stihl MS 361 [http://www.stihlusa.com/chainsaws/MS361.html (Date accessed: August 2008)] chainsaws were used to cut trees in the crop-tree release treatment. The treatment was implemented by two sawyers. The time to walk between plots was recorded for each plot. Time to cut each competing tree on a plot was not recorded because tree size was relatively homogenous, averaging 1 to 3 inches in diameter. However, the number of trees cut was recorded. Purchase prices of chainsaws were obtained from Stihl's recommended price online [http://www.stihlusa.com/chainsaws/MS361.html (Date accessed: August 2008)]. The variables used for the chainsaw calculations were from the same sources as those used for the dozer. Hauling rates for personnel and for the dozer to be transported to the worksite were not calculated as part of the hourly rates or the cost analysis.

RESULTS

Inventory

ANOVAs were conducted to compare importance values of TMP, hardwoods, and other pines in each treatment block. The analysis showed no statistically significant difference among treatment blocks (table 1). The mean TMP importance value for the prescribed burn block was 164.30, 173.87 for the strip thin, 174.66 for the control, and 165.98 for the crop tree block, indicating no difference among treatments (table 1). Thus, importance value of TMP was fairly homogenous within treatment units of the stand.

Prescribed Burn Treatment Block—The trees in the prescribed burn block inventory (n = 16 plots) had a mean diameter of 1.9 inches and a mean height of 14.9 feet (table 2). This block contained 5,388 trees per acre with a basal area of 127.5 square feet per acre. The mean diameter of the permanent plot trees (n = 16) was 2.77 inches. TMP composed 86 percent of the treatment block, followed by 5 percent chestnut oak (*Quercus prinus* L.) (table 3). The mean importance value for TMP was 164.30, 34.14 for hardwoods, and 1.56 for the other pines (*Pinus* spp.) (table 1).

Strip-Thin Treatment Block—The mean diameter of trees in the strip-thin treatment block inventory (n = 16 plots) was 1.6 inches with a mean height of 17.7 feet (table 2). The block contained 4,242 trees per acre with a basal area of 75.7 square feet per acre. The mean diameter of the permanent plot trees (n = 48) was 2.56 inches. TMP composed 90 percent of the treatment block, followed by scarlet oak (*Q. coccinea* Münchh.) at 5 percent (table 3). The mean importance value for TMP was 173.87, 24.89 for hardwoods, and 1.25 for other pines (table 1).

Control Treatment Block—The mean diameter of trees in the control block inventory (n = 8 plots) was 1.8 inches with a mean height of 21.3 feet (table 2). The block contained 5,200 trees per acre with a basal area of 110.2 square feet per acre and had a mean permanent plot tree diameter of 3.23 (n = 8). TMP composed 70 percent of the treatment block, followed by scarlet oak at 13 percent (table 3). The mean importance value for TMP was 174.66, for hardwood 24.83, and other pines 0.51 (table 1).

Table 1—Importance values for Horsehitch Gap treatment blocks from inventory plots, Nolichucky/Unaka District, Cherokee National Forest, Greeneville, TN, January 2007

		TMP		Hardwood		Pine	
Treatment	N	Mean[a]	SD	Mean[a]	SD	Mean[a]	SD
Prescribed burn	16	164.30	57.19	34.14	54.85	1.56	3.77
Strip-thin	16	173.87	29.16	24.89	29.75	1.25	2.82
Control	8	174.66	22.94	24.83	23.13	0.51	1.43
Crop tree	5	165.98	32.66	33.38	33.33	0.65	1.44
P-value		0.9179		0.9374		0.9231	
F-value		0.17		0.14		0.16	

TMP = Table Mountain pine; SD = standard deviation.

[a] Value was obtained using the Means Procedure due to the use of ranks.

Table 2—Data for Horsehitch Gap treatment blocks, Nolichucky/Unaka District, Cherokee National Forest, Greeneville, TN, January 2007

Measurement	Crop tree	Strip-thin	Prescribed burn	Control
Mean d.b.h. (inches)	1.3	1.6	1.9	1.8
Mean basal area (square feet per acre)	79.4	75.7	127.5	110.2
Mean trees per acre	7,720	4,242	5,388	5,200
Number of plot trees	73	48	16	8
Mean height (feet) of plot trees	13.4	17.7	14.9	21.3
Mean number of cones per plot tree	8.7	5.0	5.3	6.1
Mean diameter (inches) of plot trees	2.83	2.56	2.77	3.23

Table 3—Species composition for Horsehitch Gap treatment blocks, Nolichucky/Unaka District, Cherokee National Forest, Greeneville, TN, January 2007

Species	Crop tree	Strip-thin	Prescribed burn	Control
	percent			
Table Mountain pine	86.01	90.70	86.08	69.52
Blackjack oak	8.29	0.78	1.39	0.00
Chestnut oak	1.55	2.54	5.34	3.14
Blackgum	3.63	0.10	1.86	11.00
Shortleaf pine	0.52	0.00	0.93	0.00
Scarlet oak	0.00	4.89	4.41	13.20
Hickory	0.00	0.10	0.00	0.00
Pitch pine	0.00	0.88	0.00	3.14

Crop-tree Treatment Block—The crop-tree treatment block inventory (n = 5 plots) contained trees with a mean diameter of 1.3 inches and a mean height of 13.4 feet (table 2). The block contained 7,720 trees per acre with a basal area of 79.4 square feet per acre and a mean permanent plot tree diameter of 2.83 inches (n = 73). The crop tree block was composed of 86 percent TMP and 8 percent blackjack oak (*Q. marilandica* Münchh.) (table 3). The mean importance value for TMP was 165.98, 33.38 for hardwood, and 0.65 for other pines (table 1).

Cost Comparison of Treatments

Prescribed Burn Treatment Block—The prescribed burn costs were greatly skewed because the fire burned an additional 15 acres. According to the fire management officer (FMO) for the Nolichucky/Unaka district of the CNF, the average cost of a prescribed burn is about $45.00 per acre.[2] This usually represents large burn units >200 acres. The cost per acre is generally more with smaller tract size. The prescribed burn at Horsehitch Gap was $815.37 per acre (table 4). This cost included the dozer and operator, chainsaws, 29 Forest Service employees including the Hotshots and type 2 firefighters, and 2 helicopters with pilots and copilots and fuel.

Table 4—Total costs for prescribed burn at Horsehitch Gap, Nolichucky/Unaka District, Cherokee National Forest, Greeneville, TN, April 2008

Category	Cost
	dollars
Labor	8,157.97
Fleet	
Dozer transport	90.60
Dozer	208.35
Type 2 helicopter	6,000.00
Type 3 helicopter	4,190.00
Truck	6.50
Total fleet	10,495.45
Other costs	
Fuel	75.00
Saw supplies	25.00
Total other	100.00
Total cost	18,753.42
Total cost per acre	815.37

Strip-Thin Treatment Block—Table 5 provides the hourly costs associated with the use of the dozer in the strip-thin treatment. According to John Deere [http://www.deere.com/en_US/cfd/construction/deere_const/media/non_current_pdfs/noncurrent_dozers/DKA450H0209.pdf (Date accessed: August 2008)], the purchase price of the dozer was $95,000. At an insurance rate of 1.0 percent, the fixed costs were estimated to be $1.90 per hour (table 5). The operating costs totaled $41.64 per hour with fuel costs at $2.986 per gallon

Table 5—Total costs for hourly productive time, estimation of hourly owning and operating costs of the John Deere 450H LT Dozer; Horsehitch Gap, Nolichucky/Unaka District, Cherokee National Forest, Greeneville, TN, April 2008

Category	Value
Insurance	1.0 percent
Fuel	2.0 gallons per hour
Oil and lubricants	0.74 gallons per hour
Labor	$15.62 per hour
Fringe benefits	30 percent
Scheduled operating time	2,000 hours
Utilization	25 percent
Productive time	500 hours
Fixed costs	
Taxes	N/A
Insurance	$950.00 per year
Total fixed costs	$950.00 per year
	$1.90 per hour
Operating costs	
Maintenance and repair	$30.40 per hour
Fuel and lubrication	$11.24 per hour
Total operating costs	$41.64 per hour
Labor costs	
Wages	$15.62 per hour
Fringe benefits	$4.69 per hour
Total labor costs	$20.31 per hour
Total hourly costs	$63.85 per hour

Haul rates of the dozer to and from the site location were not calculated in the hourly rate for the dozer. Estimated cost is $1.52 per mile.

[2] Personal communication. 2008. Greg Salansky, District Fire Management Officer, Cherokee National Forest, 4900 Asheville Highway SR70, Greeneville, TN 37743.

and oil and lubricants set at 36.8 percent of the fuel costs (table 5). Labor and benefits were calculated to be $20.31 per hour (U.S. Department of Labor 2007). Thus, total estimated cost was $63.85 per hour. Transportation costs for hauling the dozer from the work center location to the fieldsite was $1.52 per mile according to the CNF and was not included when calculating the hourly rate. On average, each strip was about 500 feet long and took about 40 minutes each to install. Thus, the installation of the four strips on the treatment unit cost an average of $38.70 per acre.

Crop-tree Treatment Block—Table 6 shows a breakdown of hourly chainsaw costs associated with releasing crop trees. According to Stihl [http://www.stihlusa.com/chainsaws/MS361.html (Date accessed: August 2008)], the purchase price of the MS 361 was $600. At an insurance rate of 4.0 percent, the fixed costs are $0.024 per hour (table 6). The operating costs totaled $4.58 per hour with fuel costs at $3.097 per gallon and oil and lubricants set at 36.8 percent of the fuel costs (U.S. Department of Energy 2008b) (table 6). Labor and benefits were calculated to be $21.26 per hour according to the U.S. Department of Labor. Thus, the total estimated cost was $25.87 per hour. Transportation costs for the saw were $0.485 per mile according to the Internal Revenue Service for 2007 (U.S. Department of the Treasury, Internal Revenue Service 2006).

The average number of trees cut per crop tree was 6.4. At each crop tree, the sawyers spent about 2 minutes to remove the "cut" trees and an average of 41 seconds to walk between crop trees. Thus, the total time was 3.25 hours to release 73 crop trees.

Table 7 is a comparison of costs by treatment. The prescribed burn cost $815.37 per acre. Based on the forest average for prescribed burning, the average cost is $45.00 per acre. The strip-thin was $38.70 per acre. The crop tree was $17.52 per acre.

DISCUSSION

The case study at Horsehitch Gap focused on the cost of installing each TMP release treatment. Although baseline tree measurements were tallied and calculated for each treatment, the subsequent tree response to the treatments will take several years and are beyond the scope of this study. As previously mentioned, there were no significant differences in the importance values for TMP hardwoods or the other pines among treatment blocks. TMP was uniform across the stand by treatment units (table 1). Pines other than TMP were minimal having a mean importance value ranging from 0.51 to 1.56 in the various treatment blocks (table 1).

The TMP stand at Horsehitch Gap was in the stem exclusion stage and was severely overstocked and stressed by intense competition for growing space. Mean diameters ranged from 1.3 to 3.2 inches for the permanent plot trees with stem densities of 5,600 stems per acre after 27 years (table 2). The current poor-growing condition of the stand poses many questions for future management and perpetuation

Table 6—Total costs for hourly productive time, estimation of hourly owning and operating costs of the Stihl 361 chainsaw, Horsehitch Gap, Nolichucky/Unaka District, Cherokee National Forest, Greeneville, TN, 2008

Category	Value
Insurance	4.0 percent
Fuel	0.22 per horsepower per hour
Oil and lubricants	36.8 percent
Labor	$16.35 per hour
Fringe benefits	30 percent
Scheduled operating time	2,000 hours
Utilization	50 percent
Productive time	1,000 hours
Fixed costs	
Taxes	N/A
Insurance	$24.00 per year
Total fixed costs	$24.00 per year
	$0.024 per hour
Operating costs	
Maintenance and repair	$0.48 per hour
Fuel and lubrication	$4.10 per hour
Total operating costs	$4.58 per hour
Labor costs	
Wages	$16.35 per hour
Fringe benefits	$4.91per hour
Total labor costs	$21.26 per hour
Total hourly costs	$25.87 per hour

Haul rates of the chainsaw and personnel to and from the site location were not calculated in the hourly rate for the chainsaw. Estimated cost is $0.485 per mile.

Table 7—Comparison of treatment costs at Horsehitch Gap, Nolichucky/Unaka District, Cherokee National Forest, Greeneville, TN, 2008

Treatment	Cost
	dollars per acre
Prescribed burn	
Actual	815.37
Average	45.00
Strip-thin	38.70
Crop tree	17.52

of TMP. These questions include response to release, cone production, stand development, and fuel management.

How this overstocked TMP stand will respond to release treatments is unknown. The data from the release treatments will not be available for several years. How much time is needed for a response is also unknown as these trees are in a stressed condition. A delayed growth response may occur as resources are reallocated. However, there could be little to no response due to the degree of stress the trees have encountered. Ideally, the crop trees will significantly increase in diameter and crown size, thus, providing a growth advantage over other stems.

The mean number of cones on crop trees ranged from 5.0 in the strip-thin to 8.7 in the crop tree treatment (table 2). Hopefully, the release treatments will provide additional resources to increase crown size and, thus, cone production. If the chosen trees do develop a growth and size advantage over the other stems, an increase of cone production is anticipated providing a greater seed source for the future. Differing release treatments may result in differing amounts of cone production. The crop tree treatment released the crown from all sides by removing trees whose crown touched that of the crop tree. The strip-thin treatment released only one side of the crown of the trees along the strips. In this block, 32 permanent trees were along the strips and 16 were within the remaining strips of trees. It is doubtful that those within the strips of trees will have the same results as those with any amount of crown release on the edge of the strips. The crowns of trees along the edge of the strip should increase in size and fullness after release, thus, increasing the vigor and cone production potential of the trees.

The objective of the prescribed burn was to release stems by thinning the stand, not to regenerate the stand. A slow, backing fire would be of a lower intensity that would kill some smaller diameter trees, but allow larger diameter trees to survive. Fire intensity was highly variable across the stand ranging from surface to crown burning. Cones did open and seedbed conditions were created such that some TMP regeneration or possibly a new cohort may result. However, the purpose of burning was to simulate a precommercial thinning, not to create regeneration. The fire intensity needed to provide the desired mortality is unknown at this time because of the mosaic of intensities encountered. Remeasurement of permanent plots within the burn treatment during the 2009 growing season and in the future will provide some data on the survival and mortality of TMP. The desired mortality for this stand was set in the burn plan by the FMO at 60 percent. A mortality of 60 percent would free resources and growing space for the remaining trees.

Another issue when burning this young, overstocked TMP stand was the abundance of vertical fuels. These vertical fuels played a role in the fire escaping the fire line. A very narrow window of opportunity exists for prescribed burning in TMP stands because of the rough, steep terrain and the vertical fuels in stands that have not been previously burned. Fuel moisture, windspeed, and wind direction are factors that limit prescribed burning in these stands. The assumption with using prescribed burning as a release treatment is that some trees will succumb while other trees will survive. However, burning conditions to provide this assumption are unknown. Most studies of burning in TMP focus on regeneration rather than thinning. To determine which intensity is needed for desired amounts of mortality, fire intensity studies should be implemented in similar stands of TMP. By monitoring fire intensities and mortality in these overstocked stands, the fire intensity that best achieves the optimal amount of mortality while providing release of other trees in overstocked conditions may be determined.

The costs associated with prescribed burning were greater than the mechanical treatments used in this study (table 7). Typically, the costs for prescribed burning decrease on a per-acre basis as the area burned increases. A few considerations that increase the costs of prescribed burning in this study area are the installation of fire lines, adequate labor to monitor the prescribed burn, and the steep and rocky terrain.

The initial costs show that crop tree release had the least costs of all the release treatments at $17.52 per acre. Equipment costs were minimal (table 7). This cost would probably increase as average tree size increased. The trees in this study were very small and many could be removed in a matter of minutes. The primary cost associated with this treatment was labor. The strip-thin was $38.70 per acre with operating and labor costs. The cost of diesel, including off-road diesel, is continuing to rise, so the operating costs will probably increase. With the average cost of prescribed burning at $45.00 per acre, the majority of the costs would be related to labor. However, in this case, equipment costs were a major contributor considering two helicopters and a dozer were used during the prescribed fire treatment.

The response of TMP to these release treatments will not be known for several years. This study provides an estimate of initial costs of the release treatments with the crop tree release being the least initial cost and prescribed burning being the most. Cost figures will vary depending on size of treatment area as well as how many trees are released in the crop-tree release treatments. Strip treatment costs will also vary depending on the number and size of strips. In this study, strips were 8 feet wide with 32 feet between strips. Larger (or smaller) strips and area left between strips can vary depending on management or operation objectives, thus, adding to or decreasing treatment costs. Future measurements or studies may provide more information on the tree response to the release treatments installed in this study. Hopefully, these treatments are viable options for the release of TMP stands to encourage future growth, survival, and cone production.

CONCLUSION

Prescribed burning has been thought to be the most effective tool for managing these stands. This project investigates the cost effectiveness of other mechanical treatments that have proven successful as potential release methods in

silvicultural operations. This study implemented several release treatments for future growth considerations. The permanent plots in this study should be remeasured periodically to determine growth response from each treatment. The initial cost of the release treatments should be valuable in determining the future cost efficiency based on growth response rate. In typical situations where land managers are able to use prescribed fire across numerous acres, the cost of $45.00 per acre is reasonable. The crop-tree release treatment was very labor intensive, but with a cost of $17.52 per acre, it appears to be another viable option with small-diameter stems. The strip-thin treatment may be more difficult to apply in some areas due to the terrain and possible safety issues. After reassessing tree growth following these treatments, more information can be evaluated concerning the best treatment for the investment. Although the wood products value of TMP is limited due to markets, inaccessibility, and terrain, this unusual, endemic species provides many ecosystem and diversity aspects to forests in the region.

Many stand development questions about TMP are beyond the scope of this study, but can possibly be addressed in future studies. A plethora of TMP research is available and ongoing about the regeneration and stand dynamics of TMP (Armbrister 2002, DeWeese 2007, Mohr and others 2002, Randles and others 2002, Van Lear 2000, Waldrop and Brose 1999, Waldrop and others 2006). More research is needed to fully understand the natural development of TMP and management concerns to create favorable environmental conditions for the species.

TMP is an endemic species in the Southern Appalachian Mountains and a species of concern. An active management program that includes the use of fire is necessary in maintaining healthy communities of TMP. Management actions should be taken to ensure the regeneration of the species and to promote stand development of TMP trees to maturity. This study provides information on the costs of implementing various release treatments to encourage further growth and development of the species.

LITERATURE CITED

Armbrister, M.R. 2002. Changes in fire regimes and the successional status of Table Mountain pine (*Pinus pungens* Lamb.) in the Southern Appalachians, USA. Knoxville, TN: University of Tennessee. 151 p. M.S. thesis.

Brinker, R.W.; Kinard, J.; Rummer, B.; Lanford, B. 2002. Machine rates for selected forest harvesting machines. 2nd ed. Circ. 296. Auburn, AL: Auburn University, Alabama Agricultural Experiment Station. 32 p.

Christman, C.S. 2002. Harvesting small diameter hardwoods on steep slopes: evaluating costs and wood utilization. Knoxville, TN: University of Tennessee. 99 p. M.S. thesis.

DeWeese, G.G. 2007. Past fire regimes of Table Mountain pine (*Pinus pungens* Lamb.) stands in the central Appalachian Mountains, Virginia, U.S.A. Knoxville, TN: University of Tennessee. 308 p. Ph.D. dissertation.

Miyata, E.S. 1980. Determining fixed and operating costs of logging equipment. Gen. Tech. Rep. NC–55. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 18 p.

Mohr, H.H.; Waldrop, T.A.; Shelburne, V.B. 2002. Optimal seedbed requirements for regenerating Table Mountain pine. In: Outcalt, Kenneth 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: 306-309.

Randles, R.B.; Van Lear, D.H.; Waldrop, T.A.; Simon, D.M. 2002. Periodic burning in Table Mountain-pine pine stands. In: Outcalt, Kenneth 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: 114–117.

Sanders, G.L. 1992. The role of fire in the regeneration of Table Mountain pine in the Southern Appalachian Mountains. Knoxville, TN: University of Tennessee. 125 p. M.S. thesis.

University of Tennessee. 2007. IRS rules for diesel fuel and gasoline purchases. Tech. Bull., Municipal Tech. Advisory Serv. Knoxville, TN. [Not paged]. http://www.mtas.tennessee.edu/public/web.nsf/Web/Read+Pubs. [Date accessed: August 2008].

U.S. Department of Energy. 2008a. U.S. on-highway diesel fuel prices. http://tonto.eia.doe.gov/oog/info/wohdp/diesel.asp [Date accessed: November 2008].

U.S. Department of Energy. 2008b. U.S. regular weekly retail. http://www.eia.doe.gov/oil_gas/petroleum/data_publications/wrgp/mogas_history.html [Date accessed: November 2008].

U.S. Department of Labor. 2007. May 2007 national industry–specific occupational employment and wage estimates NAICS 113300-logging. http://www.bls.gov/oes/current/naics4_113300.htm. [Date accessed: September 16, 2008].

U.S. Department of the Treasury, Internal Revenue Service. 2006. IRS announces 2007 standard mileage rates. IR–2006–168. http://www.irs.gov/newsroom/article/0,,id=163828,00.html. [Date accessed: September 2008].

Van Lear, D.H. 2000. Recent advances in the silvicultural use of prescribed fire. In: Moser, W. Keith; Moser, Cynthia F., eds. Fire and forest ecology: innovative silviculture and vegetation management. Tall Timbers fire ecology conference Proceedings, no. 21. Tallahassee, FL: Tall Timbers Research Station: 183–189.

Waldrop, T.A.; Brose, P.H. 1999. A comparison of fire intensity levels for stand replacement of Table Mountain pine (*Pinus pungens* Lamb.). Forest Ecology and Management. 113(2): 155–166.

Waldrop, T.A.; Mohr, H.H.; Brose, P.H. 2006. Early dynamics of Table Mountain pine stands following stand-replacement prescribed fires of varying intensity. In: Conner, Kristina 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: 471–474.

FINANCIAL PERFORMANCE OF LOBLOLLY AND LONGLEAF PINE PLANTATIONS

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Abstract—The financial performance of selected management regimes for loblolly (*Pinus taeda* L.) and longleaf pine (*P. palustris* Mill.) plantations were compared for four cases, each with low- and high-site productivity levels and each evaluated using 5 and 7 percent real discount rates. In all cases, longleaf pine was considered both with and without pine straw harvesting. The analysis also examined the financial impacts of various sawtimber proportions valued as poles. With the addition of pine straw revenues, longleaf management can yield returns that are comparable to typical loblolly pine regimes (–16 to +3 percent, depending on site quality and discount rate). With as little as 25 percent of sawtimber considered poles, longleaf pine financially outperformed loblolly at all site and discount rate combinations. Results indicate that longleaf pine may be an attractive alternative for some landowners, due to both lower establishment costs and the favorable land expectation value comparisons.

INTRODUCTION

The past decade has seen significant shifts in timberland ownership, particularly in the Southern United States. Integrated forest product companies have sold many of their land assets, which have subsequently been acquired by institutional investors. The reasons behind corporate land sales are diverse, but investors are attracted to timberland for several key reasons, including strong historical risk-adjusted returns (Binkley and others 2001, Carroll 2003, Caulfield 1999), low correlation with other asset classes (Binkley and others 2001, Carroll 2003), and an apparent correlation with inflation (Clutter and others 2005). The interest in timberland investment is apparent by the inflow of capital into the sector, with approximately $2 billion invested annually over the past decade (Clutter and others 2005). Timberland investments are often made by timberland investment management organizations (TIMO), who both acquire and manage property on the behalf of institutional investors.

Many TIMOs function as closed-end funds, meaning a key aspect of TIMO management is a short-time horizon relative to integrated forest products companies. While forest product companies have traditionally held land forever, TIMOs are organized with a broader set of expected land tenures and management foci. In general, TIMOs are more focused on financial returns over the length of the investment, while forest products companies traditionally concentrated on wood supply, i.e., harvest volume, and environmental objectives (Clutter and others 2005). Many TIMOs plan to hold land for no more than 10 to 15 years (closed-end funds), but others intend to hold forest land forever. In all cases, the justification for forest management activities undertaken by TIMOs is higher returns for investors; many TIMOs focus on intensive, short-rotation silviculture but not all. With the proliferation of TIMOs and timberland investors has come differentiation, including TIMOs with an emphasis on natural regeneration, high-yield plantations to offset losses to natural forests throughout the World, or other objectives.

Along with shifts in forest ownership, the past decade has also seen increased interest in longleaf pine (*Pinus palustris* Mill.) management. Longleaf pine once dominated forests from Virginia to Texas, but overexploitation resulted in its widespread decline. In recent years, various organizations have begun encouraging longleaf plantation establishment with much of their effort directed towards private landowners whose objectives include factors such as wildlife habitat and aesthetics in addition to economics.

Little work has been done examining the economic viability of longleaf pine management on investment properties. This can be attributed to the commonly held belief that returns from longleaf management cannot compare to those from loblolly pine (*P. taeda* L.) plantations. Traditionally, longleaf has been a difficult species to plant and successfully establish (Johnson 2008). The persistent and variable grass stage translated into longer rotation lengths (Johnson 2008) and hampered planning efforts. As a result, longleaf was often relegated to poor sites, only perpetuating its reputation for slow growth (Johnson 2008). Improvements in nursery techniques and silvicultural practices, however, challenge these old assumptions (Johnson 2008), such that longleaf and loblolly plantation economics may compare more favorably than previously believed. TIMOs may be able to justify investments in longleaf pine plantations if they can show returns comparable to those from intensive loblolly pine management. This is particularly true given the higher amenity values attributed to longleaf pine.

The remainder of this paper focuses on a detailed economic comparison of longleaf and loblolly pine plantation management. Much of the literature about longleaf pine focuses on a diversity of management objectives, including wildlife and endangered species. This analysis differs in that the focus is solely on the economics of plantation management. While longleaf pine forests may provide additional wildlife or aesthetic benefits, this analysis ignores such amenity values.

METHODOLOGY

Selected Cases

The financial performance of selected loblolly and longleaf pine plantation management regimes were compared for

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Table 1—Selected case/species combinations

Case	Species	Site index	Discount rate	Straw harvest
		feet	*percent*	
1	Loblolly (LP)	60	5	No
	Longleaf (LL)	85	5	No
	Longleaf (LL-S)	85	5	Yes
2	Loblolly (LP)	60	7	No
	Longleaf (LL)	85	7	No
	Longleaf (LL-S)	85	7	Yes
3	Loblolly (LP)	80	5	No
	Longleaf (LL)	110	5	No
	Longleaf (LL-S)	110	5	Yes
4	Loblolly (LP)	80	7	No
	Longleaf (LL)	110	7	No
	Longleaf (LL-S)	110	7	Yes

four cases, outlined in table 1. Loblolly and longleaf pine plantations were projected with low- and high-site productivity levels. For the comparison, the loblolly pine site index values (base age 25 years) were converted to equivalent site index values for longleaf pine (base age 50 years). Discounted cashflows were generated using 5 and 7 percent real discount rates. In all cases, longleaf pine was considered both with and without pine straw harvesting. Pine straw harvesting can add substantially to overall returns from a given rotation (Johnson 2008), making its inclusion an important consideration.

Management Regimes

Management regimes were selected from a reduced set of acceptable alternatives, which were constrained by management intensity and treatment timing. Reforestation activities follow those commonly used in loblolly and longleaf pine on suitable sites in South Carolina. Planting density and first-year survival were assumed to be identical for both species. Midrotation treatment timings were restricted to ranges considered biologically reasonable and commercially feasible, with management intensities based on commonly implemented rates. The regimes that maximized land expectation value (LEV) for each site/discount rate combination were chosen for further analysis. LEV is the present value per acre of the projected costs and revenues from an infinite series of identical rotations starting from bare ground, and may also be referred to as bare land value or soil expectation value.

Loblolly Pine—Loblolly pine plantations were projected using the Forest Nutrition Cooperative Decision Support System (LobDSS) (Amateis and others 2005). LobDSS interfaces with FASTLOB2, a whole stand growth-and-yield model developed by the Loblolly Pine Growth and Yield Research Cooperative at Virginia Tech. This model provides options for evaluating thinning and/or midrotation fertilization treatments (Amateis and others 2001). The effects of site conditions, site preparation, and first-year silvicultural treatments on loblolly pine plantation survival and growth are also modeled.

The LobDSS optimization routine was used for evaluating the impacts of midrotation fertilization and thinning timing and rotation length on economic valuation. Searches were constrained to one thinning treatment between ages 12 and 20 years, with a midrotation fertilization 1 year postthin. The thinning treatment removed every fifth row combined with thinning from below to 75 square feet per acre residual basal area. Thinned plantations were fertilized with urea at a rate of 200 pounds of nitrogen per acre.

Longleaf Pine—The FORSim Longleaf Pine Growth Simulator (LPGS) was used for projecting longleaf pine plantations (FORSight Resources 2007a, 2007b). LPGS serves as an interface to the longleaf pine growth engine, a stand-level model that simulates longleaf pine survival and growth, including the ability to simulate up to five thinning treatments during stand development.

Longleaf pine management regimes with either one or two thinning treatments were considered. Two thinning regimes, common in the generally longer longleaf rotations, were included in this analysis. The first thin was a 1-in-5 row per thin from below combination to 80 square feet per acre residual basal area. The second thin was from below to 70

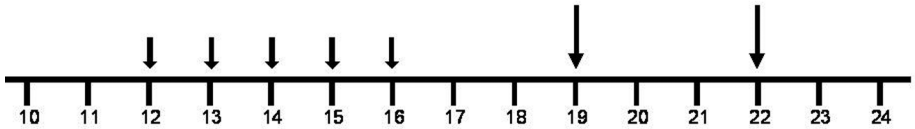

Figure 1—Time line illustration of a typical pine straw-raking cycle. Short arrows indicate enhancement activities and longer arrows indicate pine straw rakings.

square feet per acre residual basal area. The operational window for the first thin was between ages 15 and 30 years, and a second thin was considered after age 19 years. A minimum 4 years was required between thinning entries. The final harvest was allowed no earlier than 4 years after the final thinning treatment. Based on the level of site preparation, the number of years for the plantation to emerge from the grass stage and reach breast height (4.5 feet) was set at 3 years.

In addition to thinning treatments, longleaf pine management regimes with and without pine straw-raking cycles were examined. A pine straw-raking cycle included a 4-year enhancement period followed by a series of rakings (see fig. 1). The operational window for cycle commencement was limited to biologically feasible age ranges; ages 12 to 15 years for the first cycle and 23 to 32 years for the second. Thinning activities were not allowed during a raking cycle, and cycles could not commence until 2 years postthin.

Financial Analysis

Harvest volumes were merchandized using the product specifications shown in table 2. All harvested volumes were measured in tons. Site preparation, planting, and midrotation treatments are outlined in table 3, including treatment timing, application rate, and associated cost. All site preparation and planting activities are assumed to occur during the same year. Net revenues from longleaf pine straw harvest are based on typical contracts with local straw producers. Product prices and management costs used in the analysis were limited to those values available from published, verifiable sources.

LEV and present net worth (PNW) for the first rotation were calculated for each selected management regime using both 5 and 7 percent real discount rates. Because loblolly and longleaf rotation lengths differ, LEV provides the only means for directly comparing results. PNW provides a means for analyzing cashflows over the short term. Product prices and treatment costs and revenues were applied as outlined in tables 2 and 3. In all cases, activities were assumed to occur at the start of each year, i.e., January 1.

RESULTS

Selected Regimes

Table 4 shows the chosen regime for each case. The associated PNW for the first rotation and LEV are shown in table 5. Harvest removals for each case are reported in table 6. The percent-product recovery from all harvest operations and cumulative PNW over stand age by case/species combination are shown in figures 2, 3, and 4, respectively. Species-level results are analyzed in the following sections.

Table 2—Product specifications and stumpage prices

Product	Min. d.b.h.	Top diameter inside bark	Price
	---------- *inches* ----------		*dollars per ton*
Pulpwood	5	3	8.05
Chip-n-saw	9	5	18.98
Sawtimber	12	8	36.82

Loblolly Pine

Discount rate had no effect on thinning age but did result in an earlier final harvest (table 4). A comparison of case 1 to case 3 (and case 2 to case 4) indicates that reductions in both thinning and final harvest ages were associated with higher site. Figures 2 and 3 reveal that changes in site quality had little influence on product proportions removed from stands; sawable volume (sawtimber and chip-n-saw) is about 65 percent in all cases. However, with a 5-percent discount rate, the optimum rotation produced about 50 percent sawtimber, while only 35 percent sawtimber is produced using a 7-percent discount rate. The lowest ($212.75 per acre) and highest ($1,179.70 per acre) LEVs were associated with cases 2 and 3, respectively.

Longleaf Pine without Pine Straw

In all cases, regimes for longleaf pine plantations without pine straw harvests incorporated a single thinning treatment, and although the longleaf final harvests occurred slightly later than in loblolly, the economic rotations were shorter than those typically associated with longleaf management. This is expected since this study considers only economic factors and does not consider the amenity values that are often an important consideration elsewhere.

As with loblolly plantations, site quality was the driving factor behind thinning age. On the poorer site, longleaf plantations produced no sawtimber, but on the higher site sawtimber yield increased to 26 to 35 percent of total removal volume. As was the case for loblolly, total sawable volume (sawtimber and chip-n-saw) was about 65 percent in all cases. The lowest

Table 3—Costs and revenues for loblolly and longleaf pine treatment regimes

Description	Value
	dollars per acre
Loblolly pine	
Chemical hardwood control (CHEM) @ establishment	100.00
Hand plant @ 622 TPA with 95 percent first-year survival	81.10[a]
Herbaceous weed control (HWC) @ year 1	57.50
Establishment fertilization with 250 pounds DAP per acre @ year 1	47.50
Midrotation fertilization with urea @ 200 pounds nitrogen per acre	47.50
Longleaf pine	
Chemical hardwood control (CHEM) @ establishment	100.00
Broadcast burn @ establishment	15.00[b]
Hand plant @ 622 TPA with 95 percent first-year survival	102.87[a]
Pine straw harvest net revenue per raking	150.00[c]
Annual costs/revenue	
Management fee costs	5.00
Hunting lease revenues	7.00

TPA = trees per acre; DAP = diammonium phosphate.

[a] Hand planting—Georgia Forestry Commission (2008) and seedling cost—South Carolina Forestry Commission (2008).

[b] South Carolina Forestry Commission (2008).

[c] North Carolina Cooperative Extension Service (1995).

Table 4—Silvicultural treatment regimes by case/species

Case (discount rate)	Species	#1 Straw cycle[a]	#1 Thin/fert age	#2 Straw cycle[a]	#2 Thin/fert age	Final harvest
		years				
1	Loblolly (LP)	—	15/16	—	—	35
(5 percent)	Longleaf (LL)	—	25/—	—	—	33
	Longleaf (LL-S)	12	23/—	25	38/—	52
2	Loblolly (LP)	—	15/16	—	—	28
(7 percent)	Longleaf (LL)	—	25/—	—	—	33
	Longleaf (LL-S)	12	23/—	25	38/—	52
3	Loblolly (LP)	—	12/13	—	—	29
(5 percent)	Longleaf (LL)	—	23/—	—	—	32
	Longleaf (LL-S)	12	23/—	25	38/—	42
4	Loblolly (LP)	—	12/13	—	—	24
(7 percent)	Longleaf (LL)	—	23/—	—	—	27
	Longleaf (LL-S)	12	23/—	—	—	27

— = not applicable; Thin/fert = thinning and fertilization.

[a] Indicates age at which pine straw cycle begins.

Table 5—Present net worth and land expectation value by case/species

Case (discount rate)	Species	Present net worth (1st rotation)	Land expectation value
		---------- *dollars per acre* ----------	
1	Loblolly (LP)	501.93	610.64
(5 percent)	Longleaf (LL)	245.23	307.83
	Longleaf (LL-S)	548.02	592.91
2	Loblolly (LP)	182.75	212.75
(7 percent)	Longleaf (LL)	49.25	53.37
	Longleaf (LL-S)	174.82	178.11
3	Loblolly (LP)	895.10	1,179.70
(5 percent)	Longleaf (LL)	765.82	966.69
	Longleaf (LL-S)	968.24	1,109.14
4	Loblolly (LP)	469.74	582.60
(7 percent)	Longleaf (LL)	385.42	456.96
	Longleaf (LL-S)	503.31	597.46

Table 6—Thinning and final harvest volume removals by case/species

Case (discount rate)	Species	1st thin	2nd thin	Final harvest	Total
		---------- *tons per acre* ----------			
1	Loblolly (LP)	25.80	—	132.30	158.10
(5 percent)	Longleaf (LL)	27.27	—	108.08	135.35
	Longleaf (LL-S)	15.43	55.23	120.21	190.88
2	Loblolly (LP)	25.80	—	104.40	130.20
(7 percent)	Longleaf (LL)	27.27	—	108.08	135.35
	Longleaf (LL-S)	15.43	55.23	120.21	190.88
3	Loblolly (LP)	31.90	—	158.30	190.20
(5 percent)	Longleaf (LL)	64.13	—	135.78	199.91
	Longleaf (LL-S)	56.16	51.50	126.04	233.70
4	Loblolly (LP)	31.90	—	135.10	167.00
(7 percent)	Longleaf (LL)	64.13	—	116.84	180.97
	Longleaf (LL-S)	56.16	—	116.64	172.80

— = not applicable.

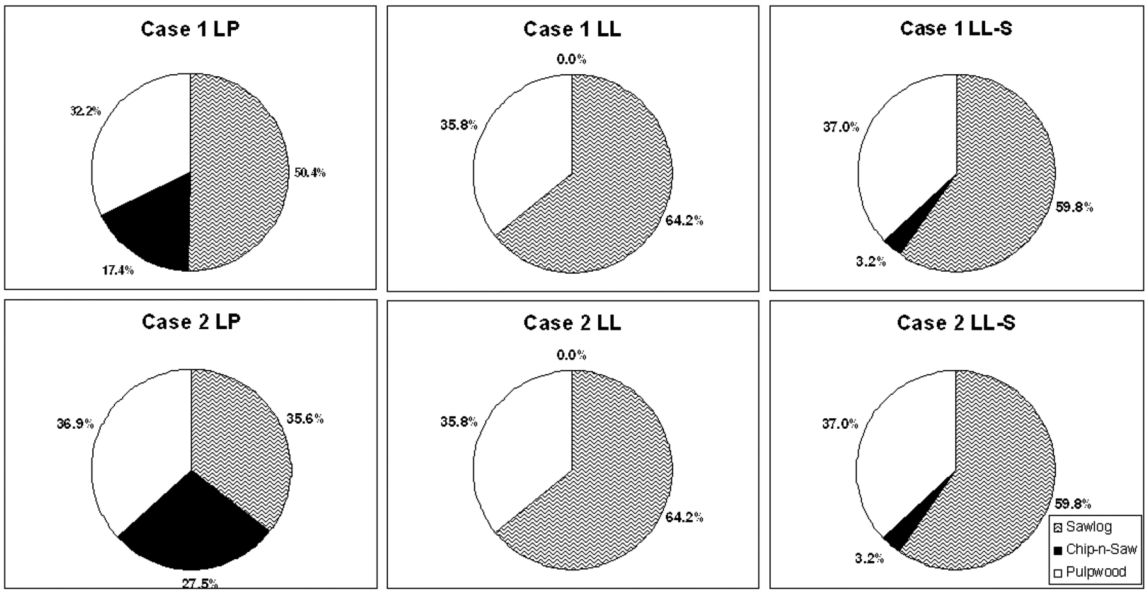

Figure 2—Percent product recovery, cases 1 and 2 (all species and regimes).

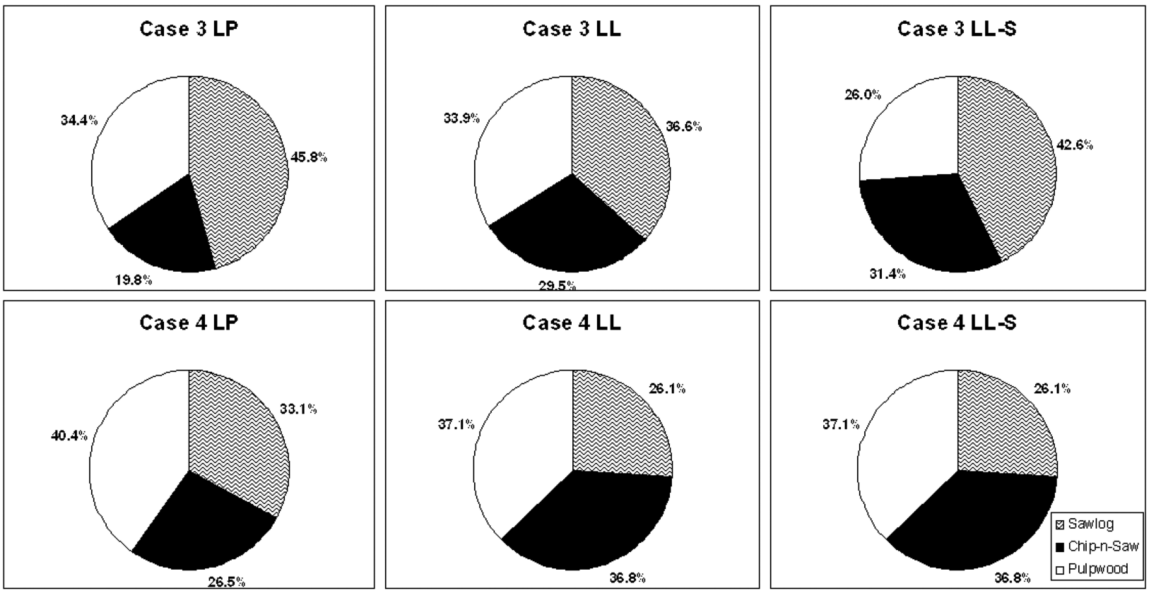

Figure 3—Percent product recovery, cases 3 and 4 (all species and regimes).

($53.37 per acre) and highest ($966.69 per acre) LEVs were associated with cases 2 and 3, respectively (table 5).

Longleaf Pine with Pine Straw

For all cases, adding pine straw harvests greatly improved the financial performance of longleaf pine plantations. Management regimes included two thins and two pine straw-raking cycles in all cases except case 4. The first pine straw-raking cycle began at the earliest feasible age in all cases, indicating the importance of early revenues to overall net present worth. Regimes were identical for the first three cases, indicating that pine straw revenues dominated the economic impacts of site and discount rate. The carrying costs of holding the stand long enough to produce a second thin and straw raking exceeded the increased revenues attributed to the longer rotation with the higher discount rate used in case 4. Site quality had the largest impact on product recovery percentages. Virtually all of the sawable material produced on the low site was sawtimber, with a more

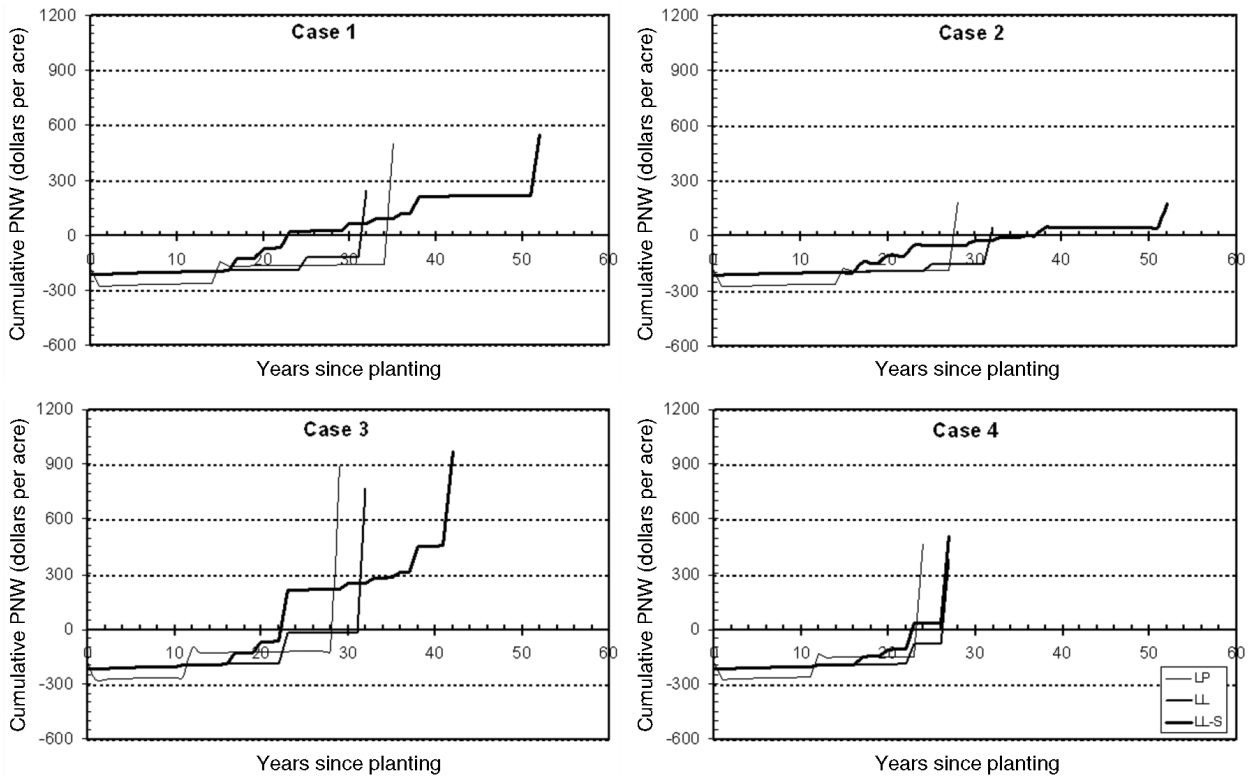

Figure 4—Cumulative present net worth by stand age for cases 1 through 4.

even breakdown among products on the high site. On the high site, more sawable wood (sawtimber and chip-n-saw) was produced than either loblolly or longleaf without pine straw raking (75 percent sawable compared to 65 percent). Minimum ($178.11 per acre) and maximum ($1,109.14 per acre) LEVs were associated with cases 2 and 3, respectively (table 5).

Species Comparison

The addition of pine straw raking to longleaf pine management regimes resulted in improved financial results (13 to 70 percent higher than without pine straw raking) that compared favorably with the loblolly pine management regimes (table 5). The loblolly regimes produced LEV values 3 to 16 percent higher than longleaf in all cases except case 4, where longleaf exceeded the corresponding loblolly LEV by 2.6 percent.

An examination of the cashflows in figure 4 reveals that the cumulative PNW (dollar per acre) from loblolly pine plantations remained negative until the final harvest in all cases. However, pine straw harvests yield positive cashflows earlier in the rotation, especially for longleaf pine plantations on lower sites and evaluated using lower discount rates. In terms of product recovery percentages, longleaf pine plantations with pine straw harvests and longer rotations produced a higher percentage of sawable wood compared to short rotations for loblolly pine plantations (figs. 2 and 3).

Research reported at Auburn University indicates greater pole production in 39-year-old longleaf stands (72 percent) than in loblolly pine stands (<8 percent) of the same age. Figure 5 illustrates the impacts of pole production on LEVs using the management regimes for LL-S in table 4. With as little as 25 percent of the sawtimber volume classified as poles, longleaf pine with straw raking financially outperforms loblolly at all site index and discount rate combinations. Note that the regimes outlined in table 4 may produce suboptimal LEVs when pole production is considered. As a result, the comparison may be even more favorable than indicated in figure 5.

DISCUSSION

Results indicate that longleaf pine regimes that do not incorporate pine straw raking yield financial results that are inferior to those from intensive loblolly management. With the addition of pine straw revenues, however, longleaf management can yield returns that are comparable to typical loblolly pine regimes (–16 to +3 percent, depending on site quality and discount rate). In fact, longleaf pine plantations with pine straw harvests produced greater LEVs than loblolly plantations on lands with higher site index (80 and 110 feet for loblolly pine and longleaf pine, respectively) when using the higher discount rate (7 percent). Other longleaf pine management regimes produced lower but comparable financial performance. Furthermore, figure 5 indicates that there may be additional upside potential for longleaf when pole production is considered.

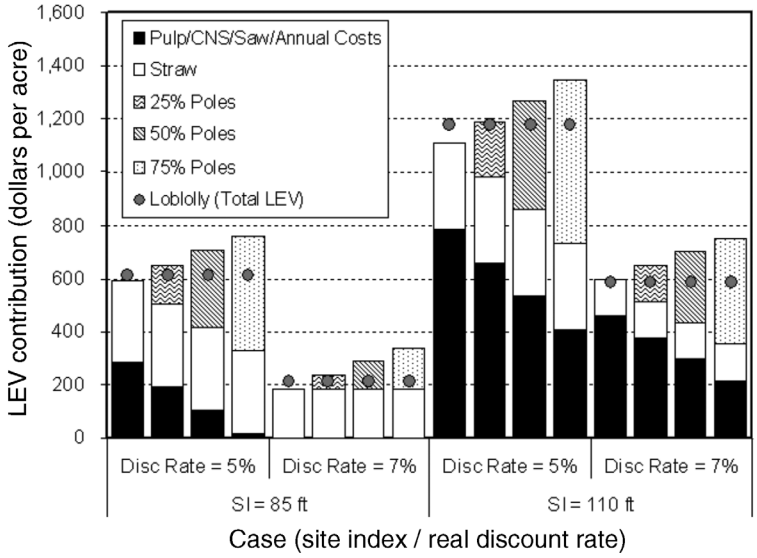

Figure 5—Land expectation value (LEV) for longleaf pine plantations with varying proportions of sawtimber considered eligible for poles.

It should be noted that LobDSS downgrades a portion of loblolly sawtimber trees into the pulpwood class, a behavior absent from LPGS. This may have caused an elevated sawtimber proportion in the longleaf product recovery. In addition, while this study assumes equal seedling survival between longleaf and loblolly pine, longleaf seedling survival may in fact be lower.

Several factors seem to indicate that longleaf may perform even better than indicated in this analysis. Citing a report by John Guthrie and Son's, Inc., Johnson (2008) points to evidence that average timber sale prices were 10 to 20 percent higher when species composition was primarily longleaf pine. Given that longleaf regimes consistently produce higher total yield (table 6), this may lead to even more favorable comparisons between the species. A longleaf price premium was not included in this analysis due to a lack of widespread, documented evidence that such a premium exists. In addition, some authors have indicated that current site preparation practices may reduce time to leave grass stage below the 3 years used in this paper (Johnson 2008). Reducing the length of time spent in the grass stage shortens the overall rotation length with commensurate improvements in PNW and LEV.

At lower discount rates longleaf pine regimes with pine straw raking provided positive cashflows sooner than loblolly (figs. 3 and 4). In all cases, however, positive cashflows were not achieved with any regime until after age 23. This result is noteworthy because this may be longer than the land tenure of closed-end funded TIMO ownerships. Because there is likely to be little to no direct return on reforestation investments under these short land tenures, a logical consequence may be the minimization of reforestation expenses. Thus, longleaf pine may be a more attractive alternative, given the 25-percent lower initial investment ($217 per acre vs. $286 per acre for loblolly) and the favorable LEV comparison.

LITERATURE CITED

Amateis, R.L.; Allen, H.L.; Montes, C.R.; Fox, T.R. 2005. Overview of the Forest Nutrition Cooperative's Silvicultural Decision Support System. Raleigh, NC: North Caroline State University; Blacksburg, VA: Virginia Polytechnic Institute and State University Forest Nutrition Cooperative. 11 p.

Amateis, R.L.; Burkhart, H.E.; Allen, H.L.; Montes, C.R. 2001. FASTLOB: fertilized and selectively thinned loblolly pine plantations (a stand-level growth and yield model for fertilized and thinned loblolly pine plantations). Loblolly Pine Growth and Yield Cooperative. Blacksburg, VA: Virginia Polytechnic Institute and State University. 32 p.

Binkley, C.; Washburn, C.; Aronow, M.E. 2001. Timberland as a portfolio diversifier. Res. Notes 2001. [Boston]: Hancock Timber Resource Group. [Number of pages unknown].

Caulfield, J. 1999. Dealing with timberland investment risk: theory versus practice for institutional owners. Journal of Forest Economics. 5(2): 253–268.

Carroll, B. 2003. Applying modern portfolio theory to timberland allocation. FORSight Resources White Paper. http://www.forsightresources.com/library/MillsStiff2008.pdf. [Date accessed: June 27, 2011].

Clutter, M.; Mendell, B.; Newmand, D. [and others]. 2005. Strategic factors driving timberland ownership changes in the U.S. South. Preliminary Report to the Southern Group of State Foresters. http://www.srs.fs.usda.gov/econ/pubs/southernmarkets/strategic-factors-and- ownership-v1.pdf. [Date accessed: June 27, 2011].

FORSight Resources. 2007a. FORSim longleaf pine growth simulator (LPGS). The FORSight Resource. 4(4): 2.

FORSight Resources. 2007b. FORSim LPGS technical sheet. FORSight Resources. 1 p. http://www.FORSightResources.com/techsheets/ForSimLPGS.pdf. [Date accessed: June 27, 2011].

Georgia Forestry Commission. 2008. Southern pine beetle (SPB) cost share program: cost share rates 2008. http://www.gfc.state.ga.us/ForestManagement/spb.cfm. [Date accessed: June 27, 2011].

Johnson, R. 2008. Does longleaf make dollars and sense? The Longleaf Alliance. http://www.auburn.edu/academic/forestry_wildlife/longleafalliance/landowners/longleafeconomics/dollarandsensearticle.pdf. [Date accessed: June 27, 2011].

North Carolina Cooperative Extension Service. 1995. Producing longleaf pine straw. Woodland Owner Notes 18. Raleigh, NC. [Number of pages unknown].

South Carolina Forestry Commission. 2008. Price guide for tree seedlings, equipment, and services. http://www.state.sc.us/forest/refprice.htm#services. [Date accessed: June 27, 2011].

ESTIMATING ANNUAL GROWTH LOSSES FROM DROUGHT IN LOBLOLLY PINE PLANTATIONS

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Abstract—Growth data over the past 10 years from loblolly pine (*Pinus taeda* L.) plantations established across the natural range of the species were linked with annual rainfall data over the same period to evaluate the impact of drought on stand growth. Regression procedures were used to determine (1) whether dominant height growth or basal area growth or perhaps both were correlated with local rainfall, (2) whether annual rainfall was a significant predictor of annual growth in the presence of other stand and site variables, and (3) whether prediction equations could be developed that would provide reasonable estimates of growth loss during periods of drought. Results show that basal area growth but not dominant height growth is correlated with rainfall for these data. A prediction equation was developed that can be used to estimate the effect of rainfall on annual basal area growth. The equation should be useful for forest managers needing to estimate basal area and volume growth losses due to drought conditions in loblolly pine plantations.

INTRODUCTION

It has long been known that the amount and distribution of annual precipitation affects tree growth. In one of the earliest studies relating precipitation to the growth of pine (*Pinus* spp.) in the South, Coile (1936) demonstrated graphically the correlation of diameter growth of four southern pine species with annual rainfall. Bassett (1964) used tree growth and daily precipitation data acquired at the Crossett Experimental Forest in southeast Arkansas to correlate seasonal growth with rainfall. He divided the growing season into "growth days" and "no-growth" days, based on rainfall and potential evaporation, and developed a growth index to quantify the relationship of basal area and volume growth to precipitation. Langdon and Trousdell (1979) examined these data further and found large differences in annual basal area and volume growth between years characterized by a dearth and glut of precipitation. Jacobi and Tainter (1988) used the methods of dendrochronology to evaluate annual ring growth in loblolly pine (*P. taeda* L.) stressed by drought on the Piedmont of South Carolina. They found significant annual loss of radial growth in loblolly pine stands during periodic drought years from the 1950s through the severe drought of 1980. Both Wheeler and others (1982) and Yeiser and Burnett (1982) documented the loss of volume growth and mortality during the severe drought of August 1979 through December 1980 in Arkansas.

While the importance of precipitation on loblolly pine growth is well known and documented, most growth-and-yield models do not explicitly account for the effects of precipitation. The major reasons for this are: (1) total precipitation across the loblolly range varies widely both spatially and temporally making it difficult to estimate precipitation at any given site for any given year and (2) the distribution through the year of the total precipitation will have an effect on tree growth and is also difficult to predict.

Given these obstacles, developers of growth-and-yield models have generally assumed that the variation in growth associated with precipitation averages out over the rotation toward some mean effect that is confounded with other site factors and included in the overall site index value for a stand. For long-term projections this may be a reasonable assumption. However, when short-term projections are needed for inventory updating or immediate planning for wood supply, the variation in precipitation can result in over- or underprediction of volume yields, especially during seasons of unusual drought.

The purpose of this study was to evaluate the impact of annual precipitation on the growth of loblolly pine plantations across the region and develop a model that can be used to adjust growth-and-yield estimates for losses due to drought.

DATA

Dominant height and basal area measurements from permanent plots established in 3- to 8-year-old intensively managed loblolly pine plantations (IMP) across the natural loblolly pine growing region in the Piedmont, Atlantic, and Gulf Coastal Plain areas (fig. 1) were used for this study (Amateis and others 2006). The plots were installed during the period 1996 to 2000 in plantations representative of current loblolly pine plantation management and silvicultural practices. All received site preparation and vegetation control treatments appropriate for the site, were planted with genetically improved stock suitable for the locale, and have received operationally applied fertilization and competition control treatments as needed. Mostly unthinned plots of 0.15-acre size were used for this study, although there were some thinned plots as well. Each plot was measured at establishment and subsequently on a 1-, 2-, or 3-year cycle. Estimates of average annual dominant height and basal area growth between measurements were obtained by differencing successive measurements and dividing by the number of years between measurements (linear interpolation). There was a total of 1,276 annual dominant height and basal area growth measurements for ages 5 to 15 (table 1).

Interpolated annual precipitation estimates occurring at each IMP site for the period 1997 to 2006 were obtained from the PRISM climate database (http://www.prismclimate.org).

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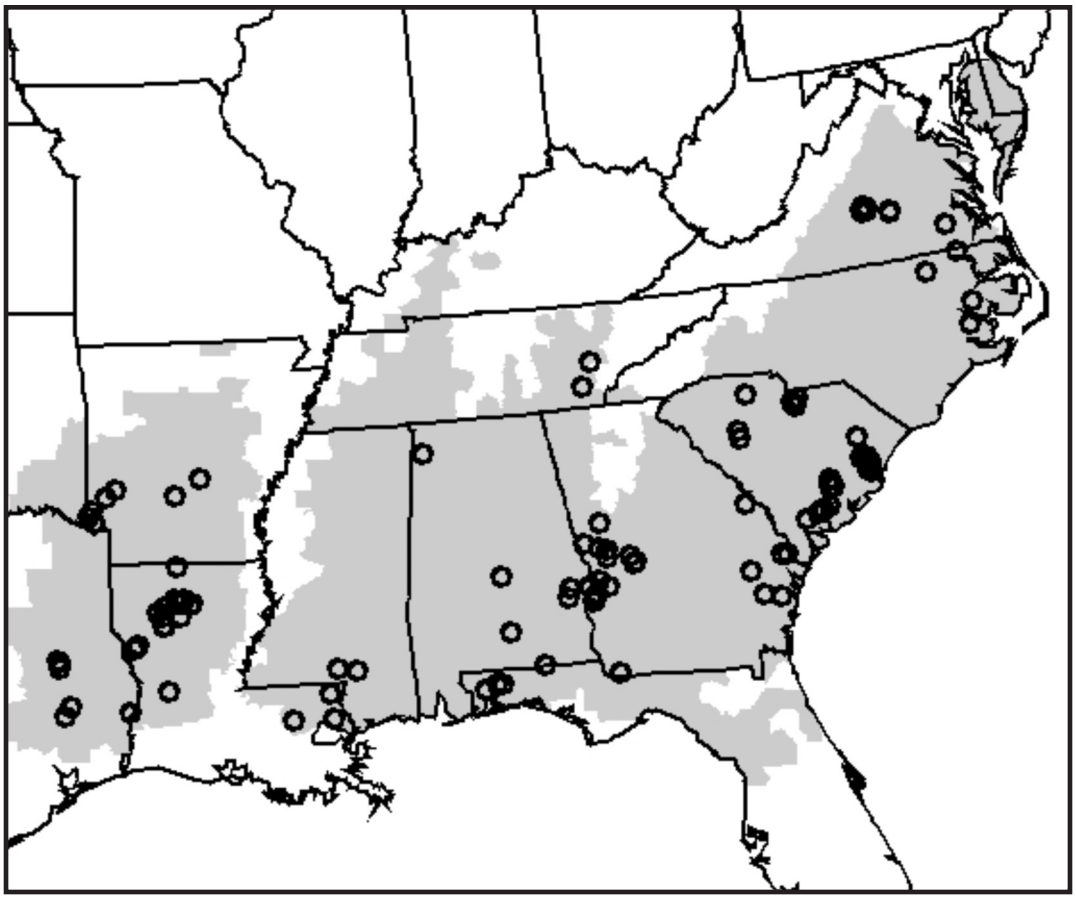

Figure 1—Plot locations for intensively managed loblolly pine plantations study (grey area indicates the natural range of loblolly pine).

Table 1—Summary statistics for 1,276 dominant height and basal area growth observations

Variable	Mean	Std. dev.	Minimum	Maximum
Age (years)	8.5	2.4	5.0	15.0
Site index (feet), base 25	71.4	7.3	51.6	94.3
Basal area (square feet per acre)	74.2	35.7	1.0	183.9
Dominant height (feet)	29.4	9.9	4.9	59.9
Basal area growth (square feet per acre per year)	13.2	4.5	1.0	28.2
Dominant height growth (feet per year)	3.6	1.0	0.6	10.5
Annual precipitation (inches)	51.4	8.7	34.5	73.1
Latitude (decimal degrees)	33.2	2.0	30.4	37.5
Longitude (negative decimal degrees)	–85.9	6.1	–95.0	–76.9

Std. dev. = standard deviation.

An average annual precipitation estimate for each growth period on each plot was computed by summing the annual precipitation estimates between measurements and dividing by the number of years between measurements. These annual precipitation estimates were merged with the annual growth data for the same year to construct a dataset that could be used to evaluate the impact of precipitation on loblolly pine growth.

METHODS AND RESULTS

The approach taken for this study was to use regression methods to develop baseline prediction equations for estimating annual dominant height (G_{HD}) growth (feet per year) and basal area (G_{BA}) growth (square feet per acre per year) from stand and site variables typically used in growth-and-yield models. Then, annual precipitation was added to the base model to determine if precipitation was a significant regressor in the presence of the other variables.

A power function was used as a base model and linearized for both G_{HD} and G_{BA}:

$$\ln(G_{HD}) = b_0 + b_1 \ln(1/A) + b_2 \ln(HD) + b_3 \ln(S) \quad (1)$$

$$\ln(G_{BA}) = b_0 + b_1 \ln(S) + b_2 \ln(HD) + b_3 \ln(BA) + b_4 \ln(LAT) \quad (2)$$

where
- A = stand age
- HD = average height of dominant and codominant trees
- S = site index (feet, base 25)
- BA = basal area (square feet per acre)
- LAT = latitude (decimal degrees)

Other stand and site variables including number of trees, longitude, and physiographic region were not significant regressors in either equation (1) or (2) and did not reduce the PRESS statistic. Base equations (1) and (2) have *r*-square values of 0.42 and 0.50, respectively.

The logarithm of annual precipitation (PPT) was added as a regressor to both base models, and the models were refitted to the data. For equation (1), PPT was not a significant regressor. For equation (2), however, PPT was significant and adding it to equation (2) to create equation (3) resulted in an *r*-square of 0.53, a reduction of the PRESS statistic and a reduction in the mean square error by about 7 percent over equation (2).

$$\ln(G_{BA}) = b_0 + b_1 \ln(S) + b_2 \ln(HD) + b_3 \ln(BA) + b_4 \ln(LAT) + b_5 \ln(PPT) \quad (3)$$

Attempts to improve equation (3) by adding additional regressor variables including the lagged precipitation value from the previous year were unsuccessful. Interaction terms were also not significant regressors. Table 2 presents the fit statistics and parameter estimates for equations (2) and (3).

DISCUSSION

An important result of this study was that annual precipitation was not correlated with dominant height growth for these data. This may be due to the fact that much of the annual height growth occurs relatively early in the growing season drawing on moisture that has accumulated during the previous dormant season. Drought conditions that might negatively impact height growth generally do not materialize until later in the growing season. Physiologically, trees will favor height growth over cambial growth when under stress which may be another reason annual rainfall is not significant.

Basal area growth, on the other hand, was significantly correlated with local annual precipitation. This may be because considerable cambial growth occurs later in the growing season when drought conditions are likely to be more severe. It may also be because, as noted above, trees will utilize resources to favor height growth at the expense of diameter growth when under stress.

Table 2—Basal area growth prediction equation fitted to stand and site variables without [equation (2)] and with [equation (3)] annual precipitation included

	Base model [equation (2)]			Base model with precipitation [equation (3)]		
Parameter	Estimate	Standard error	*P*-value	Estimate	Standard error	*P*-value
Intercept	–2.8769	0.5723	<0.0001	–6.914	0.6933	<0.0001
ln(S)	1.4405	0.0815	<0.0001	1.473	0.0787	<0.0001
ln(HD)	–1.2885	0.0383	<0.0001	–1.337	0.0373	<0.0001
ln(BA)	0.4682	0.0221	<0.0001	0.4889	0.0214	<0.0001
ln(LAT)	0.4516	0.1259	<0.0003	1.072	0.1375	<0.0001
ln(PPT)				0.4592	0.0476	<0.0001
	MSE = 0.0704 R^2 = 0.50			MSE = 0.0656 R^2 = 0.53		

S = site index (feet, base 25); HD = average height of dominant and codominant trees (feet); BA = basal area (square feet per acre); LAT = latitude (decimal degrees); PPT = annual precipitation (inches); MSE = mean square error.

Equation (3) can be used in various ways to estimate the impact of losses due to drought on the growth of loblolly pine plantations. One way is to use a growth-and-yield model to make a 1-year projection from observed stand and site conditions. This will result in an estimated basal area and volume for "normal" (or "average") levels of precipitation. Then, model 3 can be implemented and estimated annual basal area growth under different levels of precipitation can be compared to the "normal" level estimated using the growth-and-yield model.

Equation (3) can also be used directly to adjust output from growth-and-yield models by an appropriate percentage. For example, in 2007 parts of Alabama experienced a severe drought where precipitation for the year was about 26 inches instead of the "normal" 56 inches. For a stand at latitude 34° with site index 65 feet, dominant height of 50 feet, and basal area of 150 square feet per acre, equation (3) predicts basal area growth of about 8 square feet per acre per year under "normal" precipitation and 5.6 square feet per acre per year under drought conditions. This suggests a basal area growth loss for the year of about 30 percent. Since height growth will not be much affected by drought, and assuming the form factor for trees growing under drought conditions as compared to trees growing under conditions of "normal" precipitation will not be much affected either, volume growth loss for the year would be expected to be about 30 percent as well.

ACKNOWLEDGMENTS

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LITERATURE CITED

Amateis, R.L.; Prisley, S.P.; Burkhart, H.E.; Liu, J. 2006. The effect of physiographic region and geographic locale on predicting the dominant height and basal area of loblolly pine plantations. Southern Journal of Applied Forestry. 30: 147–153.

Bassett, J.R. 1964. Tree growth as affected by soil moisture availability. Soil Science Society of America Proceedings. 28: 436–438.

Coile, T.S. 1936. The effect of rainfall and temperature on the annual radial growth of pine in the Southern United States. Ecological Monographs. 6: 533–562.

Jacobi, J.C.; Tainter, F.H. 1988. The effect of drought on growth decline of loblolly pine on littleleaf sites. Plant Disease. 72: 294–297.

Langdon, O.G.; Trousdell, K.B. 1979. Stand manipulation: effects on soil moisture and tree growth in southern pine and pine-hardwood stands. In: Proceedings, symposium on soil moisture-site productivity. Atlanta: U.S. Department of Agriculture Forest Service, Southeastern Area, State and Private Forestry: 221–236.

Wheeler, G.L.; Meade, F.M.; Russell, M.W. 1982. Growth of loblolly pine in the Arkansas Ozarks. Southern Journal of Applied Forestry. 6: 215–217.

Yeiser, J.L.; Burnett, F. 1982. Fate of forest trees stressed by heat and drought in southeastern Arkansas. Southern Journal of Applied Forestry. 6: 194–195.

MODELING THINNING IN EAST TEXAS LOBLOLLY AND SLASH PINE PLANTATIONS

Dean W. Coble[1]

Abstract—A new thinning model was proposed for loblolly pine (*Pinus taeda* L.) and slash pine (*P. elliottii* Engelm.) plantations in east Texas. The new model follows the index of suppression methodology introduced by Pienaar (1979). It was implemented in a new whole stand growth model for loblolly and slash pine plantations in east Texas (Coble 2009). The new thinning model performed similarly to existing Pienaar-type models for east Texas and the Southeastern United States across a range of site quality. The predicted basal area development consistently approached the unthinned counterpart, which is consistent with results from other studies. The new thinning model should be fully tested when empirical data become available. In the meantime, it can be used to model thinned loblolly and slash pine plantations in east Texas ranging in age from 5 to 40 years.

INTRODUCTION

Plantations are routinely thinned to free growing space for residual trees to grow into larger, more valuable sawtimber-sized trees. Forest managers therefore need thinning response models to better understand the growth and yield of thinned plantations. Pienaar (1979) described a methodology that uses an index of suppression to model the growth of thinned plantations. His methodology has been subsequently used by others to model the growth of thinned plantations in the Southeastern United States (Borders and others 2004, Harrison and Borders 1996). Burrow (2001) applied Pienaar's methodology to east Texas loblolly pine (*Pinus taeda* L.) plantations and also provided a new formulation of the index of suppression.

The purpose of this study was to examine the behavior of Pienaar's and Burrow's indexes of suppression and propose a new thinning model that can be used in east Texas loblolly and slash pine (*P. elliottii* Engelm.) plantations. Currently, empirical data are unavailable to fully test a thinning model for east Texas. The proposed model in this study can be tested as thinning data become available. In the meantime, the proposed model was incorporated into a new whole stand growth-and-yield model for east Texas loblolly and slash pine plantations (Coble 2009) to examine the thinning response at three levels of site index.

METHODS

The thinning model of Pienaar (1979) is based on a competition index or index of suppression that describes the relative impact of competition among trees in thinned and unthinned stands. The competition index (CI) relates the basal area per acre of a thinned stand to an unthinned stand with the same dominant height, trees per acre, and age (the unthinned counterpart) (Borders and others 2004):

$$CI = 1 - \frac{B_{at}}{B_u} \tag{1}$$

where
B_{at} = basal area (square feet) per acre after thinning
B_u = basal area per acre of the unthinned counterpart

Since thinning prescriptions are typically expressed as residual trees per acre, basal area per acre removed should functionally relate to trees per acre removed from a row thin, select thin, or a row-select thin (Borders and others 2004:

$$\frac{B_t}{B} = \frac{N_r}{N} + \left[1 - \frac{N_r}{N}\right]\left[\frac{N_S}{N - N_r}\right]^{\gamma} \tag{2}$$

where
B_t = basal area (square feet) per acre removed in thinning
B = basal area per acre prior to thinning
N_r = trees per acre removed in row thinning
N_s = trees per acre removed in select thinning
N = trees per acre prior to thinning
γ = parameter

The CI must be projected to a future time to estimate the future basal area per acre of the thinned stand (Borders and others 2004, Pienaar 1979):

$$CI_2 = CI_1 e^{-\phi(A_2 - A_1)} \tag{3}$$

where
CI_i = CI at times i = 1 and 2
A_i = plantation age (years) at times i = 1 and 2
ϕ = parameter
e = exponential function

The CI at the projection age (time 2) can be expressed in terms of the equation 1 (Borders and others 2004):

$$CI_2 = 1 - \frac{B_{at_2}}{B_{u_2}} \tag{4}$$

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Equation 4 can be algebraically rearranged to find the basal area per acre of the thinned stand at the projection age, when the projected basal area per acre of the unthinned counterpart is known (Borders and others 2004):

$$B_{at_2} = B_{u_2}\left(1 - CI_2\right) \tag{5}$$

Based on Border and others (2004) and Burrow (2001) for loblolly pine and Pienaar (1979) for slash pine, the following hypothesized values will be assigned to the parameters in equations 2 and 3:

$\gamma = 1.2$
$\phi = 0.1$

Burrow (2001) also provided a new formulation of the CI that will also be examined in this study:

$$CI_2 = \left(\frac{A_1}{A_2}\right)CI_1 + b_1\left(\frac{A_1}{A_2} - 1\right)\left(\frac{A_2 - A_1}{A_2}\right) + b_2\left(\frac{A_1}{A_2} - 1\right)\left(\frac{SI}{A_2}\right) + b_3\left(\frac{A_1}{A_2} - 1\right)\left(\frac{CI_1}{A_2}\right) \tag{6}$$

where
SI = site index (index age = 25 years) (Coble and Lee 2006)
b_i = regression parameters

This thinning methodology was incorporated into a whole stand growth-and-yield model for loblolly and slash pine plantations in east Texas (Coble 2009) to examine thinning responses at low (SI = 50 feet), medium (SI = 70 feet), and high (SI = 90 feet) site quality. The parameter values g and f (equations 4 and 6, respectively) were changed to compare between the thinning models of this study—Burrow (2001) for loblolly pine in east Texas, Borders and others (2004) for the lower Coastal Plain, Borders and others (2004) for loblolly pine in the Upper Coastal Plain and Piedmont, and Pienaar (1979) for slash pine. Yield curves will be compared for a plantation with a planting density = 605 trees per acre (tpa) that was thinned to 250 tpa at 15 years old.

RESULTS AND DISCUSSION

For loblolly pine, the Pienaar-type thinning models (equation 4) are indistinguishable in their prediction of future basal area per acre after thinning for all levels of site quality (figs. 1A, 1B, and 1C). The modified competition index of Burrow (2001), equation 6, predicts greater basal area values than the Pienaar-type models (figs. 1A, 1B, and 1C). The Pienaar-type models all approach the unthinned counterpart at an increasing rate from lowest site quality (fig. 1A) to highest site quality (fig. 1C). At the highest site quality, the thinned stand approaches and then tracks the unthinned counterpart for all Pienaar-type models (fig. 1C). The modified competition index, equation 6, seems to approach a different unthinned counterpart than was defined in this study. In this study, the unthinned counterpart is defined as an unthinned stand that has the same density (tpa) as the thinned stand at the thinning age. Equation 6 appears to approach an unthinned counterpart defined as the unthinned version of the stand that got thinned. So, a forest manager could choose to redefine the unthinned counterpart, depending on whether they desired conservative (equation 4) or aggressive (equation 6) postthinning yield estimates from the model.

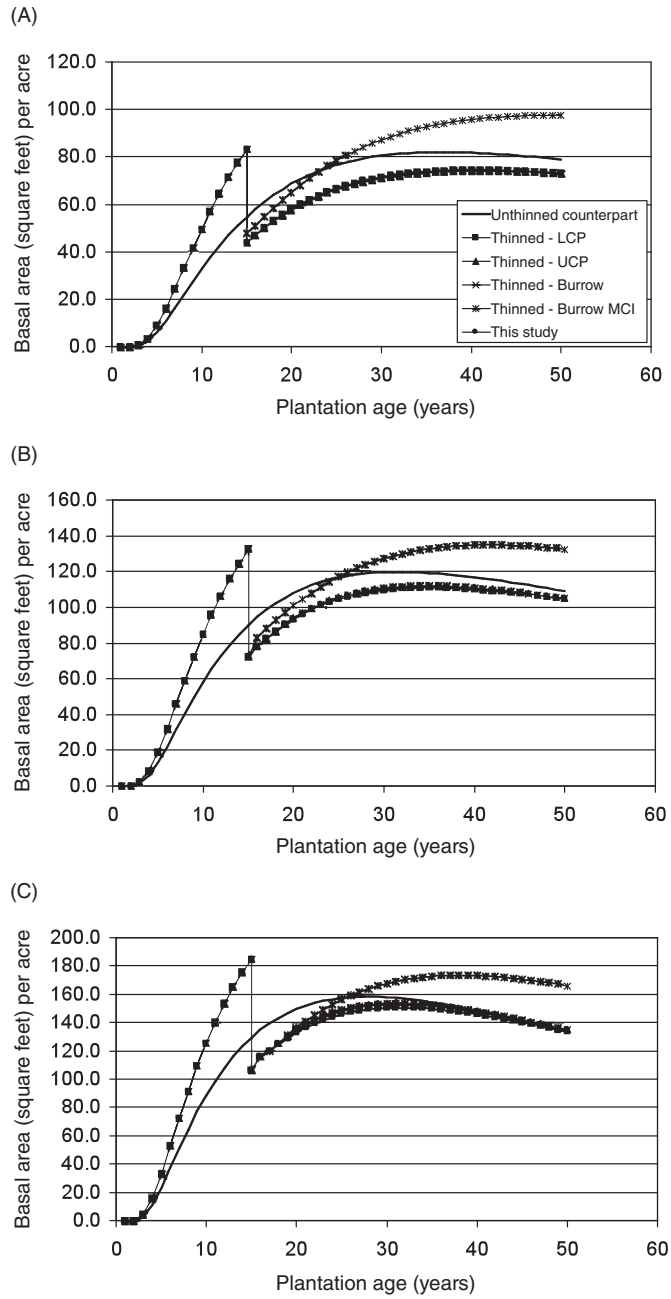

Figure 1—Projected basal area for the example loblolly pine plantation (this study) relative to its unthinned counterpart and four other thinning models at site indices: (A) 50 feet, (B) 70 feet, and (C) 90 feet.

For slash pine, the results are similar to those for loblolly pine. The models of this study and Pienaar (1979) are identical in their prediction of future basal area per acre after thinning for all levels of site quality (figs. 2A, 2B, and 2C). For low site quality, the thinned stands appear to parallel the unthinned

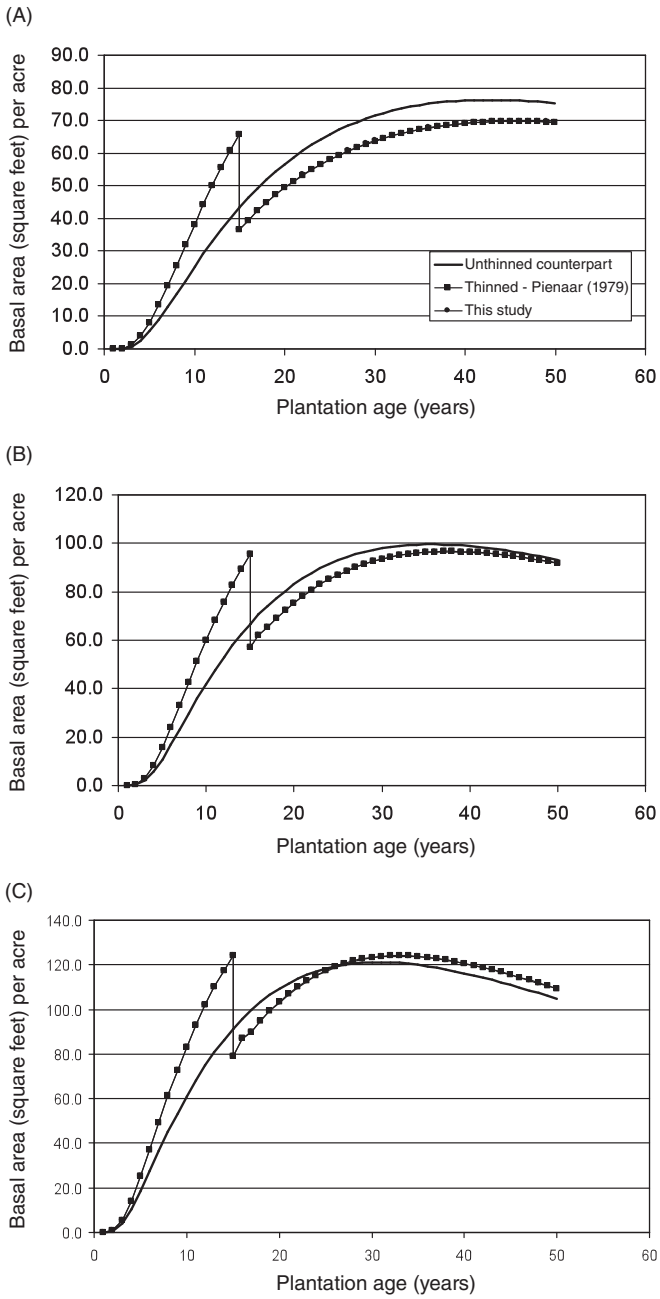

Figure 2—Projected basal area for the example slash pine plantation (this study) relative to its unthinned counterpart and four other thinning models at site indices: (A) 50 feet, (B) 70 feet, and (C) 90 feet.

counterpart (fig. 2A), but for medium site quality, they approach the unthinned counterpart (fig. 2B). For high site quality, the thinned stands approach and pass the unthinned counterpart (fig. 2C). This result for high site quality differs for that of loblolly pine. For loblolly pine, the Pienaar-type thinning models never exceed the unthinned counterpart.

CONCLUSIONS AND RECOMMENDATIONS

The Pienaar-type thinning models represented by equation 4 seem to predict postthinning basal area development reasonably well for low, medium, and high site qualities. The hypothesized parameter values in this study produce similar results as those estimated by Burrow (2001) and Borders and others (2004). Since data are unavailable to test a thinning model, I recommend a conservative approach to modeling thinning in east Texas pine plantations. Forest managers should utilize equation 4 and the hypothesized parameters in this study. When data become available, these hypothesized parameter values can be fully tested.

LITERATURE CITED

Borders, B.E.; Harrison, W.M.; Zhang, F.Y. [and others]. 2004. Growth and yield models for second rotation loblolly pine plantations in the Piedmont/Upper Coastal Plain and lower Coastal Plain of the Southeastern U.S. PMRC Res. Pap. Athens, GA: University of Georgia, School of Forest Resources. 64 p.

Burrow, A.M. 2001. Predicting basal area growth and yield of thinned loblolly pine (*Pinus taeda*, L.) plantations in the West Gulf Region. Nacogdoches, TX: Stephen F. Austin State University. 81 p. M.S. thesis.

Coble, D.W. 2009. A new whole-stand model for unmanaged loblolly and slash pine plantations in east Texas. Southern Journal of Applied Forestry. 33:69–76.

Coble, D.W.; Lee, Y.J. 2006. Use of a generalized sigmoid growth function to predict site index for unmanaged loblolly and slash pine plantations in east Texas. In: Connor, Kristina F., ed. Proceeding 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: 291–295.

Harrison, W.M.; Borders, B.E. 1996. Yield prediction and growth projection for site-prepared loblolly pine plantations in the Carolinas, Georgia, Alabama and Florida. PMRC Res. Pap. 1996-1. Athens, GA: University of Georgia, School of Forest Resources. 59 p.

Pienaar, L.V. 1979. An approximation of basal area growth after thinning based on growth in unthinned plantations. Forest Science. 25(2): 223–232.

PREDICTING DIAMETER AT BREAST HEIGHT FROM TOTAL HEIGHT AND CROWN LENGTH

Quang V. Cao and Thomas J. Dean[1]

Abstract—Tree diameter at breast height (d.b.h.) is often predicted from total height (model 1a) or both total height and number of trees per acre (model 1b). These approaches are useful when Light Detection and Ranging (LiDAR) data are available. LiDAR height data can be employed to predict tree d.b.h., and consequently individual tree volumes and volume/ha can be obtained for the tract. In this paper, we will examine alternative methods of predicting d.b.h. from total height and crown length (model 2a), or from total height, crown length, and number of trees per acre (model 2b), based on the uniform stress theory. The uniform stress theory hypothesizes that stems behave like tapered cantilever beams to equalize bending stress across their length. The four models were evaluated based on the mean difference between observed and predicted diameters, mean absolute difference, and fit index. Results revealed that the two models based on the uniform stress theory (models 2a and 2b) were more appropriate for predicting d.b.h., which is needed to compute tract volume using LiDAR data.

INTRODUCTION

Airborne laser scanning or Light Detection And Ranging (LiDAR) has been used in many forestry applications (Lefsky and others 1999, Means and others 2000, Nelson and others 1988, Nilsson 1996, Parker and Evans 2004, Parker and Mitchell 2005) and can provide measurements of height and crown dimensions. The vertical distribution of forest canopy can be characterized with LiDAR (Arp and others 1982, Dean and others 2009, Drake and others 2002, Harding and others 2001, Lefsky and others 1999, Ritchie and others 1993). In an application of LiDAR in forest inventory, Parker and Evans (2004) evaluated different functions to predict diameter at breast height (d.b.h.) from total tree height and number of trees per unit area. The predicted diameter is needed for calculation of individual tree volumes and ultimately of stand volume.

Stem diameter can also be predicted from the uniform stress theory. This theory states that the taper of tree boles allows them to equalize bending stress (produced mainly by wind pressure on the crown foliage) across their length (Dean and Long 1986). Evidence exists showing a strong relationship between foliage distribution and stem size and taper (Dean 2004, Dean and Long 1986, Dean and others 2002, West and others 1989). Dean and Long (1986) applied the uniform stress model to predict stem diameter anywhere on the tree bole, based on the length of the lever arm and total leaf area above that point. Therefore, for a fixed height such as breast height, diameter can be predicted from total height, crown length, and total leaf area. Total tree height and crown length can be obtained from LiDAR data, and total leaf area can be predicted from total height and crown length by use of a regression equation (Jerez 2002, Roberts and others 2003). Therefore, the uniform stress model allows d.b.h. to be predicted from just two parameters—total height and crown length.

The conventional method so far has been to predict d.b.h. from either total height or from total height and number of trees/ha. The objective of this study was to determine if adding crown length to the above predictor variables improves the prediction.

DATA

Data collected from a loblolly pine (*Pinus taeda*) plantation at the Hill Farm Research Station, Homer, LA, were used in this study. Twenty 0.1-ha (0.25-acre) plots were established with seedlings planted at 1.83- by 1.83-m (6- by 6-foot) spacing. The plots were thinned to 2,470, 1,482, 741, 494, and 247 trees/ha (1,000, 600, 300, 200, and 100 trees per acre) in a stepwise thinning procedure, completed by age 7. Measurements for each tree include d.b.h., total height, and height to the base of live crown. Measurements from 278 trees in 14 plots at age 21 constitute the fit dataset, used for estimation of coefficients of the regression models.

The validation dataset comprised 454 trees at age 28 from another study, also at the Hill Farm Research Station. These trees came from 26 plots of size 0.1 ha (0.25 acres), which underwent thinning (to 741, 494, and 247 trees/ha at age 11) and pruning (once at age 6, and twice at ages 6 and 11) treatments. Summary statistics for the fit and validation datasets are shown in table 1.

MODELS

Conventional Models

Conventional models were developed to predict d.b.h. from total height, or from total height and number of trees/ha. The following models were selected as most appropriate for the fit data, based on an evaluation of numerous models:

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Table 1—Summary statistics of stand and tree variables in the fit and validation datasets

Variable	*n*	Mean	Std. dev.	Min.	Max.
Fit dataset					
D.b.h. (cm)	278	23.3	6.3	11.4	48.3
Total height (m)	278	17.6	2.5	9.1	23.4
Crown ratio	278	0.41	0.12	0.15	0.83
Number of trees per ha	14	849	593	237	2303
Validation dataset					
D.b.h. (cm)	454	28.1	6.1	15.5	48.0
Total height (m)	454	21.7	2.4	13.9	27.5
Crown ratio	454	0.37	0.09	0.04	0.73
Number of trees per ha	26	440	187	194	717

Std. dev. = standard deviation; Min. = minimum; Max. = maximum.

Model 1a—This model performed slightly better than Parker and Evans' (2004) model, which used the natural logarithm of *H* instead of *H* as the independent variable.

$$D = b_1 + b_2 H^{b_3} \quad (1)$$

where
D = diameter at breast height in cm
H = total height in m
b_i's = regression coefficients

Model 1b—Stand density in terms of number of trees/ha (N) was added to equation (1) to form model 1b:

$$D = b_1 + b_2 H^{b_3} N^{b_4} \quad (2)$$

Parker and Evans (2004) evaluated four models and found that model 1b performed best in five out of six datasets.

Models Based on The Uniform Stress Theory

Dean and Long (1986) proposed the following taper model to predict tree diameter in cm (d_h) at height h in m:

$$d_h = b_1 (A_h L_h)^{b_2} \quad (3)$$

where
A_h = total leaf area (m^2) above d_h
L_h = distance in m between the center of leaf area above d_h and the point at height h

Model 2a—By fixing h at 1.37 m or 4.5 feet, one can predict d.b.h. using the above equation:

$$D = b_1 x^{b_2} \quad (4)$$

where
$x = A\ L_h$
A = total leaf area (m^2)
$L_h = H_{MC} - 1.37$
$H_{MC} = H_T - CL/2$ = height to the center of the crown
H_T = total tree height in meters
CL = crown length in meters

Total leaf area was predicted from an equation developed by Roberts and others (2003) and refitted by Jerez (2002):

$$\log(A) = -2.19715 + 7.5437 \log(H_T) - 5.422006 \log(H_{MC}) \quad (5)$$

where
$\log(A)$ = logarithm base 10 of A

Model 2b—Similar to model 1b, model 2b was obtained by adding number of trees/ha to equation (4):

$$D = b_1 x^{b_2} N^{b_3} \quad (6)$$

It is evident that we had two groups of models: models 1a and 2a required only heights, whereas models 1b and 2b required both heights and number of trees/ha as predictor variables. The regression coefficients in these models were obtained with nonlinear regression.

RESULTS AND DISCUSSION

The four models were evaluated based on three statistics: mean difference (MD) between observed and predicted diameters, mean absolute difference (MAD), and fit index (FI), which is computationally similar to R^2 in linear regression. Table 2 shows the evaluation statistics for the four models, based on the fit and validation datasets.

Diameter Models Based on Heights

Model 1a had a bias MD close to zero for the fit data, but its bias increased to –3.058 cm for the validation dataset. On the other hand, model 2a produced an MD value of only 0.252 cm for the validation data. For both the fit and validation data, MAD was lower and FI was higher for model 2a than for model 1a. For the validation data, model 2a lowered the MAD value from 4.914 cm to 3.192 cm and increased the FI value from 0.008 to 0.560. Trees in the validation dataset were older and, on the average, larger and taller than those in the fit dataset. This might explain why model 1a failed to adequately represent the validation data (fig. 1). On the other hand, model 2a characterized both the fit and validation data equally well (fig. 1). This suggests that the uniform stress model was reasonably reliable and could be employed with confidence to describe a larger segment of the population.

Diameter Models Based on Heights and Stand Density

The evaluation statistics were slightly better for model 1b and model 2b for the fit dataset; however, its prediction capability drastically diminished for the validation data. The value of FI fell from 0.760 to 0.217 for the fit and validation datasets, respectively. Testing model 2b with the same validation data, we obtained the following statistics: FI = 0.645 cm, MAD = 2.957 cm (vs. 4.404 cm from model 1b), and MD = –0.214 cm (vs. –3.595 cm from model 1b). Figure 2 shows that the modified uniform stress model with the addition of number of trees/ha did a good job in both fitting the sample data and predicting for the population.

Use of Stand Density as an Additional Variable

Adding number of trees/ha as an extra predictor variable improved both the fit and predictive ability of models 1a and 2a. Parker and Evans (2004) obtained similar results. FI value for the validation data in this study increased from 0.008 to 0.217 for model 1b and from 0.560 to 0.645 for model 2b. Because tree counts are readily available from LiDAR data, this variable should be incorporated into model for predicting diameter from remotely sensed data.

Table 2—Evaluation statistics for the four models to predict d.b.h.

Model	Equation	MD	MAD	FI
		Fit dataset		
1a	$D = b_1 + b_2 H^{b_3}$	0.000	4.174	0.293
1b	$D = b_1 + b_2 H^{b_3} N^{b_4}$	0.000	2.472	0.760
2a	$D = b_1 x^{b_2}$	0.023	3.257	0.550
2b	$D = b_1 x^{b_2} N^{b_3}$	0.010	2.552	0.734
		Validation dataset		
1a	$D = b_1 + b_2 H^{b_3}$	–3.058	4.914	0.008
1b	$D = b_1 + b_2 H^{b_3} N^{b_4}$	–3.595	4.404	0.217
2a	$D = b_1 x^{b_2}$	0.252	3.192	0.560
2b	$D = b_1 x^{b_2} N^{b_3}$	–0.214	2.957	0.645

MD = mean difference between observed and predicted diameters; MAD = mean absolute difference; FI = fit index (computationally similar to R^2 in linear regression).

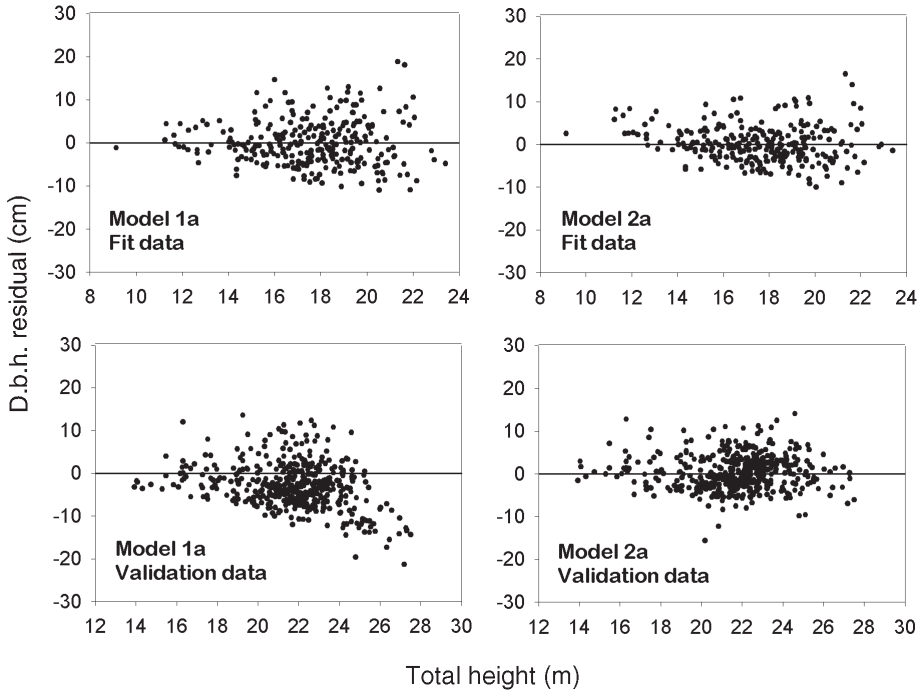

Figure 1—Fit data and validation data for model 1a and model 2a.

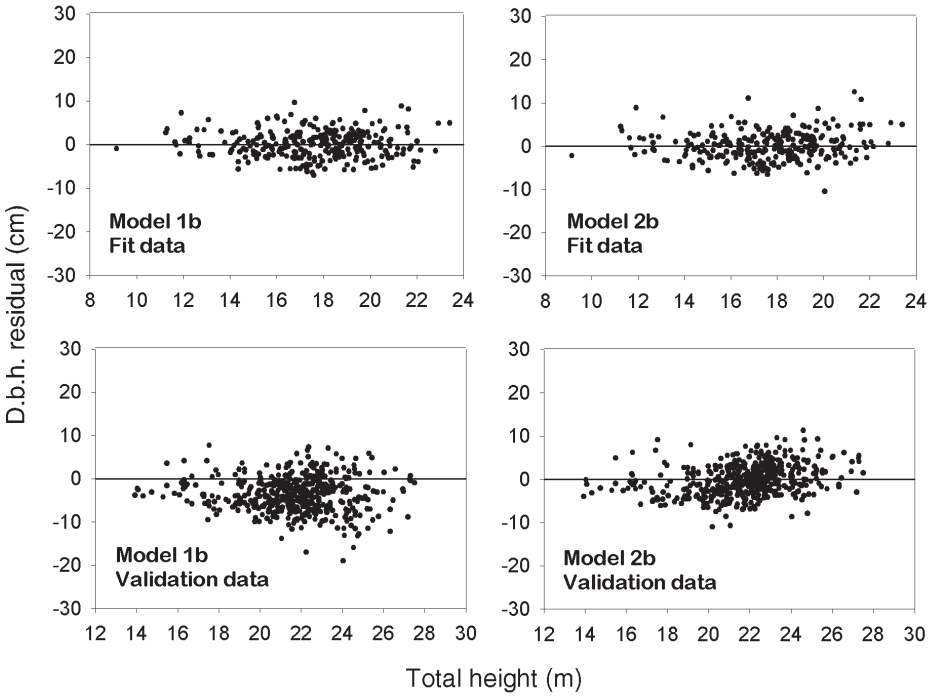

Figure 2—Fit data and validation data for model 1b and model 2b.

SUMMARY AND CONCLUSIONS

In this study, the conventional method of predicting d.b.h. from total height (model 1a) was evaluated against the new method of predicting d.b.h. from total height and crown length (model 2a). Evaluation statistics and residual plots revealed that model 2a, which was based on the uniform stress theory, was better at predicting diameters for the validation data. Similar results were obtained when number of trees/ha was included as a predictor variable. The addition of crown length (model 2b) drastically improved the d.b.h. prediction for the validation data. These results showed that the uniform stress theory can be successfully modified to predict d.b.h. from total height, crown length, and number of trees/ha, all of which can be obtained from LiDAR data.

LITERATURE CITED

Arp, H.; Griesbach, J.; Burns, J. 1982. Mapping tropical forests: a new approach using the laser APR. Photographic Engineering and Remote Sensing. 48: 91–100.

Dean, T.J. 2004. Basal area increment and growth efficiency as functions of canopy dynamics and stem mechanics. Forest Science. 50: 106–116.

Dean, T.J.; Cao, Q.V.; Roberts, S.D.; Evans, D.L. 2009. Measuring heights to crown base and crown median with LiDAR in a mature, even-aged loblolly pine stand. Forest Ecology and Management. 257: 126–133.

Dean, T.J.; Long, J.N. 1986. Validity of constant-stress and elastic-instability principles of stem formation in *Pinus contorta* and *Trifolium pretense*. Annals of Botany. 58: 833–840.

Dean, T.J.; Roberts, S.D.; Gilmore, D.W. [and others]. 2002. An evaluation of the uniform stress hypothesis based on stem geometry in selected North American conifers. Trees. 16: 559–568.

Drake, J.B.; Dubayah, R.O.; Knox, R.G. [and others]. 2002. Sensitivity of large-footprint Lidar to canopy structure and biomass in a neotropical rainforest. Remote Sensing of Environment. 81: 378–392.

Harding, D.J.; Lefsky, M.A.; Parker, G.G.; Blair, J.B. 2001. Laser altimeter canopy height profiles: methods and validation for closed-canopy, broadleaf forests. Remote Sensing of Environment. 76: 283–297.

Jerez, M. 2002. Modeling canopy structure effects on loblolly pine growth. Baton Rouge, LA: Louisiana State University. 79 p. Ph.D. dissertation.

Lefsky, M.A.; Cohen, W.B.; Acker, S.A. [and others]. 1999. LIDAR remote sensing of the canopy structure and biophysical properties of Douglas fir-western hemlock forests. Remote Sensing of Environment 70: 339–361.

Means, J.E.; Acker, S.A.; Fitt, J.B. [and others]. 2000. Predicting forest stand characteristics with airborne scanning lidar. Photogrammetric Engineering and Remote Sensing. 66: 1367–1371.

Nelson, R.; Krabill, W.; Tonelli, J. 1988. Estimating forest biomass and volume using airborne laser data. Remote Sensing of Environment. 24: 247–267.

Nilsson, M. 1996. Estimation of tree heights and stand volume using an airborne LiDAR system. Remote Sensing of Environment. 56: 1–7.

Parker, R.C.; Evans, D.L. 2004. An application of LiDAR in a double-sample forest inventory. Western Journal of Applied Forestry. 19: 95–101.

Parker, R.C.; Mitchel, A.L. 2005. Smoothed versus unsmoothed LiDAR in a double-sample forest inventory. Southern Journal of Applied Forestry. 29: 40–47.

Ritchie, J.C.; Evans, D.L.; Jacobs, D. [and others]. 1993. Measuring canopy structure with an airborne laser altimeter. Transactions of the ASAE. 36: 1235–1238.

Roberts, S.D.; Dean, T.J.; Evans, D.L. 2003. Family influences on leaf area estimates derived from crown and tree dimensions in *Pinus taeda*. Forest Ecology and Management. 172: 261–270.

West, P.W.; Jackett, D.R.; Sykes, S.J. 1989. Stresses in, and the shape of, tree stems in forest monoculture. Journal of Theoretical Biology. 140: 327–343.

ESTIMATING LOBLOLLY PINE SIZE-DENSITY TRAJECTORIES ACROSS A RANGE OF PLANTING DENSITIES

Curtis L. VanderSchaaf and Harold E. Burkhart[1]

Abstract—Size-density trajectories on the logarithmic (ln) scale are generally thought to consist of two major stages. The first is often referred to as the density-independent mortality stage where the probability of mortality is independent of stand density; in the second, often referred to as the density-dependent mortality or self-thinning stage, the probability of mortality is related to stand density. Within the self-thinning stage, segments of a size-density trajectory consisting of a nonlinear approach to a linear portion, a linear portion (maximum size-density relationship dynamic thinning line), and a divergence from the linear portion are generally assumed. Estimates of the ln of quadratic mean diameter and ln of trees per acre where the two stages of stand development and the three phases of self-thinning begin and end were obtained from segmented regression analyses and used as response variables predicted as a function of planting density. Predicted values allow for size-density trajectories to be estimated for any planting density.

INTRODUCTION

Self-thinning quantifies the relationship between average tree size and tree density and has been widely studied. Understanding self-thinning is important to better grasp intraspecific mortality patterns of a tree species growing in even-aged stands leading to more efficient management of growing stock. For instance, estimating the onset of self-thinning can help resource managers plan thinnings and reduce competition-induced mortality. Quantifying maximum size-density relationships (MSDR), or the maximum obtainable tree density per unit area for a given quadratic mean diameter (*D*), should help resource managers better understand how different management regimes affect productivity. Predictions of MSDRs can be used to constrain and verify estimated stand development of process-based models and those empirical models that were developed using data limited in ranges of density and/or age to properly estimate mortality equations. Maximum size-density relationships have been used as constraints in several growth-and-yield models (Monserud and others 2004, Poage and others 2007) both for the ln*V*-ln*N* relationship (e.g., Landsberg and Waring 1997, Smith and Hann 1984, Turnblom and Burk 2000) and the ln*N*-ln*D* relationship (e.g., Hynynen 1993, Johnson 2000). In many model systems, mortality equations are combined with height, diameter, or volume equations to estimate an approach to a linear MSDR constraint, and once the projected stand density is equivalent to the linear constraint, self-thinning occurs such that stand density is maintained equivalent to the linear constraint for some period of time.

VanderSchaaf (2006) and VanderSchaaf and Burkhart (2008) proposed using segmented regression to provide a less subjective, statistically based criteria to determine what observations are within various stages and phases of stand development and at what ln*N* and ln*D* the various stages and phases begin and terminate, where ln is the natural logarithm and *N* is trees per acre. In this paper, equations are presented to predict size-density trajectories of loblolly pine (*Pinus taeda* L.) plantations across a range of planting densities using results obtained from VanderSchaaf (2006) and VanderSchaaf and Burkhart (2008).

Stages of Stand Development and Phases of Self-Thinning

For size-density trajectories on the ln scale, two major stages of stand development are generally recognized (Drew and Flewelling 1979, McCarter and Long 1986, Williams 1994): the first being an initial stage without significant competition in which mortality is independent of stand density (fig. 1—stage I), often referred to as the density-independent mortality stage, and the second being a stage with competition-induced mortality (the self-thinning stage) often referred to as the density-dependent mortality stage (fig. 1—stage II). Within

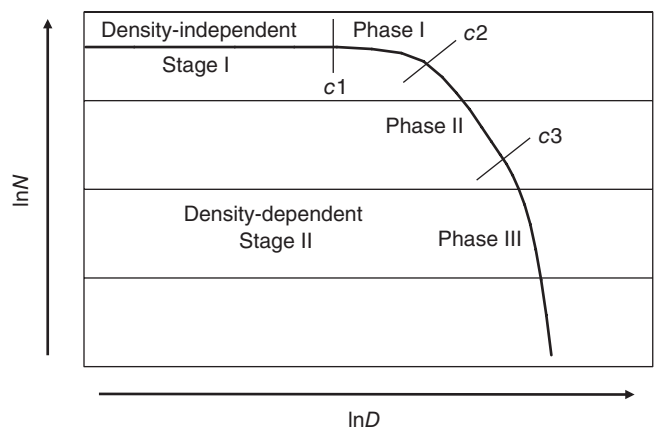

Figure 1—Depiction of a size-density trajectory for an individual stand. The two stages of stand development are shown—density-independent mortality and density-dependent mortality. Within the density-dependent mortality stage, or when self-thinning is occurring, three phases of stand development are shown. The join points (*c*1, *c*2, *c*3) used in equation (2) to differentiate stages and phases of stand development in size-density trajectories are depicted.

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the overall self-thinning stage, when density-dependent mortality is occurring, three phases are generally assumed. The first phase is represented by a nonlinear approach of a size-density trajectory, followed by a linear portion of a trajectory, and the third phase is represented by a divergence of the size-density trajectory from the linear portion. A further explanation is given below:

Phase I—Initially, the self-thinning stage of stand development can be represented by a curved approach of a size-density trajectory to a linear portion of self-thinning (or the MSDR dynamic thinning line) (fig. 1—phase I). During this initial component of self-thinning, mortality is less than the mortality at maximum competition and thus the trajectory has a concave shape (del Rio and others 2001, Harms and others 2000, Poage and others 2007).

Phase II—With increases in tree sizes and the death of other trees, eventually the size-density trajectory is assumed to become linear (fig. 1—phase II) where an increase in *D* (inches) is a function of the stand's maximum value of Reineke's (1933) stand density index (SDI), the change in *N*, and the MSDR dynamic thinning line slope (*b*). Known as the MSDR dynamic thinning line phase of stand development (Weller 1990), this is when a stand is fully stocked (del Rio and others 2001) and Reineke's SDI remains relatively constant. Reineke's SDI is expressed as:

$$\mathrm{SDI} = N(D/10)^b \qquad (1)$$

where
SDI = Reineke's SDI
N = trees per acre
D = quadratic mean diameter (inches)
b = exponent of Reineke's equation, equivalent to the MSDR dynamic thinning line slope on the ln-ln scale

Phase III—Eventually, as trees die, the residual trees cannot continue to fully occupy canopy gaps and the trajectory diverges (fig. 1—phase III) from the MSDR dynamic thinning line (Bredenkamp and Burkhart 1990, Cao and others 2000, Zeide 1995). The divergence from the MSDR dynamic thinning line has been depicted both as a line (Christensen and Peet 1981, Lonsdale 1990, Peet and Christensen 1980) and as a curve (Cao and others 2000, Zeide 1985). Whether the divergence can be depicted as linear or a curve is probably related to the amount of time since the occurrence of the MSDR dynamic thinning line phase (Cao and others 2000, Christensen and Peet 1981, Weller 1991). For example, in figure 1, the time period immediately after the MSDR dynamic thinning line phase of stand development shows an approximate linear divergence. With time, as mortality continues, the divergence becomes curvilinear eventually encompassing the disintegration portion of stand development.

Over the entire range of the density-dependent mortality stage of stand development the relationship between lnN and lnD is curvilinear; however, it is commonly assumed there is a linear phase (or portion) during self-thinning (Cao and others 2000, del Rio and others 2001, Hynynen 1993, Johnson 2000, Monserud and others 2004, Poage and others 2007, VanderSchaaf and Burkhart 2008, Yang and Titus 2002, Zeide 1985).

METHODS

Data

Tree- and plot-level measurements were obtained from a spacing trial maintained by the Loblolly Pine Growth and Yield Research Cooperative at Virginia Polytechnic Institute and State University. The spacing trial was established on four cutover sites—two in the upper Atlantic Coastal Plain and two in the Piedmont. There is one Coastal Plain site in North Carolina and one in Virginia while both Piedmont sites are in Virginia. Three replicates of a compact factorial block design were established at each location in either 1983 or 1984. Sixteen initial planting configurations were established ranging in densities from 2,722 to 302 *N*, a variety of planting distances between and within rows was used (not all spacings were square). Thus, a total of 192 experimental units were established when combining all 4 sites (4 sites by 3 replications by 16 planting configurations). For the planting densities of 2,722, 1,210, 680, and 302 *N* there was 1 plot established for a particular site and replication combination; for the planting densities of 1,815, 1,361, 605, and 453 *N* there were two plots established; and for the planting density of 907 *N* there were 4 plots established. Seed sources were of genetically improved stock considered superior for a particular physiographic region; for both sites within a particular physiographic region the same genetic stock was used. All seedlings planted at each location were lifted from the same nursery and were 1-0 stock. See Sharma and others (2002) for a more comprehensive description of the studies.

Measurements of *D* and *N* were conducted annually between ages 5 and 21 on one of the Coastal Plain sites and to age 22 on the other site. On the Piedmont sites, measurement ages end at 18 at one location and 21 at the other. At the latter Piedmont site, one replication had measurements to 22 years of age. Site quality was quantified using site index defined as the average height of all trees with diameters larger than *D* for the planting densities of 907, 680, and 605 *N* by replication. Plots intermediate in stand density were used when estimating site index for each replication in order to avoid any possible effects of high or low number of *N*. A site index equation found in Burkhart and others (2004) was used to project dominant height forward to base age 25. Table 1 contains summaries of plot-level characteristics for the entire dataset.

Using Segmented Regression to Estimate Stages and Phases of Stand Development

A segmented regression model was developed based on the two stages of stand development and the three phases of self-thinning to objectively determine what observations of size-density trajectories are within particular stages and phases. The segmented regression model can be written as:

$$\begin{aligned}\ln N = {} & (b1)J1 + (b1 + b2[\ln D - c1]^2)J2 \qquad (2)\\ & + (b1 + b2[c2 - c1]^2 + b3[\ln D - c2])J3\\ & + (b1 + b2[c2 - c1]^2 + b3[c3 - c2] + b4[\ln D - c3])J4\end{aligned}$$

Table 1—Plot-level characteristics for the entire dataset (n = 2977)

Variable	Minimum	Mean	Maximum
Trees per acre	228	917	2,722
Quadratic mean diameter (inches)	1.1	5.4	10.8
Square feet of basal area per acre	0.1	122	258
Site index at base age 25 (feet)	63	68	73

where:

D = quadratic mean diameter (inches), d.b.h. was measured at 4.5 feet above the ground

$J1$, $J2$, $J3$, and $J4$ = indicator variables for the stages and phases of stand development

$J1$ = 1 if ln$D$ is within the density-independent mortality stage of stand development (stage I in fig. 1), zero otherwise

$J2$ = 1 if ln$D$ is within the curved approach to the MSDR dynamic thinning line phase of self-thinning (phase I of stage II in fig. 1), zero otherwise

$J3$ = 1 if ln$D$ is within the MSDR dynamic thinning line phase of self-thinning (phase II of stage II in fig. 1), zero otherwise

$J4$ = 1 if ln$D$ is within the divergence phase of self-thinning (phase III of stage II in fig. 1), zero otherwise, and other variables as previously defined

Seven parameters were estimated for each planting density (VanderSchaaf 2006, VanderSchaaf and Burkhart 2008); one for the initial component where no density-related mortality occurs ($b1$), one for the curved approach to the MSDR dynamic thinning line ($b2$), one for the MSDR dynamic thinning line ($b3$), one for the divergence from the MSDR dynamic thinning line ($b4$), and three for the join points to estimate at what ln$D$ self-thinning begins ($c1$), at what ln$D$ the MSDR dynamic thinning line phase of stand development begins ($c2$), and at what ln$D$ the divergence from the MSDR dynamic thinning line begins ($c3$).

Convergence criteria were not met in parameter estimation of equation (2) for the planting densities of 453 and 302 N. In previous reports (VanderSchaaf 2006, VanderSchaaf and Burkhart 2008), a system of simultaneously estimated equations were used to estimate at what ln$D$ and ln$N$ planting density–specific MSDR dynamic thinning lines begin and terminate. This paper extends the work of those publications by using a system of simultaneously estimated equations to also estimate at what ln$D$ self-thinning begins and the size-density trajectory coefficients of various stages and phases ($b2$, $b3$, and $b4$). Additionally, this work presents estimates of the N after density-independent (or random) mortality ($b1$). The seven values for each dependent variable for the planting densities ranging from 2,722 to 605 N as estimated using segmented regression are presented in table 2. ln$N$ values by planting density were derived using the parameter estimates of the segmented regression models as presented in VanderSchaaf (2006) and shown in table 2.

Model Forms and Parameter Estimation

Due to a limited number of observations for model fitting, making it difficult to estimate the cross-equation random error correlation matrix, parameters of two distinct simultaneous systems of linear regression equations were estimated. The first system was used to model the density-independent mortality stage of stand development (stage I in fig. 1), and the second system was used to model phases I and II of the self-thinning stage of stand development (phases I and II of stage II in fig. 1). Phase III, or the divergence phase of the self-thinning stage of stand development (phase III of stage II in fig. 1), was modeled separately.

A simultaneous parameter estimation method presented in Borders (1989) was used for the two sets of simultaneous equations. The density-independent mortality stage linear system of equations is:

$$b1 = b01 + b11 \ln(N0) \tag{3}$$

$$\ln Ds = b02 + b12 \ln(b1) \tag{4}$$

The density-dependent mortality stage linear system of equations is:

$$b2 = b03 + b13 \ln(N0) \tag{5}$$

$$\ln Db = b04 + b14 \ln(N0) \tag{6}$$

$$\ln De = b05 + b15 \ln Db \tag{7}$$

$$\ln Nb = b06 + b16 \ln Db \tag{8}$$

$$\ln Ne = b07 + b17 \ln De \tag{9}$$

The equation form used to estimate the slope of the divergence phase of size-density trajectories is:

$$b4 = b08 + b18 \ln(N0) \tag{10}$$

where

lnDs = ln$D$ corresponding to the initiation of the self-thinning stage of stand development (7 $c1$ estimates from table 2)

lnDb = ln$D$ corresponding to the initiation of a particular MSDR dynamic thinning line (7 $c2$ estimates from table 2)

lnDe = ln$D$ corresponding to the termination of a particular MSDR dynamic thinning line (7 $c3$ estimates from table 2)

lnNb = ln$N$ corresponding to the initiation of a particular MSDR dynamic thinning line

lnNe = ln$N$ corresponding to the termination of a particular MSDR dynamic thinning line

$N0$ = planting density (trees per acre)

$b0i$, $b1i$ = parameters to be estimated

Table 2—Dependent variable values used in estimating parameters of equations (3) to (10) as obtained from segmented regression model results presented in VanderSchaaf (2006)

Planting density	Stage I	Stage II							
		Curved approach		MSDR dynamic thinning line					Divergence
	*b*1	ln*Ds* (*c*1)	*b*2	ln*Db* (*c*2)	ln*Nb*	*b*3	ln*De* (*c*3)	ln*Ne*	*b*4
per acre									
2,722	7.8833	1.1103	–1.8300	1.3737	7.7563	–1.8852	1.4855	7.5456	–3.7231
1,815	7.4773	1.2228	–1.3237	1.5691	7.3186	–1.6777	1.6649	7.1578	–3.4829
1,361	7.1886	1.3536	–1.3897	1.6535	7.0636	–1.1109	1.7335	6.9747	–2.7154
1,210	7.0648	1.3868	–1.1343	1.7104	6.9460	–1.4331	1.8228	6.7849	–4.3940
907	6.7691	1.5551	–1.0541	1.8382	6.6846	–1.7074	1.8940	6.5893	–1.9898
680	6.5001	1.5554	–0.5454	1.9674	6.4075	–1.4385	2.0994	6.2176	–13.7855
605	6.3635	1.6532	–0.5607	2.0320	6.2830	–1.6226	2.0908	6.1876	–2.2319

The system of equations will avoid illogical predictions of the response variables, e.g., ln*Db* estimated to be greater than ln*De*. Using a ln transformation of planting density to predict *b*1, *b*2, ln*Db*, and *b*4 allows for a nonlinear relationship between these variables. For equation (10), the divergence slope for the 680 N planting density was removed. Hence, for equation (10), $n = 6$ and for all other equations, $n = 7$. Parameter estimates (SAS 1989) are given in table 3.

Rather than directly modeling the MSDR dynamic thinning line slope (*b*3), an alternative formula as shown in VanderSchaaf (2006) and VanderSchaaf and Burkhart (2008) was used:

$$b3 = (\ln Nb - \ln Ne)/(\ln Db - \ln De) \qquad (11)$$

This helps to reduce the number of dependent variables in the simultaneous estimation equation system.

RESULTS AND DISCUSSION

All parameter estimates were significant at an alpha = 0.05 level except for equations (3) and (10). When excluding the 680 planting density observation, there is a slight trend between *b*4 and planting density. For the purposes of this paper, when depicting size-density trajectories (fig. 2), a divergence from the MSDR dynamic thinning line (phase III of stage II—fig. 1) was included as estimated using equation (10).

Based on equations (3) through (11) and the data used in fitting those equations and the original segmented regression models (VanderSchaaf 2006, VanderSchaaf and Burkhart 2008), the MSDR boundary level differs relative to planting density (fig. 2). Although the MSDR dynamic thinning lines appear to be short in duration, the level of intraspecific competition among trees when a stand's size-density trajectory is within the linear phase of self-thinning can be quite intense and a stand may stay in this phase for many years. It should be realized that the predicted size-density trajectories depicted in figure 2 provide no information about the rate of change in *D* across time. Rates of change in *D* can be relatively large during the density-independent mortality stage and the divergence phase of self-thinning.

When using equation (3), which estimates the *N* at which self-thinning begins, predicted percent survival ranged from 96.5 to 97.5 percent and varied little relative to planting density, as expected. Initial survival depends on seedling care, planting practices or depth, localized interspecific competition, diseases or infestations from the nursery, etc., and how these factors interact with local climatic conditions. Predicted survival rates from these spacing trials are not necessarily indicative of those that might be realized in operational plantings.

Due to divergences from each individual stand's linear boundary, the predicted size-density trajectories suggest that for the trajectories of the stands (plots) used in model fitting, e.g., loblolly pine plantations in the Atlantic Coastal Plain and Piedmont regions, the maximum ln*N* for a given ln*D* across all planting densities can be a conglomeration from several stands. VanderSchaaf and Burkhart (2007) noted this behavior may lead to a MSDR species boundary line slope that is not representative of how, on average, individual stands self-thin during the linear phase of the density-dependent mortality stage. For instance, for the four size-density trajectories presented in figure 2, a step interval

Table 3—Parameter estimates of equations (3) and (4) and (5) through (9) that were simultaneously estimated and equation (10)

	Intercept			Slope				
Equation	Estimate	Std. error	Sign.	Estimate	Std. error	Sign.	RMSE	Adj. R^2
(3)	–0.07677	0.0408	0.1184	1.006456	0.00576	<0.0001	0.0344	0.9691
(4)	6.319165	0.3584	<0.0001	–2.52196	0.1838	<0.0001	0.00757	0.9998
(5)	4.748849	0.6430	0.0007	–0.83049	0.0907	0.0003	0.1293	0.9208
(6)	4.769806	0.0717	<0.0001	–0.42948	0.0101	<0.0001	0.0150	0.9958
(7)	0.156933	0.0608	0.0494	0.962787	0.0344	<0.0001	0.0302	0.9818
(8)	10.81729	0.0843	<0.0001	–2.24479	0.0481	<0.0001	0.0305	0.9965
(9)	10.84589	0.0806	<0.0001	–2.22531	0.0437	<0.0001	0.0290	0.9966
(10)	4.910022	4.9455	0.3770	–1.1177	0.6894	0.1803	0.8094	0.2456

Std. error is the standard error of the estimate; RMSE = root mean squared error.

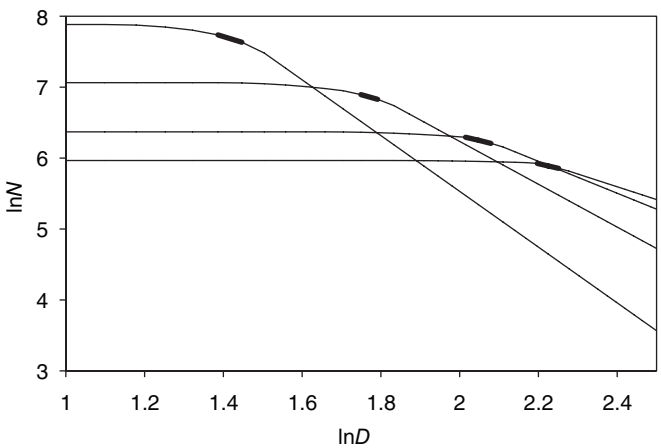

Figure 2—Predicted size-density trajectories using equations (3) through (11) for planting densities of 405 (beyond the planting density range of the model fitting dataset), 605, 1,205, and 2,722 seedlings per acre.

was used to create observations from the MSDR dynamic thinning line portion. Combining all created observations together into a single linear regression model resulted in ln*N* = 10.80946–2.21884 ln*D*. This MSDR species boundary line slope [defined by VanderSchaaf and Burkhart (2007) as the MSDR species boundary line I slope] is not representative of the slopes of the individual size-density trajectories as estimated using the density-dependent mortality stage system of equations with an average of –1.48316. VanderSchaaf and Burkhart (2010) noted similar behavior for stands differing in planting densities and site qualities.

These predicted size-density trajectories can be used to help determine rates of density-independent mortality, when self-thinning is expected to occur, and as constraints or verifications of both empirical and process-based model size-density trajectories.

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LITERATURE CITED

Borders, B.E. 1989. Systems of equations in forest stand modeling. Forest Science. 35: 548–556.

Bredenkamp, B.V.; Burkhart, H.E. 1990. An examination of spacing indices for *Eucalyptus grandis*. Canadian Journal Forest Research. 20: 1909–1916.

Burkhart, H.E.; Farrar, K.D.; Amateis, R.L.; Daniels, R.F. 2004. PTAEDA 3.1: simulation of individual tree growth, stand development and economic evaluation in loblolly pine plantations. Blacksburg, VA: Virginia Polytechnic Institute and State University, School of Forestry and Wildlife Resources. 23 p.

Cao, Q.V.; Dean, T.J.; Baldwin, V.C. 2000. Modeling the size-density relationship in direct-seeded slash pine stands. Forest Science. 46: 317–321.

Christensen, N.L.; Peet, R.K. 1981. Secondary forest succession on the North Carolina Piedmont. In: West, D.C.; Shugart, H.H.; Botkin, D.B., eds. Forest succession: concepts and application. New York: Springer-Verlag: 230–245.

del Rio, M.; Montero, G.; Bravo, F. 2001. Analysis of diameter-density relationships and self-thinning in non-thinned even-aged Scots pine stands. Forest Ecology and Management. 142: 79–87.

Drew, T.J.; Flewelling, J.W. 1979. Stand density management: an alternative approach and its application to Douglas-fir plantations. Forest Science. 25: 518–532.

Harms, W.R.; Whitesell, C.D.; DeBell, D.S. 2000. Growth and development of loblolly pine in a spacing trial planted in Hawaii. Forest Ecology and Management. 126: 13–24.

Hynynen, J. 1993. Self-thinning models for even-aged stands of *Pinus sylvestris*, *Picea abies*, and *Betula penula*. Scandinavian Journal of Forest Research. 8: 326–336.

Johnson, G. 2000. ORGANON calibration for western hemlock project: maximum size-density relationships. Willamette Industries, Inc. Internal Res. Rep. 8 p. http://www.cof.orst.edu/cof/fr/research/organon/orgpubdl.htm. [Date accessed: December 22, 2008].

Landsberg, J.J.; Waring, R.H. 1997. A generalised model of forest productivity using simplified concepts of radiation-use efficiency, carbon balance and partitioning. Forest Ecology and Management. 95: 209–228.

Lonsdale, W.M. 1990. The self-thinning rule: dead or alive? Ecology. 71: 1373–1388.

McCarter, J.B.; Long, J.N. 1986. A lodgepole pine density management diagram. Western Journal of Applied Forestry. 1: 6–11.

Monserud, R.A.; Ledermann, T.; Sterba, H. 2004. Are self-thinning constraints needed in a tree-specific mortality model? Forest Science. 50: 848–858.

Peet, R.K.; Christensen, N.L. 1980. Succession: a population process. Vegetatio. 43: 131–140.

Poage, N.J.; Marshall, D.D.; McClellan, M.H. 2007. Maximum stand-density index of 40 western hemlock-Sitka spruce stands in southeast Alaska. Western Journal of Applied Forestry. 22: 99–104.

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

SAS Institute Inc. 1989. SAS/ETS user's guide. Version 6. 4th ed. Cary, NC: SAS Institute Inc.

Sharma, M.; Burkhart, H.E.; Amateis, R.L. 2002. Modeling the effect of density on the growth of loblolly pine trees. Southern Journal of Applied Forestry. 26: 124–133.

Smith, N.J.; Hann, D.W. 1984. A new analytical model based on the -3/2 power rule of self-thinning. Canadian Journal of Forest Research. 14: 605–609.

Turnblom, E.C.; Burk, T.E. 2000. Modeling self-thinning of unthinned Lake States red pine stands using nonlinear simultaneous differential equations. Canadian Journal of Forest Research. 30: 1410–1418.

VanderSchaaf, C.L. 2006. Modeling maximum size-density relationships of loblolly pine (*Pinus taeda* L.) plantations. Blacksburg, VA: Virginia Polytechnic Institute and State University. 171 p. Ph.D. dissertation.

VanderSchaaf, C.L.; Burkhart, H.E. 2007. Comparison of methods to estimate Reineke's maximum size-density relationship species boundary line slope. Forest Science. 53: 435–442.

VanderSchaaf, C.L.; Burkhart, H.E. 2008. Using segmented regression to estimate stages and phases of stand development. Forest Science. 54: 167–175.

VanderSchaaf, C.L.; Burkhart, H.E. 2010. Comparing methods to estimate Reineke's maximum size-density relationship species boundary line slope. 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: 300–405.

Weller, D.E. 1990. Will the real self-thinning rule please stand up? – A reply to Osawa and Sugita. Ecology. 71: 1204–1207.

Weller, D.E. 1991. The self-thinning rule: dead or unsupported? – A reply to Lonsdale. Ecology. 72: 747–750.

Williams, R.A. 1994. Stand density management diagram for loblolly pine plantations in north Louisiana. Southern Journal of Applied Forestry. 18: 40–45.

Yang, Y.; Titus, S.J. 2002. Maximum size-density relationship for constraining individual tree mortality functions. Forest Ecology and Management. 168: 259–273.

Zeide, B. 1985. Tolerance and self-tolerance of trees. Forest Ecology and Management. 13: 149–166.

Zeide, B. 1995. A relationship between size of trees and their number. Forest Ecology and Management. 72: 265–272.

PINE SILVICULTURE SESSION II

Row thinning in a young loblolly pine plantation on private land in Bradley County, Arkansas. (Photo by James M. Guldin)

VOLUME AND CROWN CHARACTERISTICS OF JUVENILE LOBLOLLY PINE GROWN AT VARIOUS RATIOS OF BETWEEN AND WITHIN ROW SPACINGS

John R. Britt and Jason P. Reynolds[1]

Abstract—In plantation forestry, several silvicultural treatments can be row oriented. When rows are treated individually, planting trees in wider rows may result in lower silvicultural treatment cost, facilitate future operations, such as thinning and fire fighting, and provide a longer period with open canopy conditions. All these scenarios could provide benefit to landowners, depending on management objectives. Few studies have considered the effects of asymmetrical spacing on tree growth, stand yield, or wood quality. This study examines tree and stand attributes for loblolly pine (*Pinus taeda* L.) grown at five rectangular spacings for a common stand density. The treatments include spacings of 9 by 8 feet, 12 by 6 feet, 15 by 4.8 feet, 18 by 4 feet, and 24 by 3 feet; these planting arrangements represent between-row to within-row spacing ratios of 1 to 1, 2 to 1, 3 to 1, 4 to 1, and 8 to 1, respectively. Tree and stand volumes and branching characteristics after the ninth-growing season are presented.

INTRODUCTION

Initial planting density of trees has long been of interest to managers of plantation forests; however, the spatial arrangement has not received the same degree of study. Many "spacing" studies for southern pines have focused on the volume response and stand dynamics of plantations grown at various densities. Only a few studies have reported the response of plantations to various spacing arrangements.

The literature addressing asymmetrical arrangement (rectangularity) of southern pine plantations is limited (Lock 1977). Some agroforestry applications provide insight although the extremely rectangular spatial arrangements would possibly be beyond practical limits for commercial timber operations. Lewis and others (1985) report no statistical differences between an 8- by 12-foot and a 4- by 24-foot spacing for survival, height, diameter, and volume in 13-year-old slash pine (*Pinus elliottii* Engelm). It should be noted that these stands were in a very early stage of development with approximately 50 square feet of basal area. Sharma and others (2002) report survival, height, diameter, basal area, and volume were not statistically different nor were the distributions of height and diameter between a nominal 1-to-1 and 3-to-1 spacing ratio at age 16 years in loblolly pine (*P. taeda* L.) stands located in the Southeastern United States. At age 19 years results from the project previously cited indicate that rectangularity had no significant effect on potential timber products. The 3-to-1 spacing ratio did have a larger maximum branch size but this was not attributed to spacing arrangement (Amateis and others 2004).

Rectangularity comparisons have been published with other forest tree species, as well. Rectangular arrangements at 1-to-1 and 4-to-1 ratios with half-sib maritime pine (*P. pinaster* Ait.) in southwestern France showed no statistical differences in height, diameter, or basal area at age 16 years (von Euler and others 1992). In Lithuania, rectangularity of 4 to 1 or 5 to 1 had an insignificant influence on stem quality of *Pinus sylvestris* (Malinauskas 2003). With *Eucalyptus nitens* (Deane and Maiden), no differences in growth or the size of the largest branch in the lower 6 m were detected between square and rectangular spacing ratio up to 2.5 to 1 (Gerrand and Neilsen 2000). An objective of this study was to test the effects of varying spatial arrangement on tree and stand level attributes of plantation-grown loblolly pine trees.

METHODS

Site Description and Study Establishment

This study was established in December 1999 on an old field site located in Randolph County, GA. Soils are of the Lakeland sand and Lucy loamy sand series with slopes of <3 percent. Prior to study establishment, the site had been fallow for a number of years and had no hardwood trees. Site preparation consisted of a broadcast fertilization with 500 pounds per acre of 10-10-10 fertilizer including micronutrients and subsoiling in two directions at 90-degree intersections on 3-foot centers. Spacing treatments included 9 by 8 feet, 12 by 6 feet, 15 by 4.8 feet, 18 by 4 feet, and 24 by 3 feet. These planting arrangements represent between-row to within-row spacing ratios of 1 to 1, 2 to 1, 3 to 1, 4.5 to 1, and 8 to 1, respectively. All of these spacing ratios represent 605-trees-per-acre density. All treatments were replicated four times in a randomized complete block design, with blocking around soils series.

Two seedlings of a half-sib Atlantic Coast loblolly pine family were planted at each planting location within the study. In July of the first growing season, one seedling was randomly removed where both seedlings survived. Herbaceous weed control was applied in 6-foot bands during March 2000 and broadcast in March 2001. An additional fertilization treatment was broadcast applied in August 2002 with 47, 20, and 39 pounds per acre of nitrogen, phosphorus, and potassium, respectively.

Measurement and Analysis

During the ninth dormant season, all trees were measured for total height; diameter at breast height; and the presence of stem rust, sweep or crook, forked, and broken tops. A subset

[1] Owner, John Britt & Associates, Fortson, GA; Forest Resource Planner, Hancock Timber Resource Group, Charlotte, NC, respectively.

of 10 healthy trees was randomly selected in each plot for crown and branch measurements, consisting of crown length, crown width across and along rows, number of branches (live and dead) in the first 17.5 feet, and the diameter of the 2 largest branches in the first 8 feet. Means were calculated for each variable and subjected to polynomial regression. Trees with broken tops were excluded from the analysis for height means. Cubic-foot-volume was calculated for each tree using equations for site-prepared loblolly pine developed by Burkhart and others (1987). Individual tree volumes were summed by plot and expanded to estimate per-acre volumes.

RESULTS

Stem and Volume Attributes

Mean height did not differ among spacing treatments and averaged 30.5 feet (table 1). Mean diameter ranged from 5.21 inches for the widest spacing to 5.66 inches for the square spacing. Individual tree volume increased from 2.4 cubic feet to 2.8 cubic feet per tree as spacing ratios decreased. Volume per acre ranged from about 1,400 cubic feet at wider spacing ratios to over 1,600 cubic feet in the square spacing treatment.

Branching and Crown Attributes

The number of branches in the first log averaged 40 per tree and was not different among spacing treatments (table 2). Basal diameter of the largest branch did not differ among treatments and averaged 1.44 inches. The second largest branch diameter was largest for the 18- by 4-foot spacing but this was only 0.03 inches larger than for the 9- by 8-foot spacing.

Crown width between rows differed by treatments; the widest spacing had the widest between-row crown width. Both between-row and within-crown width differed by treatment, yet differences were not directly proportional to row spacing; similar results were reported by Sharma and others (2002). Crowns were longest in the widest row spacing and decreased as row spacing decreased.

DISCUSSION

Establishing plantations with greater rectangularity may provide economic, operations, and nontimber advantages over planting on more square spacing. Plantation establishment often includes treatments applied to rows. As the space between planting rows is increased the cost associated with establishment can be reduced (VanderSchaaf and South 2004).

In locales with strong pulpwood markets thinning operations provide the benefit of intermediate income. Contemporary commercial thinnings in plantations generally include some form of row removal and selection from the remaining trees. Rows are removed at specific intervals to allow access to inferior trees within the remaining rows, leaving trees of superior quality. In row thinnings, potential higher value trees are removed in proportion to the row removal interval. Plantations established using wider row spacing may offer the ability to access inferior trees without removing an entire row. Nontimber related advantages to wider row spacing include a delay in crown closure and the prolonged presence of early successional vegetation, as well as less soil disturbance on sites with potential for erosion.

Table 1—Stem and volume attributes for trees grown at different degrees of rectangularity

Spacing	D.b.h.	Height	Volume per tree	Volume per acre
	inches	*feet*	*cubic feet*	
9 by 8 feet	5.66	31.1	2.8	1,628
12 by 6 feet	5.52	30.3	2.6	1,514
15 by 4.8 feet	5.41	30.3	2.5	1,396
18 by 4 feet	5.42	30.3	2.6	1,434
24 by 3 feet	5.21	30.5	2.4	1,413
Polynomial contrast	*probability of a greater F-value*			
Linear	0.0085	0.7208	0.0442	0.1206
Quadratic	0.4751	0.2595	0.3518	0.1339
Cubic	0.5458	0.4576	0.4065	0.5412
Lack of fit	0.8212	0.9131	0.8698	0.5892

D.b.h. = diameter at breast height.

Table 2—Branch and crown attributes for trees grown at different degrees of rectangularity

Spacing	Between-row crown width	Within-row crown width	Crown length	Number of branches	Largest branch	Second largest branch
	feet				*feet*	
9 by 8 feet	10.3	9.7	21.6	40.2	1.40	1.21
12 by 6 feet	11.8	9.0	21.9	41.1	1.52	1.21
15 by 4.8 feet	13.1	8.58	23.3	41.9	1.43	1.20
18 by 4 feet	14.6	8.43	23.7	38.5	1.45	1.24
24 by 3 feet	15.4	8.1	23.6	38.4	1.38	1.04
Polynomial contrast	*probability of a greater F-value*					
Linear	0.0001	0.0169	0.0074	0.1484	0.4048	0.0076
Quadratic	0.0004	0.1562	0.0252	0.8403	0.4599	0.0829
Cubic	0.3928	0.6284	0.4990	0.1502	0.4066	0.4711
Lack of fit	0.7573	0.9900	0.3604	0.3986	0.2609	0.5397

Preliminary results in this study show volume per acre to be slightly lower with wider rows. However, the establishment cost savings and possibility of pure selection thinning for all rows may outweigh the slight volume loss at age 8. As more data are available from plantations with wider row spacings, managers can determine the benefits based on their management objectives.

LITERATURE CITED

Amateis, R.L.; Radtke, P.J.; Hansen, G.D. 2004. The effects of spacing rectangularity on stem quality in loblolly pine plantations. Canadian Journal of Forest Research. 34(2): 498–501.

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. FWS-1-87. Blacksburg, VA: Virginia Polytechnic Institute and State University, Department of Forestry. 28 p.

Gerrand, A.M.; Neilson, W.A. 2000. Comparing square and rectangular spacings in *Eucalyptus nitens* using Scotch plaid design. Forest Ecology and Management. 129: 1–6.

Lewis, C.E.; Tanner, G.W.; Terry, W.S. 1985. Double vs. single-row pine plantations for wood and forage production. Southern Journal of Applied Forestry. 9(1): 55–61.

Lock, W.H. 1977. The effects of square and rectangular espacements on loblolly pine on an upland Piedmont site. Raleigh, NC: North Carolina State University, School of Forestry. 49 p. M.S. thesis.

Malinauskas, A. 2003. Influence of initial density and planting design on the quality of butt logs in Scots pine (*Pinus sylvestris* L.) plantations. Baltic Forestry. 9(2): 10–15.

Sharma, M.; Burkhart, H.E.; Amateis, R.L. 2002. Spacing rectangularity effect on the growth of loblolly pine plantations. Canadian Journal of Forest Research. 32(8): 1451–1459.

VanderSchaaf, C.L.; South D.B. 2004. Rectangular spacing: an economic benefit? Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 437–440.

von Euler, F.; Baradat, P.; Lemoine, B. 1992. Effects of plantation density and spacing on competitive interactions among half-sib families of maritime pine. Canadian Journal of Forest Research. 22: 482–489.

EFFECT OF APPLICATION TIMING ON EFFICACY OF SITE PREPARATION TREATMENTS USING CHOPPER® GEN2™

A.W. Ezell, J.L. Yeiser, D.K. Lauer, and H.E. Quicke[1]

Abstract—Chopper® GEN2™ is a new imazapyr product for use in forestry site preparation. A single treatment (32 ounces of Chopper® GEN2™ per acre) was applied at three timings on three sites (Louisiana, Mississippi, and Virginia) to test the effect of application timing on treatment efficacy. Hardwood control was excellent for all applications. Pine growth varied by site, but all treatments resulted in excellent pine growth. Pine stem volume was 5 to 10 times greater in treated plots as compared to untreated plots.

INTRODUCTION

Site preparation continues to be the preeminent use of herbicides in the South. As this is typically a notable expense, it is very important that the most cost-effective applications be made. Treatment efficacy is therefore a primary concern.

Chopper® GEN2™ is the most recent formulation of imazapyr to be labeled for forestry site preparation in the South. While it contains the same active ingredient (imazapyr) as Arsenal AC or Chopper®, it is a different product and can provide different results in field applications. As is the case with most herbicides used in forestry, the timing of application can be important. Also, while short-term results are always important, long-term control and seedling growth are the true tests of site preparation.

The objectives of this study were as follows: (1) to evaluate the effect of application timing on the efficacy of Chopper® GEN2™ and (2) to evaluate the growth response of loblolly pine (*Pinus taeda* L.) seedlings following the application timing.

STUDY SITES

The study was installed at sites near Appomattox, VA; Allen, LA; and Starkville, MS. At the Virginia site, the treatments were applied soon after harvest. The principal hardwood species present were red maple (*Acer rubrum* L.), blackgum (*Nyssa sylvatica* Marsh.), white oak (*Quercus alba* L.), yellow-poplar (*Liriodendron tulipifera* L.), black cherry (*Prunus serotina* Ehrh.), hickory (*Carya* spp.), scarlet oak (*Q. coccinia* Muench.), and *Vaccinuium* spp.

The Louisiana site was bedded prior to treatment application. At the time of application, there was little hardwood competition (<4 percent cover). The principal species present were American beautyberry (*Callicarpa americana* L.) and sumac (*Rhus* spp.).

The Mississippi site had been harvested more than a year prior to treatment application. The area had heavy hardwood cover of 2,500 to 3,000 hardwood stems per acre. The principal species present were southern red oak (*Q. falcata* Michaux), cherrybark oak (*Q. pagoda* Raf.), post oak (*Q. stellata* Wang.), blackgum, red maple, and *Rubus* spp.

TREATMENTS

A single treatment was used in the study with three application timings. The treatment consisted of 32 ounces of Chopper® GEN2™ per acre with 1 percent v/v methylated seed oil. The three application timings were as follows: treatment #1—applied June 28 through July 1, 2006; treatment #2—applied August 13–17, 2006; and treatment #3—applied September 28–30, 2006. Total spray volume was 10 g/acre. Each site had untreated control plots in addition to the treated areas.

EXPERIMENTAL DESIGN

Each treatment was replicated four times at each site in a randomized complete block design. Each replication plot was 91 by 91 feet (0.19 acre).

PLANTING

All plots were planted with 1-0, bare-root loblolly pine seedlings in December 2006. Tree spacing was 6 by 11 feet. All treated plots received an herbaceous weed control treatment of 4 ounces Arsenal AC and 2 ounces Oust® XP per sprayed acre in March 2007.

EVALUATIONS

Vegetation assessments were completed in June and August 2007. At those timings hardwood control and percent ground cover of grasses, broadleaf forbs, and vines were recorded. Pine seedlings were measured in December 2007 with total height and groundline diameter (GLD) recorded.

RESULTS

Competition Control

The results for competition control as recorded in August 2007, one growing season after treatment (GSAT), can

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Table 1—Average percent cover by vegetation type in August 2007 (1GSAT), Allen, LA

Treatment date	Woody	Herb	Vine	Total
	---------- *percent* ----------			
July 1	3 a	28 b	3 a	34 b
August 1	1 ab	26 b	3 a	29 b
September 30	1 b	28 b	2 a	31 b
None	4 a	86 a	1 a	97 a

Values in a column followed by the same letter do not differ at alpha = 0.05.

Table 2—Average percent cover by vegetation type in August 2007 (1GSAT), Appomattox, VA

Treatment date	Woody	Herb	Vine	Total
	---------- *percent* ----------			
July 1	2 b	12 a	0 a	17 b
August 15	2 b	20 a	0 a	20 b
September 30	3 b	18 a	0 a	21 b
None	49 a	18 a	0 a	73 a

Values in a column followed by the same letter do not differ at alpha = 0.05.

Table 3—Average percent cover by vegetation type in August 2007 (1GSAT), Starkville, MS

Treatment date	Woody	Herb	*Rubus*	Total
	---------- *percent* ----------			
July 1	3 bc	43 a	25 a	72 b
August 15	1 c	44 a	16 ab	61 bc
September 30	5 b	23 a	15 ab	48 c
None	40 a	41 a	3 b	99 a

Values in a column followed by the same letter do not differ at alpha = 0.05.

be found in tables 1, 2, and 3. Control of hardwoods was excellent at all three sites. While the Louisiana site did not have much woody competition, the Mississippi and Virginia sites both had 40 percent or more coverage by woody species and the treatments resulted in significant reductions (1 to 5 percent cover). Treatment timing had no significant effect on hardwood control at Virginia or Louisiana, but the August application timing was significantly better than the late September timing in Mississippi, but the difference was only 1 percent vs. 5 percent coverage (both treatments provided excellent control).

Herbaceous control at the August 2007 evaluations did not differ significantly between untreated and treated plots in Virginia or Mississippi, although the late September application timing plots had about 20 percentage points less herbaceous cover than the other treatments in Mississippi. This lack of difference in herbaceous weed control is not surprising as the evaluation date is almost 1 year after all treatments. The plots did have some residual weed control earlier in the growing season which was important to a seedling establishing a root system, but the control was diminished by August. As 2007 was an especially droughty year across much of the South, competition control was very important. The significant difference in the treated vs. untreated plots in Louisiana at the August evaluation can be attributed more to the intense herbaceous pressure on the site (86 percent cover in untreated areas) than to a total lack of herbaceous cover in treated plots (26 to 28 percent).

Vines were not a problem at the Virginia or Louisiana sites (zero to 3 percent cover). However, *Rubus* was a significant component of cover at the Mississippi site. By controlling the hardwoods and herbaceous (short-term) competition, *Rubus* was released to increase ground coverage.

Pine Response

The pines in this study will be measured for a prolonged period, and this paper presents only the initial results. Pine survival data is found in table 4. Pines survived well at all sites and the only significant difference was the survival of pines planted in the August treatment plots in Mississippi. We have no explanation for this as all the trees were planted at the same time by the same personnel at each respective site, and no microsite or other differences could be identified.

Pine heights are reported in table 5. Heights varied among sites, but trees were generally significantly taller in treated plots in Mississippi and Louisiana as compared to untreated plots. The lack of statistical difference was not surprising

Table 4—Percent pine survival by site and treatment (all reps)

Treatment date	Louisiana	Virginia	Mississippi
	---------- *percent* ----------		
July 1	86 a	89 a	86 a
August 15	86 a	85 a	63 b
September 30	90 a	86 a	89 a
None	77 a	82 a	76 ab

Values followed by the same letter do not differ at alpha = 0.05.

Table 5—Average total height by site and treatment (all reps)

Treatment date	Louisiana	Virginia	Mississippi
	---------- *feet* ----------		
July 1	2.8 a	1.3 a	1.6 ab
August 15	2.6 a	1.2 a	1.5 b
September 30	2.5 a	1.3 a	1.9 a
None	1.7 b	1.1 a	1.3 b

Values in a column followed by the same letter do not differ at alpha = 0.05.

in Virginia given the more northern site with associated expectation of less growth during the first growing season. Overall, there was very little significant difference among treatment dates at any of the sites.

Pine GLD also varied by site (table 6). The trees on the Louisiana site grew extremely well which could be attributed to the mechanical site preparation and growing season precipitation as compared to the other sites. Overall, pines in treated plots had significantly larger GLD than those in untreated plots at all locations. There was no difference among treatment dates at Louisiana or Virginia and only one difference (late September) in Mississippi.

One last measure of pine growth was to examine pine stem volume (table 7). This evaluation involves both height and diameter. The results were striking. After only one growing season, the trees in treated plots in Virginia and Mississippi were 5 times larger than trees in untreated plots, and in Louisiana, trees in treated plots were 10 times larger.

Table 6—Average groundline diameter by site and treatment (all reps)

Treatment date	Louisiana	Virginia	Mississippi
	---------- *inches* ----------		
July 1	0.77 a	0.42 a	0.28 b
August 15	0.67 a	0.37 a	0.28 b
September 30	0.68 a	0.40 a	0.36 a
None	0.29 b	0.21 b	0.17 c

Values in a column followed by the same letter do not differ at alpha = 0.05.

Table 7—Stem volume on treated vs. untreated plots

Site	Untreated	Treated	Ratio
Virginia	0.17	0.82	5X
Louisiana	0.54	5.36	10X
Mississippi	0.13	0.65	5X

SUMMARY

Overall, pines responded well to Chopper® GEN2™ site preparation and herbaceous weed control as evidenced by the 5X- to 10X-volume increases. There was no consistent trend in the response to site prep timing in Virginia. In Louisiana, survival improved by 9 percentage points and pine growth was best in the earliest site prep timing. In Mississippi, pine response was best for the latest site prep timing which is thought to be due to the lower herbaceous cover during the growing season after application.

FIFTH-YEAR PINE GROWTH RESPONSE TO WOODY RELEASE TREATMENTS IN YOUNG LOBLOLLY PLANTATIONS

A.W. Ezell, J.L. Yeiser, and L.R. Nelson[1]

Abstract—The efficacy of adding Oust® XP to woody release treatments was evaluated on second-year pine plantations in Texas, Mississippi, and South Carolina. Overall, the residual control of herbaceous weeds on these sites was excellent the growing season following application. Pine height and diameter growth was evaluated for 5 years following application. Generally, the treatments of high rates of Arsenal AC alone and all treatments with Oust® XP resulted in significantly improved height and diameter growth.

INTRODUCTION

Across the South, many forest land managers may opt to use mechanical site preparation, especially when soil treatments are deemed appropriate. While these mechanical treatments may be highly effective at addressing a particular soil problem or debris issue, they are typically less effective at control of competing vegetation. In these scenarios where pines are planted, a woody release treatment using herbicides is usually applied at the end of either the first or second growing season. The purpose of these applications is to provide long-term control of the woody competitors with typically short-term control of any herbaceous species on the site.

For years, OUST® XP has been added to site preparation tank mixtures to provide residual control of herbaceous weeds the growing season following application. This addition has proven to be very effective in controlling herbaceous competition and promoting growth of the planted pine seedlings. The objectives of this study were (1) to determine if Oust® XP could provide herbaceous weed control during the growing season following a fall release application and (2) to evaluate the pine growth response to the various treatments.

STUDY SITES

Three study sites were utilized in the project. The sites were located in Texas, Mississippi, and South Carolina. In Texas, the study was located on Temple-Inland forest land near Alcoa, TX. The soil was a sandy loam with a pH = 5.2. The previous stand had been a mixed pine-hardwood composition which was harvested in 1999. The site received mechanical site preparation in 1999 and was planted with loblolly pine (*Pinus taeda* L.) seedlings in January 2000.

In Mississippi, the study was located on Weyerhaeuser Company land near Bradley, MS. The soil was a silt loam with a pH = 5.2. The previous stand had been a mixed pine-hardwood composition which was harvested in 1998. The site received mechanical site preparation in 1999 and was planted with loblolly pine in January 2000.

In South Carolina, the study was installed on Clemson school forest land near Central, SC. The soil was a clay loam with a pH = 5.3. The previous stand had been mixed pine-hardwood composition which was harvested in 1998. The site received chemical treatment and burning in 1999 and was planted in January 2000.

TREATMENTS

A complete list of treatments is found in table 1. Six of the treatments were applied on all three sites. Two of the treatments (#7 and #8) were applied in Mississippi only. Generally, the treatments were comparing two rates of Arsenal AC with or without the addition of Escort® XP or Oust XP®. The Eagre® in treatments 7 and 8 was a glyphosate product with 4 pounds active ingredient per gallon. A crop oil concentrate or nonionic surfactant was added to each treatment as noted in the table.

All treatments were applied during the period September 1–8, 2001 (date varied by site). The treatments were applied using a CO_2-powered backpack sprayer with a pole extension and KLC-9 nozzle. Total spray volume was 10 g/acre. This equipment simulates an aerial application.

EXPERIMENTAL DESIGN

The treatments were applied to rectangular plots 30 feet wide and 100 feet long. All treatments were replicated three times on all sites in a randomized complete block design.

EVALUATIONS

Prior to treatment application, all hardwoods in the sample area of each plot (10- by 80-foot area centered in each plot) were recorded by species and height class. At this same pretreatment timing, the heights and groundline diameters (GLD) of all planted pines in the sample area were recorded.

Herbaceous weed control was evaluated in April, May, June, July, August, and September of 2002. This was accomplished using ocular estimates of the percent ground cover of the major vegetation categories (grass/sedge, broadleaf forbs, and vines).

Hardwoods were recorded by species and height again in November 2002. Pine heights and diameter at breast height

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Table 1—List of treatments in 2001 DuPont fall Oust® pine release study

Treatment no.	Herbicide and rate per acre
1	Untreated
2	Arsenal AC (16 oz) + COC (3.2 oz)
3	Arsenal AC (12 oz) + COC (3.2 oz)
4	Arsenal AC (12 oz) + Escort® XP (1 oz) + COC (3.2 oz)
5	Arsenal AC (12 oz) + Escort® XP (1 oz) + Oust® XP (2 oz) + COC (3.2 oz)
6	Arsenal AC (12 oz) + Eagre (12.8 oz) + Oust® XP (2 oz) + Escort® XP (1 oz) + Entry II (10 oz)
7[a]	Eagre (25.6 oz) + Escort XP® (2 oz) + Oust® XP (2 oz) + Entry II (10 oz)
8[a]	Eagre (25.6 oz) + Escort XP® (4 oz) + Oust® XP (2 oz) + Entry II (10 oz)

oz = ounces; COC = Timbersurf 90.

[a] Mississippi only.

(d.b.h.) were remeasured in December 2002, December 2003, December 2004, and November 2006.

DATA ANALYSIS

Hardwood control was based on a percent reduction of stems and cumulative heights by species. Pine height and GLD data were subjected to analysis of variance and means separation using Duncan's new multiple range test (DNMRT) to test for significant differences among treatments. Herbaceous coverage data were averaged by treatment, subjected to arcsine transformation, with means separated using DNMRT.

RESULTS

Grass Control

With the exception of *Andropogon* in Mississippi, the addition of Oust® XP provided excellent grass control. Grass coverage was significantly less in all treatments that had Oust® XP in the mixture.

Forb Control

With the exception of wooly croton (*Croton capitatus*) in Texas, the addition of Oust® XP provided excellent broadleaf control. Again broadleaf coverage was significantly less in areas treated with mixtures containing Oust® XP.

Overall Control

All treatments resulted in >90 percent control of hardwoods. The addition of Oust® XP provided excellent results, especially in Mississippi and South Carolina. Wooly croton grew aggressively in treated areas in Texas. The results were comparable to the excellent response obtained from adding Oust® XP to site preparation applications. If *Rubus* is a problem on the site, Oust® Extra may be a better product for adding to the application.

Pine Height Growth

Average pine heights for the Texas and Mississippi sites at five growing seasons after treatment (5 GSAT) are found in table 2. Data for the South Carolina site are not available. Generally, release treatments resulted in greater total heights than untreated areas (table 2). The high rate (16 ounces per acre) of Arsenal AC alone or all the two-way and three-way mixes resulted in heights which were significantly greater than the average height in the untreated plots.

Diameter growth followed a similar pattern to height growth at 5 GSAT. All treatments resulted in trees with significantly larger diameters as compared to those in untreated areas (table 3). In Mississippi, the treatments with the high rate of Arsenal AC and those with Oust® XP added resulted in significantly greater average diameters as compared to the low rate of Arsenal AC alone or untreated. In Texas, there was no statistical separation of the treatments except treated vs. untreated. The reason for the lack of treatment separation is thought to be due to the effect of wooly croton competition.

Overall, the inclusion of Oust® XP in the release treatments provided excellent residual herbaceous control and pine growth response. Isolated species which are not controlled by Oust® XP can eliminate treatment differences, but sulfometuron methyl (Oust® XP) had a very broad spectrum of control, and loblolly pines should respond very favorably to its addition to any tank containing release treatments.

Table 2—Average total height at 5GSAT in fall Oust® release study, Mississippi and Texas (average all reps)

Treatment[a]	Height, 5 GSAT	
	Mississippi	Texas
	---------- *feet* ----------	
Untreated	24.05 c	28.95 b
A (12 oz)	23.68 c	29.38 ab
A (16 oz)	27.19 a	29.76 a
A + E (12 oz + 1 oz)	25.10 b	29.64 a
A + E + O (12 oz + 1 oz + 2 oz)	25.58 ab	29.98 a
A + G + E + O (12 oz + 12 oz + 1 oz + 2 oz)	25.66 ab	30.97 a
G + E + O (25 oz + 2 oz + 2 oz)	25.26 ab	—
G + E + O (25 oz + 4 oz + 2 oz)	28.25 a	—

GSAT = growing seasons after treatment; oz = ounces.

[a] Herbicide—A = Arsenal AC; E = Escort® XP; O = Oust® XP; G = glyphosate (Eagre).

Values in a column followed by the same letter do not differ at $\alpha = 0.05$.

Table 3—Average d.b.h. at 5GSAT by treatment and time of observation in fall Oust® release study, Mississippi and Texas (avg. all reps)

Treatment[a]	D.b.h., 5 GSAT	
	Mississippi	Texas
	---------- *inches* ----------	
Untreated	4.06 c	4.76 b
A (12 oz)	4.92 b	5.61 a
A (16 oz)	5.67 a	5.88 a
A + E (12 oz + 1 oz)	4.85 b	5.72 a
A + E + O (12 oz + 1 oz + 2 oz)	5.04 ab	5.95 a
A + G + E + O (12 oz + 12 oz + 1 oz + 2 oz)	5.31 a	5.69 a
G + E + O (25 oz + 2 oz + 2 oz)	5.37 a	—
G + E + O (25 oz + 4 oz + 2 oz)	5.42 a	—

GSAT = growing seasons after treatment; oz = ounces.

[a] Herbicide—A = Arsenal AC; E = Escort® XP; O = Oust® XP; G = glyphosate (Eagre).

Values in a column followed by the same letter do not differ at $\alpha = 0.05$.

PINE GROWTH FOLLOWING CHEMICAL SITE PREP AND POSTPLANT HERBACEOUS WEED CONTROL COMPARED TO CHEMICAL SITE PREP ONLY

Dwight K. Lauer and Harold E. Quicke[1]

Abstract—Three site prep vegetation control systems were compared on two Piedmont and two Upper Coastal Plain sites. Systems were (1) a one-time site prep application of Chopper® GEN2™ [2], (2) a one-time application of Chopper® GEN2™ tank mixed with sulfometuron, and (3) two applications consisting of site prep with Chopper® GEN2™ followed by herbaceous weed control with Arsenal AC plus sulfometuron in March/April following planting. Each of these systems was repeated with a July/August, September, and October site prep timing. The third system, consisting of two applications, resulted in better pine response and vegetation control for site prep in July through September on Upper Coastal Plain sites. The first system, a one-time application of Chopper® GEN2™ , provided good weed control and pine growth on Piedmont sites. The sulfometuron tank mix did not improve vegetation control and had negative effects on pine growth on Piedmont sites.

INTRODUCTION

In recent years, herbicide site prep tank mixes have been used to provide both long-term control of woody vegetation and residual control of herbaceous weeds in the first pine year. While this option eliminates the cost of a second herbicide application, there is no published information on the effects of these different vegetation control systems on the growth of planted pine.

This research project examines vegetation control and loblolly pine (*Pinus taeda*) response to different vegetation management systems on Upper Coastal Plain and Piedmont sites. Chopper® GEN2™ herbicide and the herbicide sulfometuron were used in site prep treatments. Chopper® GEN2™ herbicide is both foliar and soil active. With foliar broadcast applications, Chopper® GEN2™ controls a broad spectrum of woody and herbaceous vegetation and provides some residual control of herbaceous weeds into the year following treatment. However, Chopper® GEN2™ alone often does not provide adequate residual weed control. The herbicide sulfometuron can be tank mixed with Chopper® GEN2™ to enhance residual herbaceous weed control, thereby potentially eliminating the need for an additional postplant herbaceous weed control application. Residual control of weeds into the first growing season may be dependent on site prep timing since residual herbicide in the soil is expected to decrease as time on the ground increases.

METHODS

Three vegetation management systems were compared—(1) a one-time site prep application of Chopper® GEN2™, (2) a one-time application of Chopper® GEN2™ tank mixed with sulfometuron, and (3) two applications consisting of site prep with Chopper® GEN2™ followed by herbaceous weed control with Arsenal AC plus sulfometuron in March/April following planting. Chopper® GEN2™ was used at 40 ounces per acre. The sulfometuron site prep rate was 3 ounces product per acre using a 75-percent active ingredient formulation. Postplant herbaceous weed control was 4 ounces per acre Arsenal AC tank mixed with 2 ounces product per acre sulfometuron. Site prep applications included 1 percent methylated seed oil (MSO) except for Carson, MS, where 12.5 percent MSO was used to improve control of yaupon (*Ilex vomitoria*). Burning was combined with site prep only at Appomattox, VA, where the site was burned in June 2005 about a month before the first site prep application.

Studies were installed at four locations, two on Upper Coastal Plain sites and two on Piedmont sites. Locations are characterized in terms of geographic location, soils, and planting in table 1. To quantify the effects of site prep timing, each vegetation management regime was repeated using site prep applications in July/August, September, and October (table 2). The September application in Mississippi was missed due to Hurricane Katrina.

At each location, a randomized complete block design experiment was installed with three replications. Treatment plots were 60 feet in length and seven tree rows in width. Measurement plots were three rows in width and 30 feet in length centered within each treatment plot. This resulted in approximately 15 measurement trees for each plot. Pine groundline diameter and total height were measured at the end of the first, second, and third years. Stem volume index was calculated as the volume of a cone using groundline diameter and total height. Vegetation cover was assessed in June of the first year using ocular estimates of percent cover.

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[2] Chopper® GEN2™ is a registered trademark of BASF Corporation.

Table 1—Study locations

Region	Location	Soils	Slope	Drainage	Planting
			percent		
Upper Coastal Plain	Greenville, AL	Orangeburg sandy loam	1 to 5	Well drained	Machine
Upper Coastal Plain	Carson, MS	Ora sandy loam (two reps) Smithdale sandy loam (one rep)	2 to 5 12 to 17	Moderate well to well drained	Machine
Piedmont	Appomattox, VA	Tatum silt loam	2 to 7	Well drained	Hand
Piedmont	Saluda, SC	Georgeville silt loam (two reps) Herndon silt loam (one rep)	2 to 6	Well drained	Machine

Table 2—Herbicide application details

	Site prep dates			
Location	July/August	September	October	HWC date
Greenville, AL	7/31/2005	9/13/2005	10/16/2005	3/31/2006
Carson, MS	7/28/2005	—	10/18/2005	3/26/2006
Appomattox, VA[a]	7/23/2005	9/03/2005	10/01/2005	4/19/2006
Saluda, SC	8/03/2005	9/09/2005	10/23/2005	3/19/2006

HWC = selective postplant herbaceous weed control application; — = application not made due to Hurricane Katrina.

[a] Appomattox was the only location that included burning with site prep (June 2005 prior to herbicide application).

RESULTS AND DISCUSSION

Vegetation Control

Chopper® GEN2™ site prep provided good control of hardwood species on all sites. However, control of herbaceous vegetation differed greatly between Upper Coastal Plain and Piedmont sites.

Upper Coastal Plain Sites—Single vegetation control treatments of Chopper® GEN2™ herbicide resulted in heavy herbaceous weed competition the year following treatment. Applying this treatment late in the year did not help to reduce the level of herbaceous competition in the year following treatment. The tank mix of Chopper® GEN2™ with sulfometuron applied in September and October greatly reduced vegetation cover with cover never exceeding 28 percent compared to Chopper® GEN2™ alone that was never <48 percent. However, the late July application of the tank mix was too early for good herbaceous weed control and resulted in >40 percent cover. The two applications management system of Chopper® GEN2™ followed with postplant herbaceous weed control provided robust weed control no matter when the site prep treatment was applied. Total cover was <28 percent for all site prep timings. For October site prep applications, one application of Chopper® GEN2™ plus sulfometuron provided better or equivalent vegetation control to the two-application system. For site prep in July through September, two applications resulted in substantially better weed control than one application of Chopper® GEN2™ plus sulfometuron.

Piedmont Sites—The single vegetation control treatment of site prep with Chopper® GEN2™ herbicide resulted in good herbaceous weed control the year following treatment with vegetation cover never exceeding 25 percent and cover decreased with later season site prep. The addition of sulfometuron to Chopper® GEN2™ site prep had little impact on weed cover since cover was already low for Chopper® GEN2™ without sulfometuron. Similarly, adding postplant herbaceous weed control had little impact on weed cover.

Pine Response

Upper Coastal Plain Sites—Vegetation control and pine response generally increased with treatment intensity. Chopper® GEN2™ alone was not adequate on these sites with aggressive vegetation. Chopper® GEN2™ combined with postplant herbaceous weed control provided better response than the single application of the Chopper® GEN2™ with sulfometuron tank mix but timing was also important.

Chopper® GEN2™ applications made in September or earlier followed with postplant herbaceous weed control resulted in substantially better pine growth than one application of Chopper® GEN2™ tank mixed with sulfometuron. Growth increases in year 3 pine volume index were over 50 percent for July site prep and 14 percent for September site prep. For October site prep, results were variable with two applications resulting in similar pine growth at Carson, but less pine growth at Greenville compared to one application of Chopper® GEN2™ tank mixed with sulfometuron.

Piedmont Sites—The most consistent treatment was one application of Chopper® GEN2™ alone. This treatment resulted in very good weed control in the year following treatment with little room for improvement from more intensive treatments. Adding sulfometuron to Chopper®GEN2™ often had a negative effect on pine growth.

Site prep timing was a factor at Saluda. Pine response was better for the earlier applications of Chopper® GEN2™ alone even though vegetation cover decreased from 25 to 8 percent as site prep was delayed from August to October. This trend was not evident at Appomattox where there was little difference in pine growth among site prep dates. The hot burn at Appomattox prior to the first site prep treatment may have negated some of the benefits of early season site prep.

OPERATIONAL RECOMMENDATIONS

While single application vegetation control consisting of site prep with Chopper® GEN2™ tank mixed with sulfometuron has been widely adopted, there is no published information on pine growth responses compared to other treatments. Results may vary by year because of changing environmental conditions and will depend on site-specific weed species. However, these studies provide a framework for planning and evaluation of operational treatments for pine response.

Upper Coastal Plain Recommendations

Use two applications on sites with sandy-loam soils consisting of Chopper® GEN2™ site prep applied early (July through September) followed by postplant herbaceous weed control in the spring. In the event that site prep must be delayed until October, use a single application of Chopper® GEN2™ tank mixed with sulfometuron with the understanding that this timing does not result in the best pine growth.

Piedmont Recommendations

Use one application consisting of Chopper® GEN2™ site prep on sites with silt-loam or finer textured soils. Do not add sulfometuron and do not follow with postplant herbaceous weed control. Apply the site prep treatment early in the growing season for best pine response.

ACKNOWLEDGMENTS

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CONTROL OF UNWANTED HARDWOODS WITH JUNE-APPLIED CHOPPER® GEN2™ AND CHOPPER® ON FIVE SITES

Jimmie Yeiser, Andrew Ezell, and Michael Blazier[1]

Abstract—Chopper® GEN2™ is an improved formulation of Chopper®, offering enhanced early product performance. Managers of southern timberlands routinely apply Chopper® from July through October. A wider application window allows managers more time to complete the applications needed for effective weed control. GEN2™ offers to expand the application window into June. One site each in Louisiana, Mississippi, Oklahoma, South Carolina, and Virginia were selected for testing. The objective was to compare the control of unwanted hardwoods with Chopper (32 ounces product per acre) and Chopper® GEN2™ (32 ounces product per acre) mixed with zero, 2.5 percent, 7.5 percent, and 12.5 percent methylated seed oil (MSO). All sites were treated between June 2–14, 2005. Red maple (*Acer rubrum* L.), blackgum (*Nyssa sylvatica* Marsh.), sweetgum (*Liquidambar styraciflua* L.), red oak (*Quercus falcata* Michx.), white oak (*Q. alba* L.), and yellow-poplar (*Liriodendron tulipifera* L.) were the major hardwoods tested. Two growing seasons after treatment, there were two findings of operational significance. First, percent control of Chopper® GEN2™+2.5 percent MSO was better than Chopper®+2.5 percent MSO and equivalent to Chopper®+12.5 percent MSO. This finding allows early (pre-June 15) application at reduced MSO volumes and cost without reduced control. Second, the overall control for the best treatment in this study was 87 percent, indicating higher rates could provide even higher control.

INTRODUCTION

Chopper® is commonly used for herbicidal preparation of pine sites for planting. Chopper® GEN2™ is an improved formulation of Chopper®, offering enhanced early product performance. Both products contain the active ingredient, imazapyr.

Managers of southern timberlands routinely apply Chopper® from July through October. Unfavorable weather between July and October increases the difficulty in completing timely applications. A wider application window allows managers more time to complete the applications needed for effective weed control. Opportunity exists to expand the application window into June.

METHODS

The objective of this study was to compare the control of unwanted hardwoods with a single foliar application of Chopper® (32 ounces product per acre) or Chopper® GEN2™ (32 ounces product per acre) mixed with zero, 2.5 percent, 7.5 percent, and 12.5 percent Conquer MSO when preparing pine sites for planting.

Five sites, one each in Louisiana, Mississippi, Oklahoma, South Carolina, and Virginia were selected for testing. Site physiography and major hardwood species groups are summarized in table 1.

Trees were visually evaluated at 2, 4, and 8 weeks after treatment (WAT) for brownout in 10 percent intervals with zero = no browned foliage and 100 = total browned foliage. Percent control was computed after one, two, and three growing seasons after treatment (GSAT). Percent control was defined as (sum of initial height-sum of height on evaluation day) per sum of initial height. Initial height was recorded pretreatment and evaluation height in September 2006 and 2007 and in Virginia only again in 2008. In Oklahoma, both timings were oversprayed with an operational treatment prior to the fall 2006 assessment.

RESULTS AND DISCUSSION

Brownout

Numerical patterns in brownout were detected (table 2). The numerical data pattern is very consistent. Commonly, brownout was greater for GEN2™+2.5 percent MSO than Chopper®+2.5 percent MSO for 9 of 12 State-species combinations at 2 and 4 WAT and 10 of 12 State-species combinations at 8 WAT (table 2). At 2, 4, and 8 WAT, sweetgum in South Carolina and red maple in Virginia browned more from 2.5 percent MSO with Chopper® than GEN2™ (table 2).

Control

Numerical patterns for percent control for major species groups at each location were observed and are presented in table 3. The industry standard Chopper®, mixed with 2.5 percent MSO, seldom exceeded GEN2™+2.5 percent MSO control. For example, Chopper®+2.5 percent MSO control exceeded GEN2™+2.5 percent MSO by 10 percent or more only 3 of 15 or 2 of 12 site-species combinations 1 or 2 GSAT, respectively. This speaks to the high consistency with which June-applied GEN2™ controlled woody weeds on test sites. GEN2™+2.5 percent MSO control remained the same (100 percent) or increased from one to two GSAT for all site-species combinations (table 3). This was only slightly better than Chopper®+2.5 percent MSO that decreased in control

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Table 1—Physiography, major species groups, and herbicide application date for each site

Site	Appomattox, VA	Homer, LA	Mathiston, MS	Broken Bow, OK	Pendleton, SC
Physiography	Piedmont	Hilly Coastal plain	Hilly Coastal plain	Hilly Coastal plain	Piedmont
Species groups	Blackgum		Blackgum		
				Elm	
	Red maple	Red maple	Red maple	Red maple	
	White oak		Red oak		
		Sweetgum	Sweetgum		Sweetgum
					Yellow-poplar
	All	All	All	All	Average
Sprayed	June 11, 2005	June 14, 2005	June 3, 2005	June 2, 2005	June 7, 2005

from one to two GSAT only once (all species in Louisiana). This consistent performance suggests GEN2™ may provide the same reliable control managers have long observed with Chopper®.

Control by GEN2™+2.5 percent MSO was similar to the operational standard, Chopper®+12.5 percent MSO, for most species and sites (table 4). This was not always true. One GSAT, Chopper® out performed GEN2™ at controlling all species in Louisiana, sweetgum in South Carolina and red maple in Virginia. But, for all three examples control two GSAT was similar or better for GEN2™+2.5 percent MSO than Chopper®+12.5 percent MSO.

Control three GSAT is not presented. Limited data exists. From this data, white oak in Virginia was the only example where Chopper®+12.5 percent MSO (100 percent) provided more long-term control than GEN2™+2.5 percent MSO (89 percent). Rather, GEN2™+2.5 percent MSO provided more control three GSAT than Chopper®+12.5 percent MSO for red maple, blackgum, and all species.

When averaged for common species, table 4 illustrates the strong data patterns favoring June-applied GEN2™+2.5 percent MSO rather than Chopper®+12.5 percent MSO for control of several species commonly occupying pine sites. This pattern is significant because this shows managers can reduce the MSO cost without decreasing control. Furthermore, GEN2™+2.5 percent MSO provided 87 percent mean control across sites and species leaving managers the opportunity to use higher rates for more control if desired.

SUMMARY

After two growing seasons, there were two findings of operational significance. First, percent control of unwanted hardwoods with Chopper® GEN2™+2.5 percent MSO was better than Chopper®+2.5 percent MSO and equivalent to Chopper®+12.5 percent MSO, the industry standard. When compared with the industry standard, this finding supports early (June 2–14) application of GEN2™+2.5 percent MSO for reduced cost without reduced control. Second, the overall control for the best treatment in this study was 87 percent, indicating higher rates could provide even higher control.

Table 2—Brownout (percent) at 2, 4, and 8 weeks after treatment from 32 ounces of Chopper® or Chopper® GEN2™ with different amounts (percent) of methylated seed oil

State, Species, and treatment	Weeks after treatment[a] and methylated seed oil (percent)											
	2				4				8			
	0	2.5	7.5	12.5	0	2.5	7.5	12.5	0	2.5	7.5	12.5
Louisiana												
Red maple												
GEN2™	4	3	29	1	5	4	38	3	57	37	70	84
Chopper®	17	3	6	25	18	5	7	48	19	7	20	60
Sweetgum												
GEN2™	23	28	24	22	30	52	37	33	38	70	63	50
Chopper®	17	27	18	20	37	28	30	22	44	28	38	27
Mississippi												
Blackgum												
GEN2™	82	91	6	82	97	97	54	93	100	100	86	100
Chopper®	5	64	85	92	17	96	97	99	26	100	99	100
Red maple												
GEN2™	10	17	6	20	34	84	35	73	93	100	85	100
Chopper®	8	17	18	11	32	68	86	48	59	97	100	100
Red oak												
GEN2™	3	5	3	4	5	7	5	6	24	76	49	51
Chopper®	1	1	3	3	3	3	8	6	22	43	59	79
Sweetgum												
GEN2™	22	—	14	30	72	—	86	97	99	—	99	100
Chopper®	5	27	54	27	14	22	88	89	23	40	100	99
Oklahoma												
Elm												
GEN2™	1	16	3	7	2	20	12	22	2	36	15	25
Chopper®	1	6	2	11	2	9	9	15	2	13	16	22
Red maple												
GEN2™	1	20	2	5	10	27	10	15	12	59	36	63
Chopper®	1	2	5	8	5	16	17	17	6	28	37	29
South Carolina												
Sweetgum												
GEN2™	40	37	34	21	40	37	34	21	39	49	49	56
Chopper®	28	55	34	38	28	55	34	38	44	54	51	67
Yellow-poplar												
GEN2™	8	7	10	9	11	12	11	12	13	38	32	24
Chopper®	8	10	11	11	11	11	19	13	43	22	32	30
Virginia												
Blackgum												
GEN2™	53	78	78	62	64	85	90	72	69	90	96	84
Chopper®	16	57	81	82	20	68	88	86	24	73	97	89
Red maple												
GEN2™	30	40	33	29	47	58	50	54	62	76	76	84
Chopper®	11	55	27	42	23	72	51	60	33	90	68	83
White oak												
GEN2™	4	10	8	11	10	18	15	17	18	56	55	51
Chopper®	4	6	10	10	5	12	24	17	8	42	57	42

[a] For all evaluations: Louisiana and South Carolina check brownout = zero, Mississippi check brownout was ≤1 percent, Oklahoma check brownout was <14 percent for elm (*Ulmus* spp.) and <4 percent for red maple, and Virginia check brownout was <7 percent.

Table 3—Control (percent) one or two growing seasons after treatment from a foliar application of Chopper® or Chopper® GEN2™ (32 ounces) with different amounts (percent) of methylated seed oil

State and treatment	Growing seasons after treatment and methylated seed oil (percent)															
	1				2				1				2			
	0	2.5	7.5	12.5	0	2.5	7.5	12.5	0	2.5	7.5	12.5	0	2.5	7.5	12.5
Louisiana				Red maple								All				
GEN2™	24	66	40	32	79	98	85	72	27	51	42	27	65	77	74	65
Chopper®	12	24	42	31	45	54	95	93	65	77	74	65	25	55	73	78
Check	–14				–47				–10				–38			
				Sweetgum												
GEN2™	29	36	44	22	51	56	64	58								
Chopper®	6	26	38	37	5	55	51	63								
Check	–7				–30											
Mississippi				Blackgum								Sweetgum				
GEN2™	97	100	71	100	100	100	100	100	97	—	80	87	100	—	100	100
Chopper®	48	100	91	100	78	100	100	100	33	45	93	90	80	64	100	100
Check	–27				–49				–64				–182			
				Red maple								All				
GEN2™	100	100	93	100	100	100	93	100	86	94	75	92	90	100	98	100
Chopper®	54	100	100	100	92	100	100	100	49	74	90	92	80	84	99	100
Check	–36				–27				–35				–78			
				Red oak												
GEN2™	49	83	54	80	58	100	100	100								
Chopper®	61	50	75	79	70	73	96	100								
Check	–12				–53											
Oklahoma				Elm								All				
GEN2™	–9	34	21	38	—	—	—	—	6	59	47	49	—	—	—	—
Chopper®	4	37	56	15	—	—	—	—	22	58	46	36	—	—	—	—
Check	–17								–15							
				Red maple												
GEN2™	30	62	53	82	—	—	—	—								
Chopper®	24	58	59	68	—	—	—	—								
Check	–25				—	—	—	—								
South Carolina				Sweetgum								Yellow-poplar				
GEN2™	37	39	59	52	76	92	96	96	6	16	21	22	100	100	100	100
Chopper®	39	49	50	64	81	83	89	95	19	8	31	21	100	84	97	100
Check	0				–57				–8				–63			
Virginia				Blackgum								White oak				
GEN2™	–43	95	93	79	–66	96	100	62	19	59	56	47	35	71	93	81
Chopper®	33	59	87	86	–77	65	100	100	–5	42	45	32	–9	81	96	93
Check	–163				–421				–40				–134			
				Red maple												
GEN2™	34	59	54	56	72	86	65	43								
Chopper®	–12	84	50	78	–19	100	79	78								
Check	–101				–238											

Table 4—A summary of control (percent) two growing seasons after a foliar treatment of Chopper® or Chopper® GEN2™ herbicide (32 ounces) was applied with different amounts of methylated seed oil

		Virginia			Louisiana		Mississippi			South Carolina		
Herbicide	MSO	BLG	REM	WHO	REM	SGM	BLG	REM	REO	YEP	SGM	Mean
	percent											
GEN2™	0.0	–52	86	60	79	—	100	100	58	100	76	61
	2.5	100	86	89	98	—	100	100	100	100	92	87
	7.5	100	74	93	85	—	100	93	100	100	96	85
	12.5	60	41	97	72	—	100	100	100	100	96	78
Chopper®	0.0	–75	5	–5	45	—	78	92	70	100	81	39
	2.5	65	100	88	54	—	100	100	73	84	83	75
	7.5	100	81	100	95	—	100	100	96	97	89	87
	12.5	100	67	100	93	—	100	100	100	100	95	87

MSO = methylated seed oil; BLG = blackgum; REM = red maple; WHO = white oak; SGM = sweetgum; REO = red oak; YEP = yellow-poplar.

ESTABLISHMENT TRIAL OF AN OAK-PINE/SOYBEAN-CORN-WHEAT ALLEY-CROPPING SYSTEM IN THE UPPER COASTAL PLAIN OF NORTH CAROLINA

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Abstract—Alley cropping may prove useful in the Southeast United States, providing multiple products and income streams, as well as affording sustainable land use alternatives to conventional farming. An alley-cropping system may be a good alternative in agriculture because of the benefits provided by trees to crops and soils, as well as the income generated from wood products and timber. In the current study triple row single-species strips of cherrybark oak (*Quercus pagoda* Raf.), loblolly pine (*Pinus taeda* L.), or longleaf pine (*P. palustris* Mill.) were planted separated by 12- or 24-m wide areas of agricultural crops. Survival, seedling height, and diameter growth were measured in the first and second year of the study. Cherrybark oak, loblolly pine, and longleaf pine seedlings all had over 80 percent survival rates and showed positive height and diameter growth over the first two growing seasons.

INTRODUCTION

Agroforestry has been practiced for centuries around the World, and new agroforestry systems and strategies are currently under research, with an emphasis on improving practices and preserving the quality of the environment (Jama and others 2006, Nair 2007, Sanchez 2005, Williams and others 1997). In recent years, agroforestry research has focused on developing techniques and improving efficiency in various systems (Erdmann 2005, Sanchez 1995, Young 2004). One type of agroforestry that has been gaining interest in the Southeastern United States is alley cropping (Stamps and Linit 1997). Alley cropping is a system of trees and crops that are managed on the same parcel of land at the same time, and generally consists of tree rows running in narrow strips along the length of an agricultural field planted in annual crops. The current study was designed as an alley-cropping system that incorporated cherrybark oak (*Quercus pagoda* Raf.), loblolly pine (*Pinus taeda* L.), and longleaf pine (*P. palustris* Mill.) into an agricultural crop rotation.

In alley-cropping systems, trees can provide a diverse and extensive range of ecological benefits (Huxley 1983). Land managed for both trees and agriculture can host and support a wider range of arthropods and members of the soil community, which has been shown to increase aboveground productivity (Crutsinger and others 2006). Trees affect soil nutrition by helping to stabilize soil structure, which can also determine the ability of root growth and nutrient accessibility of annual crops (Leakey 1999). Trees can access and recycle nutrients that would otherwise be unattainable for annual crops (Sanchez 1987, Young 1989) and pump water out of soils (Erdmann 2005, Ledgard 2001, Wood and Burley 1991) which can reduce salinity and enhance soil fertility (Jose and others 2008). Trees can provide extra ground cover for better protection of waterways (Bernstein 1975, Prinsley 1992), create an improved microclimate for crops (Long and Nair 1999), minimize weed competition (Basavaraju and Gururaja Rao 2000), and contribute to economic diversity (Jose and others 2008), including the potential for landowners growing trees to earn carbon credits (Rizvi and others 1999).

METHODS

The study consisted of cherrybark oak, loblolly pine, and longleaf pine in an alley-cropping management scheme with annual crops that included soybean [*Glycine max* (L.) Merr.], corn (*Zea mays* L.), and wheat (*Triticum aestivum* L.) in a 3-year rotation. It was deployed in Goldsboro, NC, in January 2007. The field site is located at the North Carolina Department of Agriculture & Consumer Services, Cherry Research Farm on the Coastal Plain in the eastern-central part of North Carolina. The Cherry Research Farm hosts the Center for Environmental Farming Systems (CEFS), established in 1994, and is one of the largest centers for the study of environmentally sustainable farming practices in the Nation (www.cefs.ncsu.edu).

The site is a 10-ha (25-acre) agriculture field that had been in corn and soybean production for several years prior to the current study. The southern edge of the field borders the Neuse River, and the eastern edge borders a tree-lined ditch for drainage. According to the U.S. Department of Agriculture soil survey (http://soils.usda.gov/survey), the field site includes four soil types: Lakeville sand (49.7 percent of total field ha), Coxville loam (37.7 percent of total field ha), Chewacla loam (9.3 percent of total field ha), and Leaf loam (3.3 percent of total field ha).

The three tree species planted were selected for their regional relevance and market value. The agriculture production was set up in a 3-year annual crop rotation with long-term plans

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for establishing perennial forage, with a possibility for grazing, as the trees mature. The field was blocked into five replicates from north to south to account for the gradient of soil variation and slope. Each block was 165 m (540 feet) wide and 128 m (140 feet) long with 1.5-m (5-foot) buffers (fig. 1).

Each block was then divided into annual agricultural crop plots and tree plots. The annual crop sections were 12 m (40 feet) wide by 128 m long, or 24 m (80 feet) wide by 128 m long. Each block contained two of each of 12-m width, and 24-m width areas, and included five tree plots, 6 m (20 feet) wide by 128 m long. The tree plots (triple-row wide strips) were laid out in between the annual crop areas (fig. 1). They were divided into thirds lengthwise, each species randomly assigned to each section, measuring 6 m wide by 43 m long (20 by 140 feet). Trees were planted in plots, three rows wide, at 1.5- by 2-m (5- by 7-foot) spacing. The total number of seedlings planted, including all species, was 5,000. Equipment access roads were delineated between blocks, as 4.5 m (15 feet) of unplanted areas, running across the width of the field.

The entire field had been conservation tillage planted to corn in 2006 and harvested before the end of that year. In that year it was not cultivated before the start of the current study. Trees were planted by block in January 2007. All the seedlings came from the North Carolina Division of Forest Resources, F.H. Claridge Nursery in Goldsboro. The loblolly pine seedlings were 1-0 bare root from genetically improved seed. The longleaf pine was planted as 1-0 container seedlings grown from seed that originated from Bladen County, NC. The cherrybark oak was planted as 1-0 bare-root seedlings with seed that came from Pee Dee River Basin in upper South Carolina.

In March 2007, before the trees had broken bud, the preemergent herbicide, Oust® (Du Pont, Wilmington, DE; sulfometuron methyl) was sprayed overtop at 219 ml/ha (3 ounces per acre) using a 6.5-m (20-foot) spray boom so that the three rows of trees, each row 1.5 m apart, were covered with herbicide. Between the triple-row plots of trees, in the agricultural areas either 12 or 24 m wide, the field was disk harrowed in April, and again in May 2007, but not closer than 1.5 m from the outer tree row in each plot. Potash [potassium oxide (K2O)] was then broadcast applied to the entire field, including trees and agriculture areas, at 224 kg/ha (200 pounds per acre). The field areas between the trees were again disked twice in May, the area was then soil-surface conditioned with a Lorenz device (Lorenz Mfg. Co., Watertown, SD). The Asgrow 5905 (Asgrow Seed Company LLC, St. Louis, MO) variety of glyphosate-resistant soybeans were planted on May 21 on 76-cm (30-inch) row spacing at 49 398 seeds/ha (123,493 seeds per acre). The soybeans were sprayed in June with glyphosate at 210 g/ha (40 ounces per acre) with a hooded sprayer, and again in July with glyphosate at the same rate along with the herbicide FirstRate™ (Dow AgroSciences, Minneapolis, MN, cloransulam-methyl) at 21.9 ml/ha (0.3 ounce per acre), with

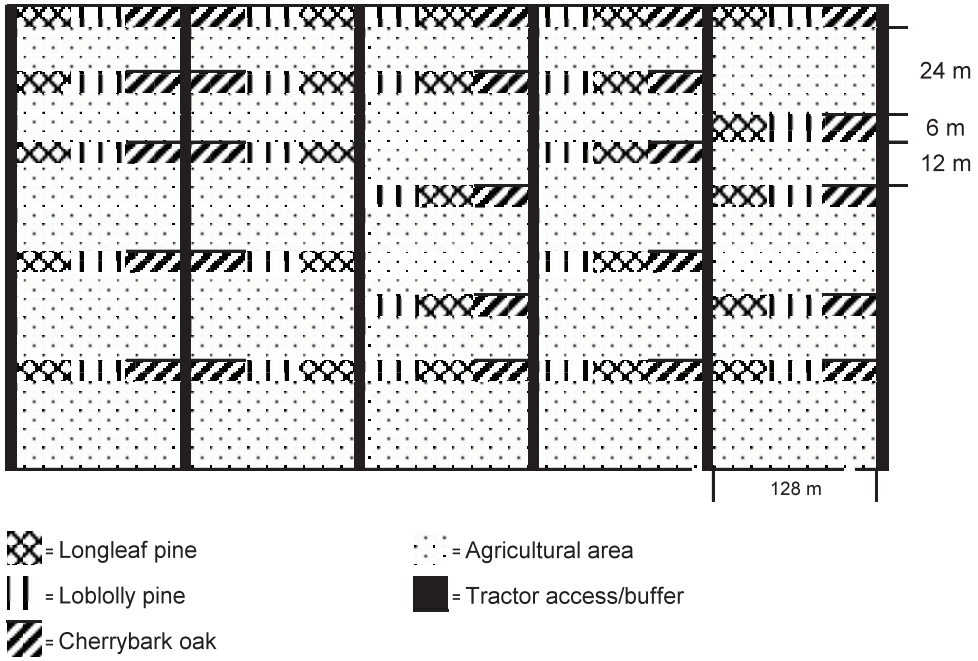

Figure 1—Layout and dimensions of agroforestry field trials, including tree species, of longleaf pine (*Pinus palustris*), loblolly pine (*P. taeda*), and cherrybark oak (*Quercus pagoda*) planted in early 2007 in triple-row strips between agricultural areas (two different widths) which were planted to soybean (*Glycine max*) in 2007 and corn (*Zea mays*) in 2008 near Goldsboro, NC.

a hooded sprayer. The soybean harvest yielded 670 kg/ha (10 bushels per acre).

Tree height (±1 cm) and diameter (±0.1 cm) measurements were taken from a subsample of 450 trees to represent the size of trees at the time of planting, and again after their first and second growing seasons, in November 2007 and December 2008. Cherrybark oak and loblolly pine height were determined by measuring from the soil surface to the tallest resting bud. For longleaf pine seedlings, height was measured from the soil surface to the top of the grass bunch. Diameters were measured with a dial caliper 2 cm above the root collar. Longleaf pine diameter was not recorded. Annual height and diameter growth was calculated. Seedling survival was recorded in November 2007 at the end of the growing season, and confirmed in spring 2008.

RESULTS AND DISCUSSION

The mean first year (2007) height growth of cherrybark oak seedlings was 10.7 cm, and diameter growth was 3.0 mm (table 1). In the second year, cherrybark oak seedling height growth increased by 218 percent, and the diameter growth increased 263 percent over the previous year's growth, with an average of 35 cm height growth and 10.9 mm diameter growth during the 2008 season. The total height growth over the two growing seasons was 45 cm, and diameter growth was 13.9 mm. Cherrybark oak survival was 93 percent. These results are within range of growth statistics reported in other studies of cherrybark oak seedlings in the Southeast (Dubois and others 2000, Stanturf 1995).

Loblolly pine seedlings averaged 7.5 cm height growth and 4.9 mm diameter growth for the 2007 growing season (table 1). In 2008, the loblolly pine seedlings' height growth increased by 380 percent, and diameter growth by 263 percent, over the previous year's growth. For the 2 years of the study, the total height growth was 85 cm, and diameter 28.6 mm. Loblolly pine seedling survival was 89 percent. The loblolly pine seedlings suffered Nantucket tip moth [*Rhyacionia frustana* (Comst.)] attack in the first 2 years, damaging the apical stems, and affecting height growth and possibly diameter growth, and introducing variability into the growth estimates.

Longleaf pine seedlings had an average needle height growth of 16 cm for the 2007 growing season, but in 2008 height growth slowed, averaging only 10 cm (table 1). The total 2 year-needle height growth was 27 cm. Longleaf pine seedlings survival was 86 percent. By the end of the 2008 season, some of the longleaf pine were emerging from the grass stage and beginning to show stem elongation.

This alley-cropping trial will be maintained and monitored for several decades. It will be used as an integral part of the demonstration and education goals of the research farm, and as a template for continued research. Studies will include tree and agronomic productivity, financial implications, soil system attributes, insect-disease-weed interactions between the crop and tree areas, compatibility of weed control systems between the closely associated crop and tree plantings, and agroforest-ecosystem changes as the trees mature and provide more above- and belowground competition to the crop/forage zones. In 2007 an additional study, by the current authors, was superimposed on the tree strip to evaluate the impacts of on-farm wastes (discarded hay, hog manure-corn stover mix, black plastic) used as mulches around the newly planted seedlings. Other studies addressing management techniques and efficiency will follow.

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Table 1—Two-year height and diameter growth of 1-0 seedlings planted in early 2007 in an agroforestry trial near Goldsboro, NC

	Height growth	Diameter growth	Height growth	Diameter growth	Total height growth	Total diameter growth
Species	2007	2007	2008	2008	2007–08	2007–08
	cm	*mm*	*cm*	*mm*	*cm*	*mm*
Cherrybark oak	10.7	3.00	34.5	10.90	45.2	13.89
Loblolly pine	7.5	4.94	77.4	23.72	84.9	28.66
Longleaf pine	16.1	—	9.7	—	26.7	—

LITERATURE CITED

Basavaraju, T.B.; Gururaja R.M.R. 2000. Tree-crop interactions in agroforestry systems: a brief review. Indian Forester. 126 (11): 1155.

Bernstein, L. 1975. Effects of salinity and sodicity on plant growth. Annual Reviews in Phytopathology, 13 (1): 295–312.

Crutsinger, G.M.; Collins, M.D.; Fordyce, J.A. [and others]. 2006. Plant genotypic diversity predicts community structure and governs an ecosystem process. Science. 313(5789): 966–968.

Dubois, M.R.; Chappelka, 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(2): 105–118.

Erdmann, T.K. 2005. Agroforestry as a tool for restoring forest landscapes. In: Mansourian, S.; Vallauri, D.; Dudley, N., comps., ed. Forest restoration in landscapes: beyond planting trees. New York: Springer: 274–284.

Huxley, P.A. 1983. Characteristics of trees. In: Plant research and agroforestry. 1st ed. Nairobi, Kenya: International Council for Research in Agroforestry: 3–12.

Jama, B.; Elias, E.; Mogotsi, K. 2006. Role of agroforestry in improving food security and natural resource management in the drylands: a regional overview. Journal of the Drylands. 1(2): 206–211.

Jose, S.; Allen, S.C.; Nair, P.K.R. 2008. Tree-crop interactions: lessons from temperate alley-cropping systems. In: Batish, D.R., ed. Ecological basis of agroforestry. Boca Raton, FL: CRC Press. Chapter 2.

Leakey, R. 1999. Agroforestry for biodiversity in farming systems. In: Collins, W.W.; Qualset, C.O., comps., eds. Biodiversity in agroecosystems. 1st ed. Boca Raton, FL: CRC Press: 127–145.

Ledgard, S.F. 2001. Nitrogen cycling in low input legume-based agriculture, with emphasis on legume/grass pastures. Plant and Soil. 228(1): 43–59.

Long, A.J.; Nair, P.K.R. 1999. Trees outside forests: agro-, community, and urban forestry. New Forests. 17(1): 145–174.

Nair, P. 2007. The coming of age of agroforestry. Journal of the Science of Food and Agriculture. 87(9): 1613–1619.

Prinsley, R.T. 1992. The role of trees in sustainable agriculture—an overview. In: Prinsley, R.T., comps., eds. The role of trees in sustainable agriculture. 1st ed. Dordrecht, The Netherlands: Kluwer Academic Publishers. 87 p.

Rizvi, S.; Tahir, M.; Rizvi, V. [and others]. 1999. Allelopathic interactions in agroforestry systems. Critical Reviews in Plant Sciences. 18(6): 773–796.

Sanchez, P.A. 1987. Soil productivity and sustainability in agroforestry systems. In: Steppler, H.A.; Nair, P.K.R., comps., eds. Agroforestry: a decade of development. Nairobi, Kenya: International Council for Research in Agroforestry: 205–226.

Sanchez, P.A. 1995. Science in agroforestry. Agroforestry Systems. 30(1): 5–55.

Sanchez, P.A. 2005. Benefits from agroforestry in Africa, with examples from Kenya and Zambia. In: Pretty J.N., comps., eds. London; Sterling, VA: The Earthscan Reader in Sustainable Agriculture: 368–405.

Stamps, W.T.; Linit, M.J. 1997. Plant diversity and arthropod communities: implications for temperate agroforestry. Agroforestry Systems. 39(1): 73.

Stanturf, J.A. 1995. Survival and growth of planted and direct-seeded cherrybark oak in the Santee River floodplain after 11 years. In: Edwards, M.B., eds. Proceedings for the eighth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-1. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 489–494.

Williams, P.A.; Gordon, A.M.; Garrett, H.E.; Buck, L. 1997. Agroforestry in North America and its role in farming system. In: Gordon, A.M.; Newman, S.M. comps., eds. Temperate agroforestry systems. 1st ed. Wallingford, UK: CAB International: 9–84.

Wood, P.J.; Burley, J. 1991. A tree for all reasons: the introduction and evaluation of multipurpose trees for agroforestry. Nairobi, Kenya: International Council for Research in Agroforestry. 158 p.

Young, A. 1989. Agroforestry for soil conservation. Wallingford, Oxon, UK: CAB International. 29 p.

Young, A. 2004. Effects of trees on soils. In: Elevitch, C.R., comps., eds. The overstory book: cultivating connections with trees. Holualoa, HI: Permanent Agriculture Resources: 179–562. Vol. 2.

ASSESSING POTENTIAL GENETIC GAINS FROM VARIETAL PLANTING STOCK IN LOBLOLLY PINE PLANTATIONS

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Abstract—Forest landowners have increasingly more options when it comes to loblolly pine (*Pinus taeda* L.) planting stock. The majority of plantations in recent decades have been established with seedlings produced from second-generation open-pollinated (second-Gen OP) seed. However, foresters have begun recognizing the increased gains obtainable from full-sib families produced using mass-controlled pollination (MCP) techniques. The next step in genetic improvement is varietal forestry. Forest biotechnology firms are producing loblolly pine varietal planting stock for deployment in the Southeastern United States. Continued testing of this material will determine individual genotypes best suited for specific sites or desired products. In 2007, a Loblolly Pine Genetic Level Study was installed in northern Mississippi to examine differences in growth among second-Gen OP seedlings, MCP seedlings, and loblolly pine varieties derived from somatic embryogenesis. Following two growing seasons, the average height of the MCP material was slightly but significantly taller than that of the varietal planting stock. The average height of the top five performing varieties was 18 percent taller than the varietal average, 14 percent taller than the average height of the second-Gen OP, and 6 percent taller than the MCP material.

INTRODUCTION

The southeastern region of the United States is the largest wood-producing region in the country, accounting for over 56 percent of the total U.S. harvest (Adams and others 2006). There are a variety of reasons why the region has become such an important wood producer. One is the prevalence of plantations. There are currently over 30 million acres of pine plantations in the Southeast, with some projections showing this to increase to as high as 40 to 50 million acres by 2040 (Wear and Greis 2002). Given the typically higher yields of plantations over naturally regenerated forests, this increase in plantation acreage alone accounts for much of the regional increase in production.

In addition to the large increases in southern pine plantation acreage, the region has realized dramatic increases in per-acre yields of southern forests resulting from increased management intensity over the past 30 to 40 years. Over the past three decades, the average mean annual increment of southern pine plantations has more than doubled, while rotation lengths have been cut nearly in half. Productivity of newly established plantations can operationally achieve over 300 cubic feet per acre per year, and in some cases to over 400 cubic feet per acre per year (Fox and others 2007). These increases in plantation yields have resulted from several factors, such as improved site preparation techniques, effective competition control, and fertilization. Another major factor in the increased productivity seen over the past 40 years has been the genetic improvement of loblolly pine (*Pinus taeda* L.).

Genetically improved seedlings from first-generation seed orchards started becoming available for plantation establishment by the end of the 1950s. By the mid-1980s virtually all southern pine plantations were established with seedlings produced from genetically improved seed; and by the early 1980s, harvests from plantations established using genetically improved planting stock were beginning to show the benefits of the tree improvement programs. Gains in volume from first-generation plantations in the 1980s were generally in the range of 7 to 12 percent (Li and others 2000), with estimated gains in harvest value exceeding 20 percent (Fox and others 2007).

By the early 2000s, over half of all southern pine planting stock was coming from second-generation seed orchards. Estimates of average volume gain from second-generation plantations range from 7 to 23 percent over first-generation stock, to 10 to 35 percent over unimproved stock (Fox and others 2007; Li and others 1997, 1999, 2000; McKeand and others 2003, 2006a). Improvements in fusiform rust resistance, stem straightness, and wood quality further increase harvest values over unimproved stock.

Realized genetic gains from open-pollinated seedlings can be further increased by planting seedlings in single half-sib family blocks through the selection of female parent trees exhibiting greater breeding values (Duzan and Williams 1988, McKeand and others 2006b). 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 2006a). Further genetic gains can be made by planting full-sib families produced using mass-controlled pollination (MCP) techniques (also known as supplemental-mass pollinations) (Bramlett 2007). Crossing elite parents can produce volume gains of up to 30 percent, although actual gains are typically reduced somewhat due to pollen contamination (Sutton 2002). Jansson and Li (2004) showed potential volume gains from full-sib families of up to 60 percent over unimproved stock, with realized gains dependant on the selection intensity of the specific cross.

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Clonal forestry offers the hope of providing even greater genetic gains in forestry through mass propagation of highly selected genotypes. The most commonly used technique for many conifers has been rooted cuttings. However, this technique allows mass production of a given genotype for a limited number of years due to problems associated with tissue maturation (Park 2002, Sutton 2002). 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. However, recent advances in techniques of SE and cryopreservation have increased the potential for clonal, or varietal, southern pine planting stock. An important 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).

Varietal planting stock currently accounts for only a minor proportion of loblolly pine seedlings planted in the Southeast, and there remain issues that need further investigation. Field testing of varietal lines is needed in order to realize the potential genetic gains of varietal forestry. In addition, varietal planting stock is currently much more expensive than other options. 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 costs. How much greater yield can be expected from varietal stock? Can varietal stock be tailored to suit specific desired end products? Are there genotypes by site interactions that can be exploited with varietal stock? The answers to these and other questions are needed before large-scale adoption of varietal planting stock can be expected.

Here we describe second-year results from a study designed to examine differences in growth and form among loblolly pine planting stock of three different levels of genetic improvement. Height growth of SE varietal "seedlings" is compared to that of a second-generation, open-pollinated (second-Gen) half-sib family and a full-sib family produced via MCP techniques. The long-term objectives of the study are to determine if the best performing SE varietal stock will outperform the second-Gen and the MCP stock; and if so, how much more could a landowner afford to spend on varietal planting stock. The specific objective of this analysis was to examine early height growth trends among the three stock types.

METHODS

This study was established in 2007 at the Mississippi State University, North Mississippi Branch Experiment Station near Holly Springs, MS. Soils on the site are a Loring silt loam. The site had previously been a pasture, thus, the soils were somewhat compacted.

The study was set up in a complete block design containing three distinct levels of genetic improvement in loblolly pine planting stock: open-pollinated second-generation (second-Gen), MCP, and varietal stock produced using SE techniques. The second-Gen material, planted as bare-root stock, was a selected MeadWestvaco half-sib family that has been found to perform well in southwest Tennessee. The MCP material, also a MeadWestvaco selection, and SE material, provided by ArborGen LLC, were both produced in containers. Each of the 6 replicated blocks contained a single 100-tree plot of each stock type, with a 64-tree internal measurement plot.

In January 2007, prior to planting, the site was subsoiled to a depth of approximately 18 inches. A banded application of 64 ounces per acre glyphosate occurred in early March to eliminate existing herbaceous vegetation. Plots were handplanted on March 23–24, 2007, at a density of 403 trees per acre (12 by 9 foot spacing). In May 2008, the site received a broadcast application (14 ounces per acre) of Oustar®. 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.

Fifty-six SE varieties were included in the study, with 1 ramet of each variety included in each varietal treatment plot. The remaining trees in the 64-tree plots included checks and filler trees. Only varieties with at least four of the original six ramets surviving after the second growing season were included in this analysis. Standard analysis of variance techniques were used to determine whether there were significant differences in average height among the second-Gen, MCP, and the SE stock. In addition, the SE varieties were ranked by age 2 height, and the average height of the five tallest varieties was examined relative to the other stock types.

RESULTS

At the end of first growing season, there were no significant differences in height among the three genetic stock types (table 1). The MCP trees were slightly taller on average with a mean height of nearly 2.0 feet, while the second-Gen and varietal stock both had average heights of about 1.9 feet.

The MCP stock showed the greatest amount of height growth in year 2, and by the end of second growing season had increased its height advantage (table 1). Average second-year height of MCP material was over 5.4 feet, which was significantly taller than the SE varietal stock by over 0.5 feet. The maximum height attained by any MCP tree at age 2 was 8.1 feet. The maximum height for any of the varietal and second-Gen trees was 7.7 feet and 7.6 feet, respectively.

At the end of first year, differences in mean height among the individual varieties were minimal. By second year, these differences had increased, although few of the height differences were statistically significant (fig. 1). Only one of the five tallest varieties after the first year remained among the tallest five varieties following year 2. Four of the varieties had a mean height greater than the 7-56 OP check after year 2, and five varieties had a mean height greater than the average of the MCP material, although these differences were not significant (fig. 1).

By the end of the second growing season, the top five performing varieties were 18 percent taller than the overall average height of all varieties. The top five varieties were 14

Table 1—Mean height following the first- and second-growing seasons for loblolly pine derived from three genetic stock types

Genetic type	Height – age 1	Height – age 2	Growth
	---------- *feet* ----------		
Mass-controlled pollination	1.98 a	5.41 a	3.43 a
Second-generation open-pollinated	1.90 a	5.02 ab	3.13 ab
Somatic embryogenesis varietals	1.88 a	4.85 b	2.96 b

Numbers within the same column followed by different letters are significantly different at alpha = 0.05.

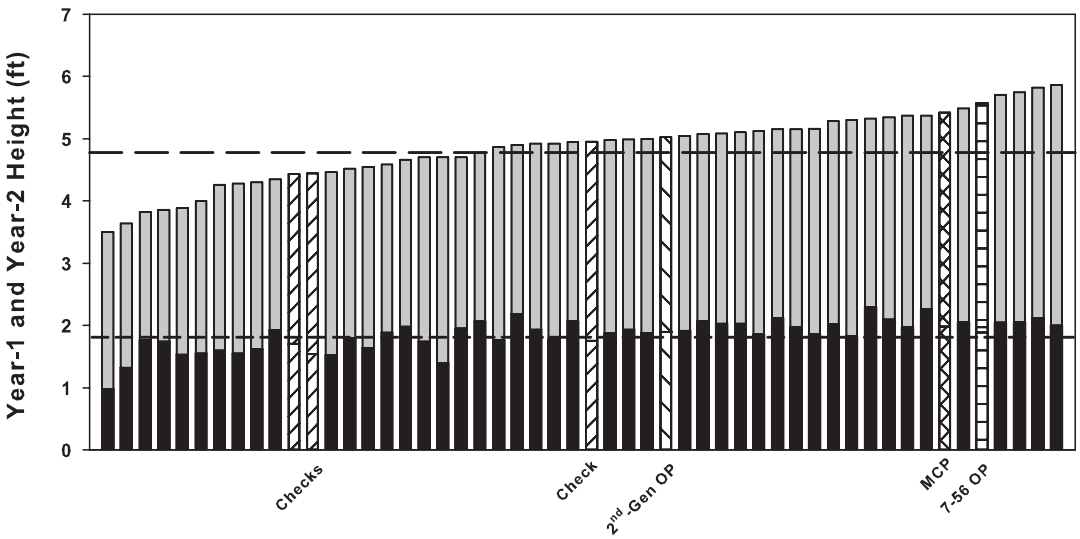

Figure 1—Forty-six somatic embryogenesis (SE) varieties of loblolly pine ranked by average height following year 2. Black portion of the each bar represents the average height of the variety following year 1. Horizontal dashed lines show the mean height of all 46 varieties following years 1 and 2. Crosshatched bars show the mean height following year 2 for the second-generation seedings, the mass-controlled pollination seedlings, nursery checks, and a 7-56 OP family.

percent taller than the average height of the second-Gen material, and 6 percent taller than the average height of the MCP material (fig. 2). Differences between the varietals and the MCP were not significant.

DISCUSSION

The knowledge and technologies needed to mass produce loblolly pine somatic seedlings are currently available, although current cost of production remains relatively high (Dougherty and Wright 2009, Wright and Dougherty 2006). There is an ongoing need for additional studies addressing the economic effectiveness of varietal stock. There is also a continued need to field test clonal varieties to determine which individual genotypes are best suited for mass production of planting stock for specific characteristics.

Currently, companies producing varietal stock produce thousands of varietal genotypes from several dozen elite parental crosses. Based on results from extensive field testing, fewer than 5 percent of these varietals will likely be selected for mass production of varietal planting stock (Sutton 2002).

Some estimates predict that clonal selections may produce volume gains 40 to 50 percent greater than the average of open-pollinated second-generation seed orchards (Sutton 2002). After two growing seasons, the varietal material in our study is not, on average, outperforming the other two genetic stock types. In fact, the average height of the MCP material is statistically greater than that of the varietal material. However, the best performing varieties are growing better than the

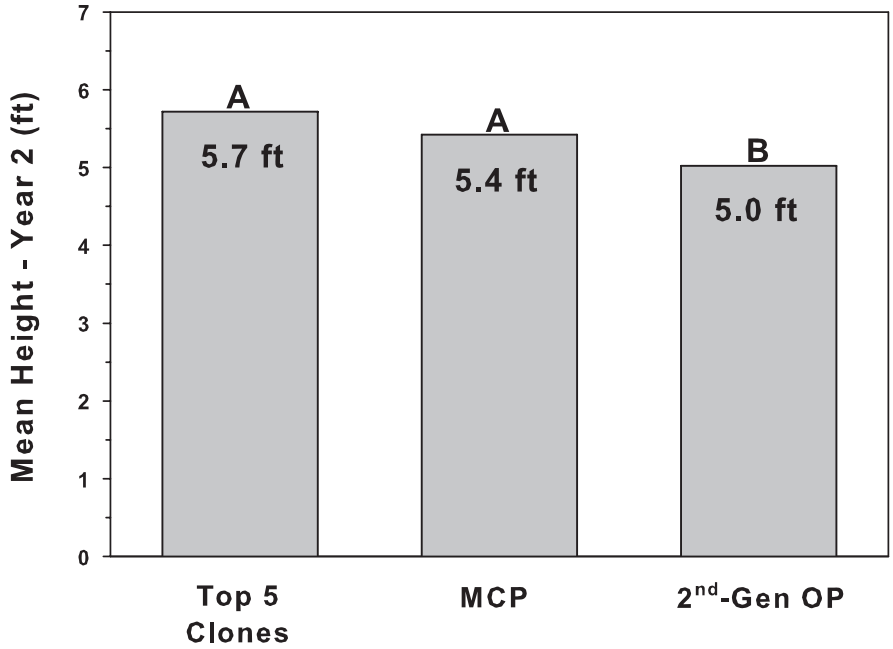

Figure 2—Average height following the second growing season for the top five performing somatic embryogenesis varieties compared to the average height of the mass-controlled pollination and second-generation genetic stock. Bars with different letters above indicate mean heights are significantly different at alpha = 0.05.

more common stock types. Our results showed that the top 10 percent of the varieties tested were outperforming both the second-Gen and the MCP material.

As expected, we observed considerable variability in growth among the 56 varieties tested in this study. This variability is due to the specific crosses that made up these varieties, as well as physiological differences associated with production of the somatic tissue. However, we feel that this test, when combined with data from many others across the Southeastern United States, will provide information on specific varieties suited to the local area.

Two years is certainly very early to draw firm conclusions about the performance of individual varieties. The relative performance of these varieties may change in future years, much like they did between the first and second year when only one of the best growing varieties after year 1 remained among the tallest varieties after the second year. However, our early results are beginning to suggest that the best varietal material may be able to outperform MCP stock, and especially the standard second-Gen stock, on these sites.

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LITERATURE CITED

Adams, D.M.; Haynes, R.W.; Daigneault, A.J. 2006. Estimated timber harvest by U.S. region and ownership, 1950-2002. Gen. Tech. Rep. PNW-GTR-659. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 64 p.

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

Dougherty, D.; Wright, J. 2009. Improved returns on forestlands: a financial analysis of mass control pollinated and varietal seedlings. Tree Farmer. January/February: 42–46.

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: Proceedings of the 24th southern tree improvement conference. Gainesville, FL: University of Florida: 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.

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.

Park, Y.-S. 2002. Implementation of conifer somatic embryogenesis in clonal forestry: technical requirements and deployment considerations. Annals of Forest Science. 59: 651–656.

Sutton, B. 2002. Commercial delivery of genetic improvement to conifer plantations using somatic embryogenesis. Annals of Forest Science. 59: 657–661.

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

Wright, J.; Dougherty, P. 2006. Varietal forestry: a giant step-up for increasing timber value on your land. Forest Landowner. September/October: 3–4.

REGENERATING SHORTLEAF PINE IN CLEARCUTS IN THE MISSOURI OZARK HIGHLANDS

David Gwaze and Mark Johanson[1]

Abstract—A shortleaf pine (*Pinus echinata* Mill.) regeneration study was established by the Missouri Department of Conservation in 1986 at the Current River Conservation Area. The objective of the study was to compare natural to artificial regeneration methods, and site preparation prescribed burning to bulldozing for shortleaf pine establishment and growth. Eighteen years after establishment, the control treatment (natural regeneration) had only 94 stems/ha, the burn treatment had 727 stems/ha, and the doze treatment had 1,680 stems/ha. Mean volume growth per tree was greatest in the doze treatment (53.7 dm^3) followed by the burn treatment (34.1 dm^3) with the control having the least volume growth (22.3 dm^3). Hardwood competition was greatest in the control treatment (3,132 stems/ha) followed by the burn treatment (2,470 stems/ha) and least in the doze treatment (1,210 stems/ha). The results suggest that (1) survival and growth of shortleaf pine increases with increase in site preparation intensity, (2) natural regeneration may not achieve stocking goals and adequate growth, and (3) prescribed burning is a viable site preparation method.

INTRODUCTION

Shortleaf pine (*Pinus echinata* Mill.) dominated southern Missouri, but today it only occupies <10 percent of its original range. Sites formerly occupied by shortleaf pine have many hardwood species, many of which are not as well adapted as shortleaf pine to the dry, nutrient-poor, and eroded sites. These less adapted hardwood species are experiencing problems with oak decline associated with red oak borers and Armillaria root rot. The decline and mortality is affecting mostly black oak (*Quercus velutina* Lam.) and scarlet oak (*Q. coccinea* Münchh.). It is estimated that 200 000 ha of forest is affected by severe oak decline on the Mark Twain National Forest (Law and others 2004). Restoring shortleaf pine on former pine and oak-pine sites is a long-term strategy for mitigating chronic oak decline (Law and others 2004). Shortleaf pine restoration is being achieved through natural or artificial regeneration. Natural regeneration is sometimes preferred because it has lower establishment and capitalization costs than artificial regeneration (Vesikallic 1981). When harvesting does not coincide with a good seed crop or where few seed trees exist, artificial regeneration may be preferred. A good seed crop occurs once every 5 to 7 years in shortleaf pine (Brinkman and Rogers 1967). The uncertainty regarding predicting good shortleaf pine seed crops limits the success of natural regeneration. Artificial regeneration also provides an opportunity to improve productivity by planting improved shortleaf pine seedlings, and it is a more precise method than natural regeneration for obtaining stocking goals. Although both natural and artificial regeneration methods are currently used for restoring shortleaf pine in the Missouri Ozarks, there is lack of information on their comparative effectiveness.

Adequate site preparation is critical for germination of seeds, and survival and growth of shortleaf pine seedlings. For successful germination and establishment, the heavy hardwood leaf litter, forbs, woody vegetation, and grass should be eliminated or reduced. Common site preparation methods include prescribed burning, and mechanical and chemical treatments. Prescribed burning or mechanical disturbances are effective site preparation methods because they remove the dense leaf litter, grass, and vegetation, and expose seed to the mineral soil. Chemical methods on the other hand do not expose mineral soil but are effective in reducing or eliminating competing vegetation. In Oklahoma at least three times as many seedlings emerged on burned sites than on unburned sites (Boggs and Wittwer 1993). In contrast, Yocom and Lawson (1977) found that prescribed burning provided little additional seedbed benefit in sites disturbed by logging in Arkansas. In the Missouri Ozarks, cultivation after removal of litter was a superior site preparation method compared to burning or raking (Liming 1945). These contrasting results point to the need for more information on effectiveness of the different site preparation methods. Progress in restoring shortleaf pine will probably be greatly accelerated when information on the efficacy of the regeneration and site preparation methods are made available to resource managers.

The objective of the study was to compare natural to artificial regeneration methods, and site preparation prescribed burning to bulldozing for shortleaf pine establishment and growth. We also evaluated the effectiveness of the different treatments on controlling hardwood competition.

MATERIALS AND METHODS

Study Site

The study is located in compartment 15 on the Current River Conservation Area of the Missouri Department of Conservation (fig. 1). The Current River Conservation Area is located in Reynolds and Shannon Counties located in southeast Missouri. The conservation area is approximately 11 300 ha of forest land. The study sites are located completely within the Current River and Black River Oak/

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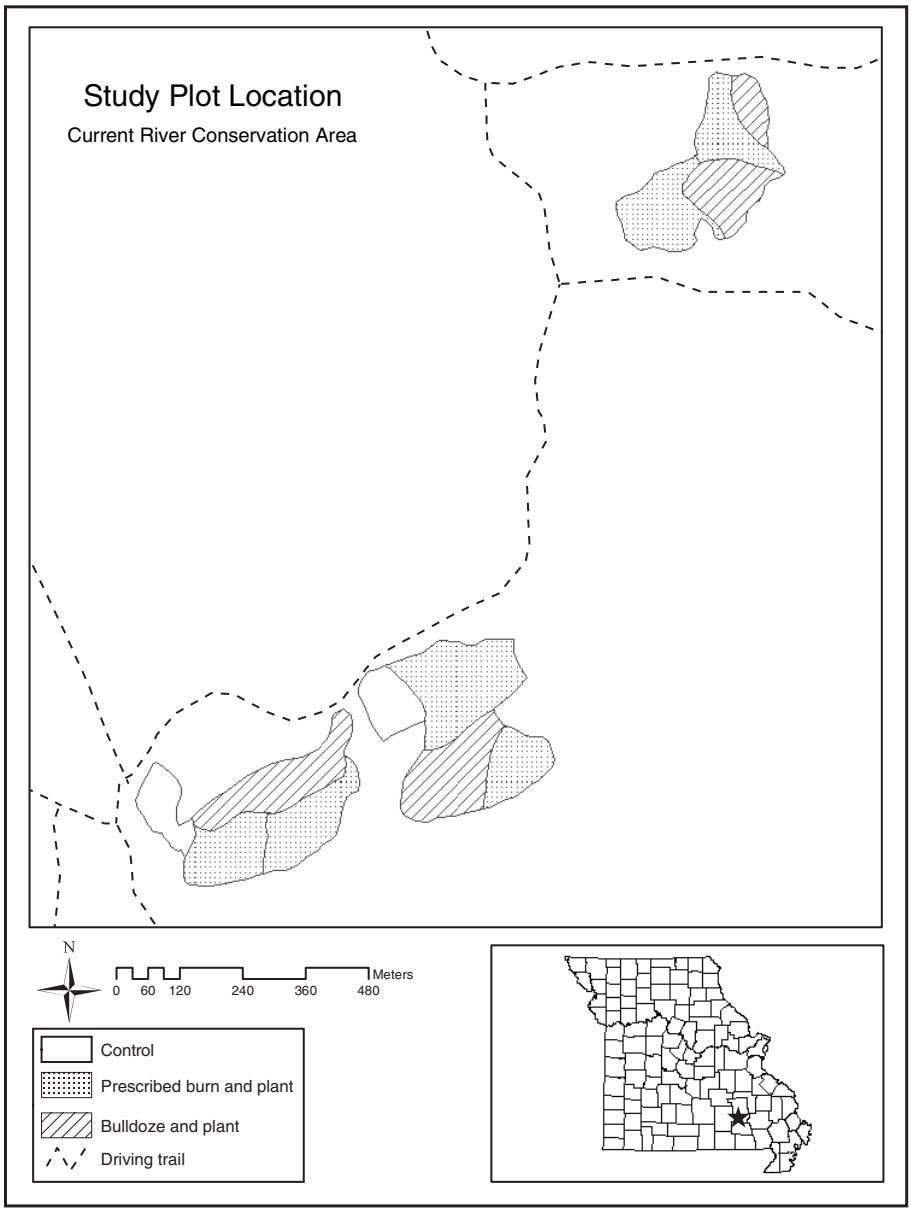

Figure 1—Location of study site.

Pine Woodland/Forest Hills Land Type Association (Nigh and Schroeder 2002). These land types are characterized by hilly landscapes with narrow ridges, narrow valleys, and steep slopes with 46 to 76 m of local relief. The ridges and upper slopes are formed from the Roubidoux Formation whereas the lower hillslopes and valleys cut into the Gasconade Formation. Historically, this area was dominated by shortleaf pine and shortleaf pine-oak woodland complexes.

Compartment 15 had several pockets of oak decline during the mid-1980s. Timber was salvaged from these pockets between December 1984 and May 1985. The timber harvest was advertised and sold as three sale units, measuring 10, 18, and 25 ha. Each sale unit was composed of 4 to 11 stands. On several stands in two larger sale units an attempt was made to regenerate shortleaf pine using different regeneration and site preparation methods.

Site Preparation, Planting, and Assessment

Treatments included burn, doze, and control. The burn treatment included complete overstory removal followed by a spring burn. The doze treatments consisted of complete removal of the overstory followed by stump and slash removal using a bulldozer. After the burn and doze treatments were applied, the stands were handplanted with unimproved 1-0 shortleaf pine bare-root seedlings at a spacing of 2.4 by 2.4 m in spring 1986. The control treatment consisted of complete removal of the overstory, and no planting or site preparation was carried out. Layout of the stands is shown in figure 1. Groups of stands were allocated in blocks by spatial proximity.

The blocking allowed for replication of the three treatments in space. Three blocks were identified, one block located in the north with two treatments represented (burn and doze) and two blocks located in the south with all three treatments represented in each block (fig. 1).

Two random plots were established in each treatment within a block in July 2004. Each plot was 10 by 10 m. The number of trees, species, height, and diameter at breast height (d.b.h.) of all trees in a plot were measured. Height was measured using height poles and d.b.h. (cm) was measured using diameter tapes. Conical volume (V, dm^3 per tree) was calculated for all trees using the equation:

$$V = \frac{1}{3}\pi\left(\frac{D}{2}\right)^2 H$$

where
D = d.b.h. (dm)
H = height (dm)

Statistical Analysis

Plot means were used for all analyses. Using the PROC GLM procedure in SAS version 9.1 (SAS Institute Inc., Cary, NC), analysis of variance (ANOVA) was used to test for significant differences among blocks and treatments for height, diameter, and volume. Stocking was analyzed using chi-square test. All analyses were carried out at the $P \leq 0.1$ probability level.

RESULTS

Stocking and Growth

Eighteen years after applying the treatments, naturally regenerated shortleaf pine in the control treatment had an average density of only 94 stems/ha, a significantly lower density ($P < 0.001$) than in the burn treatment (727 stems/ha) and doze treatment (1,680 stems/ha) (fig. 2). Survival in the doze treatment was significantly higher than in the burn treatment ($P < 0.001$).

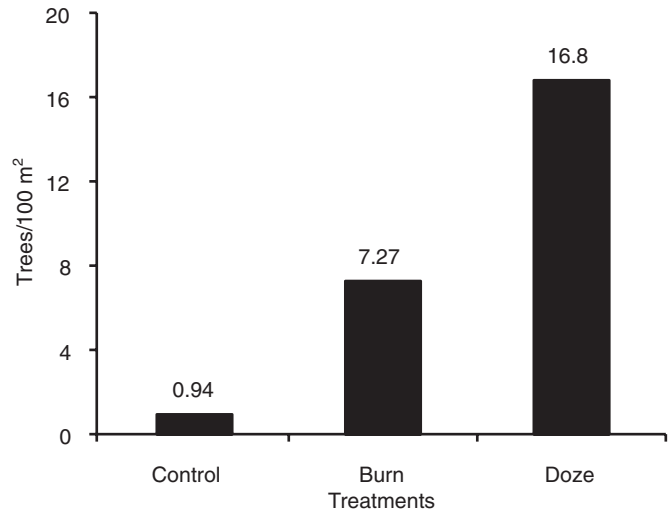

Figure 2—Stocking of shortleaf pine trees 18 years after establishment. Control = no site preparation and no planting, burn = prescribed burn and plant, and doze = bulldoze and plant.

Bulldozing increased height growth by 27.3 percent, diameter by 27.5 percent, and volume by 140.8 percent over the control treatment (table 1). Bulldozing increased height growth by 20.1 percent, diameter by 18.1 percent, and volume by 57.4 percent over the burn treatment. Although the burn treatment increased height by 6.0 percent, diameter by 7.9 percent, and volume by 89.6 percent over the control, the increases were not statistically significant.

Table 1—Treatment effects on height, diameter, and volume at 18 years for a shortleaf pine clearcut study

Regeneration technique[a]	Height	D.b.h.	Volume
	m	*cm*	*dm^3*
Control	7.5	10.0	22.3
Burn	7.9	10.8	34.1
Doze	9.5	12.8	53.7
MSE	1.35	3.02	250.85
Treatment contrasts	---------- *P-value* ----------		
Control vs. burn	0.595	0.357	0.283
Control vs. doze	0.043	0.038	0.025
Burn vs. doze	0.027	0.063	0.050

[a] Control = no site preparation and no planting; burn = prescribed burn and plant; doze = bulldoze and plant; MSE = mean square error.

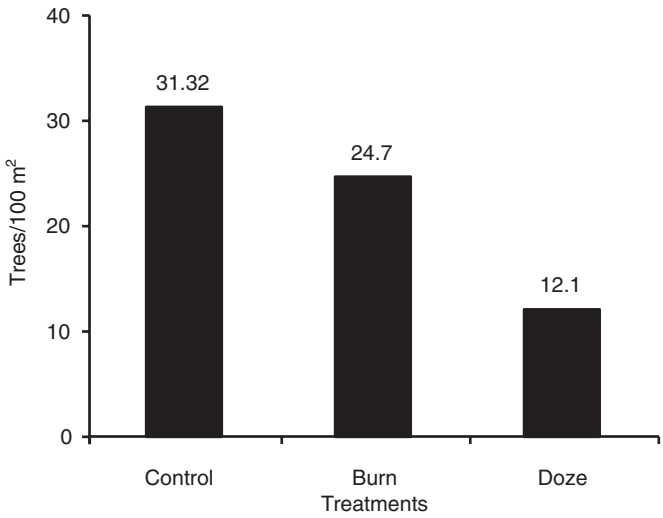

Figure 3—Density of hardwoods 18 years after establishment. Control = no site preparation and no planting, burn = prescribed burn and plant, and doze = bulldoze and plant.

Hardwood Competition

The control treatment had a significantly higher density of hardwood competitors than the burn treatment (27 percent higher, $P = 0.092$) and a significantly higher density of hardwood competitors than the doze treatment (150 percent higher, $P < 0.001$) (fig. 3). Density of naturally regenerated hardwoods was significantly higher in the burn treatment than in the doze treatment (100 percent higher, $P < 0.001$). Hardwoods were taller in the control treatment (6.28 m) than in the burn (5.96 m) and doze treatments (5.88 m). Hardwoods in the doze treatment were smaller in diameter (4.86 cm) than those in the burn and control treatments (5.53 and 5.75 cm, respectively). Species richness within the 3 treatments was similar (10 to 11 hardwood species). The dominant oak species occurring across all treatments was white oak (*Q. alba* L.) followed by post oak (*Q. stellata* Wangenh.), scarlet oak, and black oak had the least average density.

DISCUSSION

Eighteen years after establishment, stocking of naturally regenerated shortleaf pine was low and inadequate to meet the minimum stocking goals for a shortleaf pine forest (1,000 stems per acre) or shortleaf pine-oak forest (500 stems/ha). The low density of naturally regenerated shortleaf pine trees was most likely due to harvesting not coinciding with a good seed crop. It may also be attributed to inadequate seedbed preparation, insufficient moisture for germination and seedling establishment, or excessive competition. Naturally regenerated loblolly pine (*P. taeda* L.) seedlings released from woody and herbaceous competition have been reported to have better survival and more vigor than those not released on a site in southern Arkansas (Cain and Barnett 1996). At our study site, stands naturally regenerated required followup release to increase vigor of shortleaf pine trees. Underplanting or direct seeding was necessary to increase the stocking of shortleaf pine in naturally regenerated stands. However, dozed and prescribed burned stands did not need any followup release or enrichment planting because both treatments achieved higher than the minimum desired stocking for pine-oak forest.

Although the doze treatment had better survival and less hardwood competitors than the burn treatment, the burn treatment was successful at achieving the desired stocking goals. Furthermore, growth achieved by the burn treatment was comparable to that achieved by the doze treatment. Thus, planting on sites prepared by prescribed burning appears to a viable method of restoring shortleaf pine. Gwaze and others (2006) found that prescribed burning was as effective as subsoiling as a site preparation method at a site in the Salem Ranger District, Mark Twain National Forest. Our results are not consistent with other studies in Florida where burning did not improve survival or growth of slash pine (*P. elliottii* Engelm.) 10 years after planting, but mechanical site preparation did (Outcalt 1983). The effectiveness of prescribed burning can be unpredictable due to varying amounts and types of fuel, slope and aspect, and unstable weather conditions.

The stocking of naturally regenerated shortleaf pine of 94 trees/ha (38 trees per acre) is below the minimum required for a fully stocked shortleaf pine stand or for a pine-oak mixed stand in Missouri. In Missouri a minimum of 400 or 200 shortleaf pine trees in a free-to-grow condition is required to fully stock a pine stand or pine-oak stand at 5 years. Thus, results from this study suggest that natural regeneration may not meet adequate stocking goals and seedlings regenerated by this method exhibit poor growth. Because natural regeneration continues to be important for restoring shortleaf pine in the Missouri Ozarks, and is desirable for landowners who prefer low-cost establishment methods, more effective techniques to establish and recruit naturally regenerated seedlings need to be identified. Also, methods for predicting seed yields need to be developed so that harvesting can be planned to coincide with a good seed crop. The study suggests that planting on sites prepared by prescribed burning is a viable method of restoring shortleaf pine.

ACKNOWLEDGMENTS

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LITERATURE CITED

Boggs, J.A.; Wittwer, R.F. 1993. Emergence and establishment of shortleaf pine seeds under various seedbed conditions. Southern Journal of Applied Forestry. 17(1): 44–48.

Brinkman, K.A.; Rogers, N.F. 1967. Timber management guide for shortleaf pine and oak-pine types in Missouri. Res. Pap. NC-19. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 15 p.

Cain, M.D.; Barnett, J.P. 1996. An 8-year field comparison of naturally seeded to planted container *Pinus taeda*, with and without release. Canadian Journal of Forest Research. 26: 1237–1247.

Gwaze, D.P.; Melick, R.; McClure, L. [and others]. 2006. Effects of site preparation sub-soiling and prescribed burning on survival and growth of shortleaf pine in the Mark Twain National Forest: results after 20 growing seasons. 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: 129–133.

Law, J.R.; Melick, R.; Studyvin, C.; Steinman, J.R. 2004. Oak decline in Missouri: history revisited. In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 257–258.

Liming, F.G. 1945. Natural regeneration of shortleaf pine in the Missouri Ozarks. Journal of Forestry. 43(5): 339–345.

Nigh, T.A.; Schroeder, W.A. 2002. Atlas of Missouri ecoregions. Jefferson City, MO: Missouri Department of Conservation. 212 p.

Outcalt, K.W. 1983. Mechanical site preparation improves growth of genetically improved and unimproved slash pine on a Florida flatwoods site. In: Jones, P., ed. Proceedings of the second biennial southern silvicultural conference. Gen. Tech. Rep. SE-24. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 11–13.

Vesikallic, H. 1981. The economy of artificial and natural regeneration. In: Symposium of engineering systems for forest regeneration. Publ. 10-81. American Society of Agricultural and Biological Engineers: 266–270.

Yocom, H.A.; Lawson, E.R. 1977. Tree percent from naturally regenerated shortleaf pine. Southern Journal of Applied Forestry. 1(2): 10–11.

HARDWOOD REGENERATION

Multiple age classes of oaks in an uneven-aged upland oak stand, Pioneer Forest, near Salem, Missouri. (Photo by James M. Guldin)

REGENERATION DYNAMICS DURING OAK DECLINE WITH PRESCRIBED FIRE IN THE BOSTON MOUNTAINS OF ARKANSAS

Martin A. Spetich[1]

Abstract—Northern red oak (*Quercus rubra* L.) seedlings were inventoried in 2000 and again in 2005 to better understand survival of advance regeneration during oak decline. In 2000, basal stem diameter was measured and recorded for 861 individually tagged seedlings that were <5 cm d.b.h. By mid-2001, the stand containing these seedlings began to exhibit symptoms of severe oak decline. In 2004, a prescribed fire was applied to one-third of the study area. Based on logistic regression analysis, survival probability of northern red oak increased with increasing initial basal stem diameter and decreasing site index. In unburned areas, greatest mortality occurred on seedlings with basal stem diameters <7 mm. However, in burned areas, seedlings <12 mm exhibited the greatest mortality. This information can aid managers in developing prescriptions that help mitigate the impacts of oak decline by promoting practices that maximize oak regeneration to basal stem diameters >7 mm in areas not burned and to diameters >12 mm in areas to be burned.

INTRODUCTION

Drought is an inciting factor of oak decline (Manion 1991, Starkey and others 2004). A 3-year drought occurred across the Interior Highlands region of Arkansas and Missouri from 1998 to 2000. Through much of the Boston Mountain forests that were impacted by oak decline, this was coupled with stands of high tree density and mature trees, making these forests especially vulnerable to oak decline (Oak and others 2004). These and other factors led to an oak decline event across the Ozark Highlands in Arkansas and Missouri (Starkey and others 2004).

At the research site discussed in this paper, oak decline lasted through 2005 and altered snag (Spetich 2004a, 2006), overstory (Spetich 2006), and down-deadwood dynamics (Spetich 2007). However, regeneration dynamics have not been examined.

Fire was a once-frequent and important part of these ecosystems. However, during much of the past century, these once-frequent fires have been reduced to infrequent events (Guyette and Spetich 2003). Fire suppression has altered species dynamics of these upland hardwood forests (Spetich 2004b).

Understanding how to predict survival of oak (*Quercus* spp.) advance regeneration provides important information that managers can use to more effectively restore this valuable resource to the landscape. The objectives of this paper are to (1) identify predictors of survival of oak advance reproduction on a site with severe oak decline in burned and unburned areas, (2) provide survival models based on this information, and (3) develop management recommendations based on this information.

STUDY SITE

The study site is a 32-ha area in an upland oak-hickory (*Carya* spp.) stand that was approximately 75 years old in 2005. It is located in the Boston Mountains of Arkansas, part of the southern lobe of the central hardwood region (Merritt 1980). The Boston Mountains are the highest and most southern member of the Ozark Plateau Physiographic Province (Croneis 1930). They form a band 48 to 64 km wide and 320 km long from northcentral Arkansas westward into eastern Oklahoma. Elevations range from about 275 m in the valley bottoms to 760 m at the highest point. The plateau is sharply dissected. Most ridges are flat to gently rolling and generally are <0.8 km wide. Mountainsides are alternating steep simple slopes and gently sloping benches. Vegetation across much of the landscape is a forest matrix with nonforest inclusions.

More specifically, the study site is located in the northwestern corner of Pope County, approximately 3 km southeast of Sand Gap, AR. The stand is dominated by oak and hickory and has become the center of a local patch of oak decline. In August 2000, mean basal area for all standing trees was 25.9 m^2/ha, and there were 417 standing trees/ha. Of those standing trees 1.8 m^2/ha of basal area and 53 trees/ha were standing dead trees mostly in smaller diameter classes. Stocking was 88 percent.

METHODS

From 2000 to 2005, measurements were taken on 861 northern red oaks (*Q. rubra* L.) <5 cm d.b.h. These seedlings were individually tagged and monitored. The study site was located in an oak-hickory dominated stand in the Boston Mountains of northern Arkansas. For each seedling, we measured basal stem diameter in mm and tracked survival status.

A prescribed fire was applied to one-third of the study site on March 12, 2004. The fire burned an area that included 268 of the seedlings. Fire weather conditions during the fire were—relative humidity, 28 percent; wind from the northeast

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at 2 to 6 km/hour; and ambient temperature averaged 21 °C. In 2005, we reinventoried all 861 seedlings for basal stem diameter and survival.

Site index was determined by measuring age and height of 48 dominant or codominant northern red oak and/or white oak (*Q. alba* L.). Site indices for red oaks were converted to white oak site indices using a published conversion formula (Carmean and Hahn 1983). The average of the nearest two site index trees was used to determine site index for each area of regeneration trees. None of the site index trees were used more than once. Site index trees were generally located within 22 m or less of regeneration trees.

Logistic regression was used to model survival of northern red oak in both the burned and unburned areas. Results are expressed as the probability of survival of northern red oak advance regeneration. The versatility of logistic regression to analyze dichotomous data first gained recognition in the 1960s (Hosmer and Lemeshow 1989: vii). Since then, researchers have used this method to examine a range of data types including human health issues, the growth and survival of naturally regenerated trees (Lowell and others 1987), and dynamics of artificial regeneration of northern red oak (Spetich and others 2002).

To evaluate logistic regression model performance, we selected predictors with a *p*-value of 0.05 or less based on the chi-square distribution with one degree of freedom. We used the Hosmer-Lemeshow goodness-of-fit statistic (Hosmer and Lemeshow 1989: 140) to test the null hypothesis that the equation described the data. For Hosmer-Lemeshow goodness-of-fit, the null hypothesis was rejected for *p*-values of 0.05 or less (indicating a poor fit of the equation to our data). Consequently, it is important to note that predictor *p*-values of 0.05 or less have a different interpretation than the Hosmer-Lemeshow goodness-of-fit *p*-values of 0.05 or less.

RESULTS

Survival probability of northern red oak seedlings increased with increasing basal stem diameter and decreasing site index. The models are presented below:

Burned area, model 1:

$$PSb = 1/(1 + EXP(-(1.255-(0.178 * SI(m) * (1/BSD)))))$$

PSb = probability of survival in burned area
BSD = basal stem diameter in 2000, mm
SI = site index for white oak, m

Predictor [SI(*m*) * (1/BSD)] *p*-value = <0.001
Goodness-of-fit[2] (*p*-value) = 0.443

Unburned area, model 2:

$$PSu = 1/(1 + EXP(-(1.293 - (0.0675 * SI(m) * (1/BSD)))))$$

PSu = probability of survival in unburned area
BSD = basal stem diameter in 2000, mm
SI = site index for white oak, m

Predictor [SI(*m*) * (1/*BSD*)] *p*-value = 0.03
Goodness-of-fit[2] (*p*-value) = 0.418

[2]Goodness-of-fit—Based on the Hosmer-Lemeshow goodness-of-fit statistic, differences between estimated probabilities and observed responses are not significant. Small *p*-values designate a poor fit of the equation to the data while large values (>0.05) indicate a good fit.

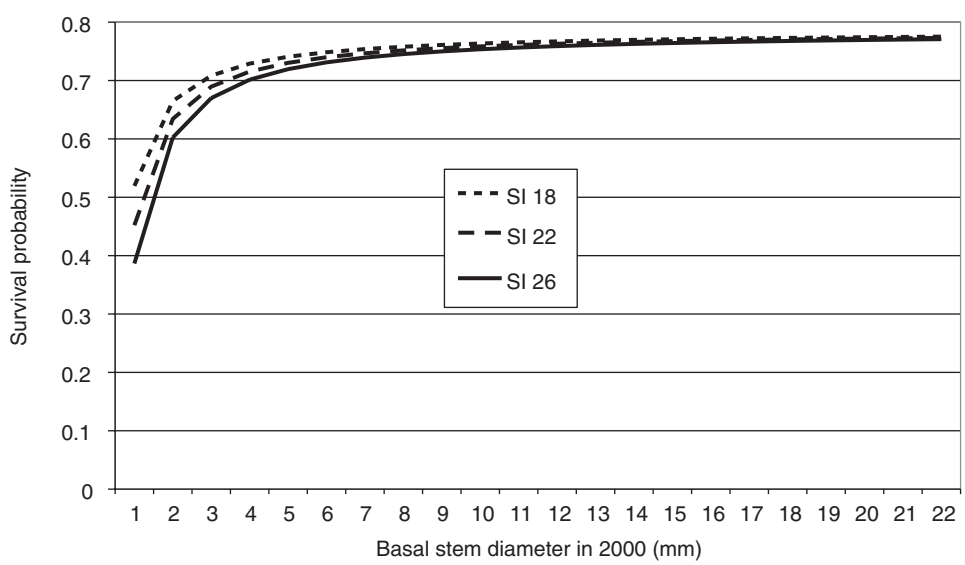

Figure 1—Survival of northern red oak advance regeneration in 2005 in unburned area, *n* = 539.

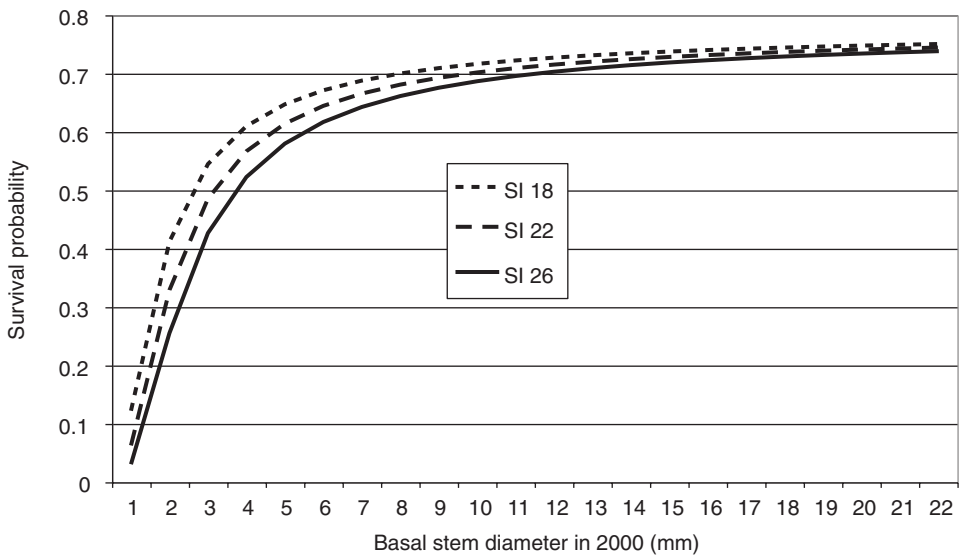

Figure 2—Survival of northern red oak advance regeneration in 2005 in the 2004 prescribed burn area, *n* = 268.

The relationship of survival, basal stem diameter, and site index is illustrated in figures 1 and 2. In both cases survival increased with increasing diameter and decreasing site index. For a given site index, northern red oak seedlings in the area receiving the 2004 prescribed fire had lower survival than seedlings of equal diameter in the unburned area (figs. 3 and 4).

DISCUSSION

A better understanding of regeneration dynamics on oak decline sites can help managers prepare prescriptions to help mitigate the impacts of oak decline and help restore a valuable species to the landscape. For seedlings with <7 mm basal stem diameter there was a greater difference in survival between the burned and unburned areas on the high site index site (fig. 4) compared to the low site index site (fig. 3). For seedlings with 1.0 mm basal stem diameter, survival probability in the burned areas was nearly zero (fig. 2) versus about 0.4 to 0.5 in the unburned area (fig. 1).

As basal stem diameter increased, survival probability increased. These results reinforce the importance of large basal stem diameter advance regeneration. In the unburned area, basal stem diameters of <7 mm rapidly decrease in survival probability with decreasing diameter (figs. 1, 3, and 4). However, on the site receiving the 2004 prescribed fire, basal stem diameters of <12 mm rapidly decrease in survival probability with decreasing diameter (figs. 2, 3, and 4). Beyond these benchmark diameters of 7 mm and 12 mm there were relatively small gains in survival with increasing diameter of natural regeneration.

The general relationship of increasing site index with a corresponding decrease in survival has been noted for underplanted northern red oak (Spetich and others 2002). A similar relationship exists for the relationship of decreasing dominance probability of natural regeneration with increasing site index (Loftis 1990, Sander and others 1984). Dominance probability has seedling survival incorporated into it.

Based on these results, on oak decline sites where fire is not a prescriptive option, managers should consider optimizing conditions to grow advance regeneration to basal stem diameters of >7 mm. Moreover, on areas where fire is a prescriptive option, managers should consider burning only after a significant proportion of advance oak regeneration reaches basal stem diameters of >12 mm in order to increase the probability of survival. Implementing these suggestions may help to mitigate northern red oak regeneration losses in the Boston Mountains during oak decline.

CONCLUSIONS

Because this is a long-term study with repeated fires, the results reported here are preliminary. Therefore, these results should be considered in the context of this single fire event. Results reinforce the importance of large basal stem diameter advance regeneration. In the unburned area, basal stem diameters of <7 mm rapidly decrease in survival probability with decreasing diameter. In the burned area, basal stem diameters <12 mm rapidly decrease in survival probability with decreasing diameter.

RECOMMENDATIONS

Prior to burning, assess oak regeneration and diameter distribution. Burn if oak regeneration is sufficiently established to meet management goals and if oak is well represented across diameter classes. If oak regeneration exists mainly in smaller diameter classes, postpone burning until regeneration is more evenly distributed.

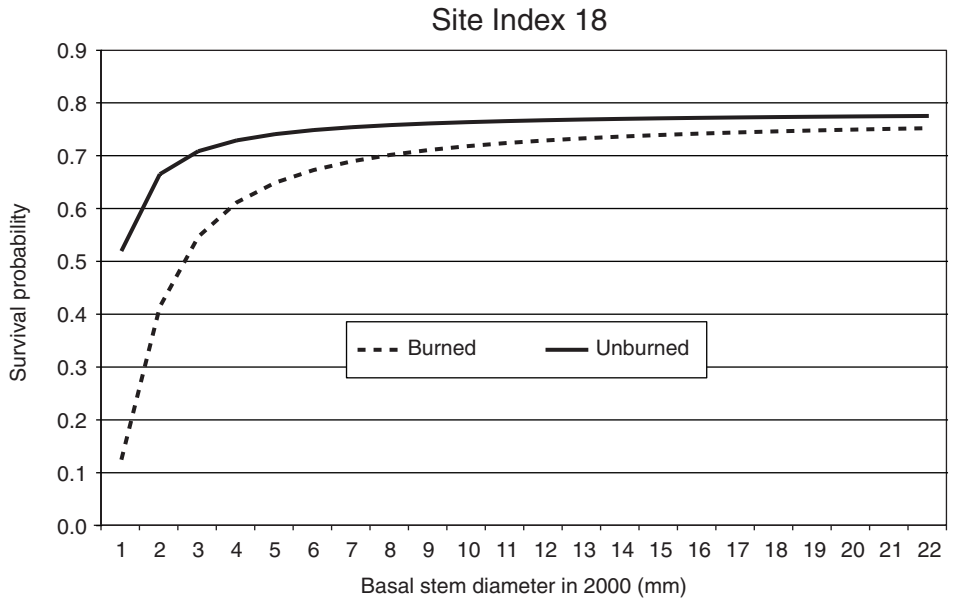

Figure 3—Advance regeneration on site index 18 m for white oak: survival of northern red oak advance regeneration in 2005.

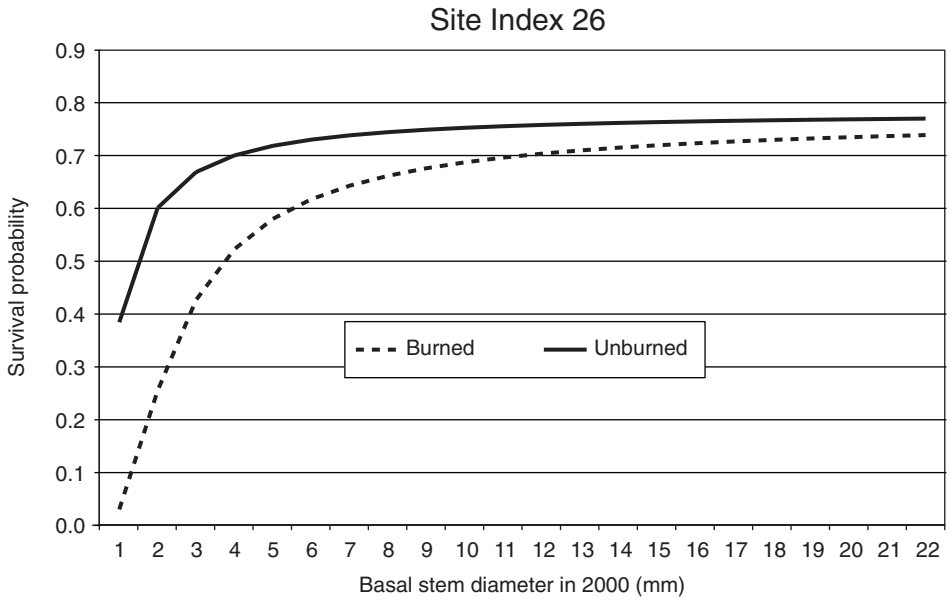

Figure 4—Advance regeneration on site index 26 m for white oak: survival of northern red oak advance regeneration in 2005.

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LITERATURE CITED

Carmean, W.H.; Hahn, J.T. 1983. Site comparisons for upland oaks and yellow-poplar in the central states. Journal of Forestry. 81: 736–739.

Croneis, C. 1930. Geology of the Arkansas Paleozoic area. Bull. 3. Arkansas Geological Survey. 457 p.

Guyette, R.P.; Spetich, M.A. 2003. Fire history of oak-pine forests in the lower Boston Mountains, Arkansas, USA. Forest Ecology and Management. 180: 463–474.

Hosmer, D.W.; Lemeshow, S. 1989. Applied logistic regression. New York: John Wiley. 307 p.

Loftis, D.L. 1990. Predicting post-harvest performance of advance red oak reproduction in the Southern Appalachians. Forest Science. 36: 908–916.

Lowell, K.E.; Mitchell, R.J.; Johnson, P.S. [and others]. 1987. Predicting growth and "success" of coppice-regenerated oak stems. Forest Science. 33: 740–749.

Manion, P.D. 1991. Tree disease concepts. 2d ed. Englewood Cliffs, NJ: Prentice-Hall. 402 p.

Merritt, C. 1980. The central region. In: Barrett, J.W., ed. Regional silviculture of the United States. 2d ed. New York: John Wiley: 107–143.

Oak, S.W.; Steinman, J.; Starkey, D.A.; Yockey, E.K. 2004. Assessing oak decline incidence, distribution, and severity in the Southeastern U.S. using forest inventory and analysis data. In: Spetich, M.A., ed. Proceedings of the upland oak ecology symposium: history, current conditions and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 236–242.

Sander, I.L.; Johnson, P.S.; Rogers, R. 1984. Evaluating oak advance reproduction in the Missouri Ozarks. Res. Pap. NC-251. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 17 p.

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

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

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

Spetich, M.A. 2007. Down deadwood dynamics on a severely impacted oak decline site. In: Buckley, D.S.; Clatterbuck, W.K., eds. Proceedings, 15th central hardwood forest conference. Gen. Tech. Rep. SRS-101. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 206–213.

Spetich, M.A.; Dey, D.C.; Johnson, P.S.; Graney, D.L. 2002. Competitive capacity of *Quercus rubra* L. planted in Arkansas' Boston Mountains. Forest Science. 48: 504–517.

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

REGENERATION IN BOTTOMLAND FOREST CANOPY GAPS 6 YEARS AFTER VARIABLE RETENTION HARVESTS TO ENHANCE WILDLIFE HABITAT

Daniel J. Twedt and Scott G. Somershoe[1]

Abstract—To promote desired forest conditions that enhance wildlife habitat in bottomland forests, managers prescribed and implemented variable-retention harvest, a.k.a. wildlife forestry, in four stands on Tensas River National Wildlife Refuge, LA. These treatments created canopy openings (gaps) within which managers sought to regenerate shade-intolerant trees. Six years after prescribed harvests, we assessed regeneration in 41 canopy gaps and 4 large (>0.5-ha) patch cut openings that resulted from treatments and in 21 natural canopy gaps on 2 unharvested control stands. Mean gap area of anthropogenic gaps (582 m^2) was greater than that of natural gaps (262 m^2). Sweetgum (*Liquidambar styraciflua*) and red oaks (*Quercus nigra*, *Q. nuttallii*, and *Q. phellos*) were common in anthropogenic gaps, whereas elms (*Ulmus* spp.) and sugarberry (*Celtis laevigata*) were numerous in natural gaps. We recommend harvest prescriptions include gaps with diameter >25 m, because the proportion of shade-intolerant regeneration increased with gap area up to 500 m^2. The proportion of shade-intolerant definitive gap fillers (individuals likely to occupy the canopy) increased with gap area: 35 percent in natural gaps, 54 percent in anthropogenic gaps, and 84 percent in patch cuts. Sweetgum, green ash (*Fraxinus pennsylvanica*), and red oaks were common definitive gap fillers.

INTRODUCTION

Within bottomland hardwood forests, low-intensity harvests, e.g., individual selection, favor regeneration and development of shade-tolerant tree species, whereas for regeneration of shade-intolerant tree species, managers have relied on clearcut or shelterwood harvests (Meadows and Stanturf 1997). However, harvest methods that remove most of the forest canopy may be unacceptable when forests are managed for multiple objectives or where maintaining forest integrity is paramount.

Alternative management methods are needed that regenerate shade-intolerant species while retaining forest integrity. This joint objective has been assessed by studies that found regeneration of shade-intolerant species was greater in anthropogenic gaps than in natural gaps (Dickinson and others 2000) and that an intermediate level of harvest may be optimal for regeneration and growth of some shade-intolerant species (Battaglia and Sharitz 2005, Battaglia and others 2004, Collins and Battaglia 2002, Gardiner and others 2004, Paquette and others 2006). These alternative methods have been espoused by the Forest Resource Conservation Working Group of the Lower Mississippi Valley Joint Venture to achieve desired forest conditions for priority wildlife habitat (Wilson and others 2007) that generally are attained through a reduction in canopy cover and basal area. Variable-retention clustered-thinning (VR-CT) harvest is a silvicultural practice that promotes development of desired forest conditions. VR-CT harvests have been undertaken to enhance wildlife habitat and retain biodiversity (Mitchell and Beese 2002), whereas traditional silvicultural thinning has been used to maximize growth and promote the health of residual stems so as to increase future timber volume (Nix 2006). Residual large, dominant stems surrounded by smaller trees and multiple canopy gaps that vary in area are hallmarks of VR-CT harvests. The resultant forest structure is spatially heterogeneous with dense shrubs and herbaceous understory intermixed with clusters of retained trees often with larger diameter trees at their foci (Twedt and Wilson 2007). Within the bottomland forests of the Mississippi Alluvial Valley, this silvicultural system is intended to be economically viable and provide sustainable habitat for priority wildlife, such as Louisiana black bear, migratory birds, and resident game species (Wilson and others 2007).

An additional expectation of VR-CT harvest is regeneration, development, and retention of shade-intolerant tree species, especially within canopy gaps. To further encourage regeneration of shade-intolerant tree species and to support prolonged retention of dense, shrubby understory conditions, up to 10 percent of VR-CT harvest areas may be in patch cut openings of 0.5 to 1.5 ha (Wilson and others 2007). Even so, a concern regarding implementation of VR-CT harvests is that canopy reduction may be insufficient to promote widespread regeneration of shade-intolerant trees [particularly oaks (*Quercus* spp.)], resulting in successional change favoring shade-tolerant tree species. To assess potential promotion of shade-intolerant species to the forest canopy following prescribed VR-CT harvest, we evaluated regeneration and dominance of trees within anthropogenic canopy gaps and compared these with regeneration in natural canopy gaps.

STUDY AREA

Tensas River National Wildlife Refuge (NWR) encompasses >26 000 ha of bottomland hardwood forest in northeast Louisiana. Habitat on the refuge is predominately mature second-growth forest intermixed with recently (<15 years) reforested land that is surrounded by private agriculture. On Tensas River NWR, silviculturally prescribed timber harvests have been used to enhance wildlife habitat within

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bottomland hardwood forests. Although most timber harvests were undertaken before formulation of current management recommendations (Wilson and others 2007), they presaged recommended prescriptions via implementation of VR-CT harvests and incorporation of 0.5 to 1.5 ha patch cut openings within VR-CT harvests.

We surveyed canopy gaps within six separate forest areas (hereafter stands). Each stand was >40 ha and subjected to one of three silvicultural treatments that were equally divided between two forest management units on Tensas River NWR. All forest stands were second growth, having been subjected to historical timber harvest, but prior to the treatments evaluated in this study, stands had not been harvested since refuge establishment in 1980. Stands were predominately sweetgum (*Liquidambar styraciflua*)-willow oak (*Q. phellos*) (Eyre 1980), but the ridge and swale topography supported "ribbons" of other forest types such as sugarberry (*Celtis laevigata*)-American elm (*Ulmus americana*)-green ash (*Fraxinus pennsylvanica*), and overcup oak (*Q. lyrata*)-water hickory (*Carya aquatica*). Upon recommendation of forest managers, three treatments (one per stand) were applied within each management unit: (1) VR-CT, (2) VR-CT with embedded 0.5 to 1.5 ha patch cut openings, and (3) untreated control. Treatment harvests were initiated during summer of 1999 and completed during summer 2000.

METHODS

Forest Sampling

During summer 2004 we characterized species composition, diversity, and basal area of canopy trees within the extant forest on each study stand using a one-basal-area factor metric prism for live stems ≥10 cm diameter at breast height (d.b.h.) at systematically located grid points (250 m apart). We made estimates of angular canopy cover at these points using a spherical densiometer (Fiala and others 2006). Additionally, we recorded the number of species and number of woody stems <10 cm d.b.h. (excluding vines) within 5 m (78.5 m^2) of each point.

Gap Regeneration

During fall 2005, we surveyed canopy gaps within the six study stands. Gaps were surveyed as encountered along line intercept transects (Runkle 1982) that spanned each stand. Gaps were defined as areas lacking forest canopy cover as a result of mortality owing to harvest, fall, or death of one or more canopy trees. Canopy gap area (basic) was estimated for each gap from six laser rangefinder distance measurements, at equally spaced azimuths (60 degrees apart), from the gap "center" to points directly below the edge of canopy vegetation of trees with dominant or codominant crown classes (Runkle 1981). If gap boundaries were excessively irregular, additional distance measurements at intervening azimuths were obtained. We estimated expanded gap area (Runkle 1981) using laser distance measurements from gap center to the base of all dominant or codominant boundary trees—we recorded species and diameter (d.b.h.) of these boundary trees.

We assessed tree regeneration within each gap via a census of all trees with heights >1 m but <10 m. We recorded the species and height (m) of each regenerating tree and categorized its location within the gap as "interior" when within the exposed (basic) gap area and thus potentially a canopy gap filler, or "edge" when within the expanded gap area and thus unlikely to fill the canopy gap due to competition with boundary trees. Up to four definitive gap fillers were identified within each gap as those individuals likely to occupy the canopy void by virtue of their species, stature, and location within the gap. We recorded the species, height, and d.b.h. of definitive gap fillers.

All stems within gaps that were >10 m tall but that did not have a dominant or codominant crown class were identified and recorded as "residual" stems. Residual stems were present as advanced regeneration, or suppressed crown class trees, within the expanded gap at the time of gap formation. Due to their frequent position near boundary canopy trees, most residual stems were unlikely to fill the canopy gap. However, any residual stem deemed likely to fill a canopy gap was recorded as definitive gap filler. The presumed "age" of randomly selected definitive gap fillers within each canopy gap was determined using annual growth rings from basal stem wafers of felled small-diameter saplings or from basal increment bore cores of larger trees (Telewski and Lynch 1991).

For patch cuts, we assessed boundary trees and expanded gap area similarly to other gaps but a complete census of saplings within these large "gaps" was not practical. Therefore, we sampled regeneration using three, randomly located, 0.04-ha (11.3-m radius) circular plots. However, we identified and recorded species and d.b.h. of all definitive gap fillers deemed likely to occupy canopy space within the entire patch cut opening.

Analysis

Gap and expanded gap areas were determined from field measurements by converting polar coordinates to Cartesian coordinates and employing Geographic Information System software (ArcGIS 9.2, Environmental Systems Research Institute, Redlands, CA). We assigned regenerating trees, by species, as either shade-intolerant or shade-tolerant (table 1), and we calculated the proportion of shade-intolerant regeneration within each canopy gap.

We used nonlinear regression (PROC NLIN, SAS Institute Inc., Cary, NC) to assess the relationship between the proportion of shade-intolerant stems within gaps and expanded gap area. We used analysis of variance (PROC GLM) to compare densities, proportions of shade-intolerant stems, and mean heights of regeneration between natural gaps on unharvested stands and anthropogenic gaps that resulted from VR-CT harvest. Finally, we used logistic regression (PROC GENMOD) to assess the relationship between the presence of a definitive gap filling red oak species (*Q. nigra*, *Q. nuttallii*, or *Q. phellos*) and gap area or treatment.

Table 1—Shade tolerance of regenerating hardwood species detected within natural canopy gaps and silviculturally induced (6 years postharvest) canopy gaps on Tensas River National Wildlife Refuge during fall 2004[a]

Shade-intolerant species	Shade-tolerant species
Eastern cottonwood (*Populus deltoides*) (VI)	Red mulberry (*Morus rubra*) (VT)
Black willow (*Salix nigra*) (VI)	Persimmon (*Diospyros virginiana*) (VT)
Honeylocust (*Gleditsia triacanthos*)	Sugarberry (*Celtis laevigata*) (VT)
American sycamore (*Platanus occidentalis*)	Red maple (*Acer rubrum*)
Cherrybark oak (*Quercus pagodifolia*)	Water elm (*Planera aquatica*)
Water oak (*Q. nigra*)	Cedar elm (*Ulmus crassifolia*) (MT)
Nuttall oak (*Q. nuttallii*)	American elm (*U. americana*) (MT)
Willow oak (*Q. phellos*)	Water hickory (*Carya aquatica*) (MT)
Sassafras (*Sassafras albidum*)[b]	Green ash (*Fraxinus pennsylvanica*) (MT)
Black locust (*Robinia pseudoacacia*)[b]	Boxelder (*A. negundo*) (MT)
Sweet pecan (*Carya illinoinensis*) (MI)	Blackgum (*Nyssa sylvatica*) (MT)
Overcup oak (*Q. lyrata*) (MI)	
Bald cypress (*Taxodium distichum*) (MI)	

Shrub and understory species = dogwood (*Cornus* spp.), swampprivet (*Forestiera acuminata*), deciduous holly (*Ilex decidua*), plum (*Prunus* spp.), buttonbush (*Cephalanthus occidentalis*), American beautyberry (*Callicarpa americana*), redbud (*Cercis canadensis*), blue-beech (*Carpinus caroliniana*), Chinese privet (*Ligustrum chinensis*), pawpaw (*Asimina triloba*), hawthorn (*Crataegus* spp.), sumac (*Rhus* spp.), baccharis (*Baccharis halimifolia*), devil's walkingstick (*Aralia spinosa*), and snowbell (*Styrax* spp.).

[a]Shade-tolerance rating from Meadows and Stanturf (1990): VI = very intolerant, VT = very tolerant, MT = moderately tolerant, MI = moderately intolerant.

[b] Shade tolerance rating from Putnam and others (1960).

RESULTS

Extant Forest

Basal area and percent canopy cover were less on VR-CT treated stands than on untreated control stands (table 2). Conversely, small stem density on treated stands was greater than on control stands. Despite the reduction in canopy trees, the overall species composition of the most common canopy trees was similar between treated and untreated stands (table 2). Notably, the ordinal rank of the three species with greatest basal area remained unchanged by treatment. Similarly, the proportion of shade-intolerant species within the canopy was relatively high on treated (71 percent) as well as untreated (60 percent) stands. Shade-intolerant species were the most abundant boundary trees surrounding gaps on both treated (84 percent) and untreated (70 percent) stands. Sweetgum and red oak species were the most common boundary trees surrounding silviculturally created gaps. Reflecting their prevalence within extant forest canopy (table 2), green ash was among those species commonly surrounding natural canopy gaps.

Gap Area

The basic (internal) area of canopy gaps was positively correlated with their expanded area ($r = 0.81$). Mean area of gaps resulting from VR-CT treatment (basic = 142±16 m^2, expanded = 582±54 m^2, $n = 41$) was greater ($F > 9.5$, $P < 0.01$) than mean area of natural gaps (basic = 56±22 m^2, expanded = 262±76 m^2, $n = 21$). Patch cut openings ranged from 0.47 to 1.78 ha.

Gap Regeneration

We recorded 5,188 stems within anthropogenic gaps and 1,854 stems within natural gaps. Additionally, within sample plots in patch cuts we recorded 443 stems. The proportion of woody stems comprised of shrubs or understory tree species (table 1) within natural gaps (15 percent) was similar to that within anthropogenic gaps (14.5 percent). The density of regenerating canopy species varied widely among canopy gaps (range = 295 to 20 976 stems/ha), being greater ($F = 4.3$, $P = 0.04$) in the smaller natural gaps (4415±801) than in the larger silviculturally created gaps (2363±573). These differences were largely due to much greater densities of elms (*Ulmus* spp.), but also more sugarberry, maples (*Acer* spp.), and other shade-tolerant species within natural gaps (fig. 1). Conversely, sweetgum, red oak species, and other shade-intolerant species were more abundant within anthropogenic gaps (fig. 1). We found an overall density of 685 shade-intolerant stems/ha in gaps on treated stands

Table 2—Mean basal area of live trees >10 cm diameter at breast height, density (number of stems <10 cm/ha), and angular canopy cover, as well as the distribution of basal area among the most common species and among shade-tolerance classes on untreated control bottomland hardwood forest stands and after variable-retention clustered-thinning silvicultural treatments on Tensas River National Wildlife Refuge, LA (stands were surveyed during summer 2004, 5 years after treatments were initiated)

Treatment	*n*	Basal area	Canopy	Density	Species (BA)	Shade (BA)
		m^2/ha±SE	*percent*			
VR-CT	4	14.4±0.8	89.3±1.1	3912±488	*Liquidambar styraciflua* = 3.1±0.5	I = 10.3±1.0
					Quercus nigra = 2.6±1.4	MI = 0.5±0.3
					Q. nuttallii = 1.5±0.3	MT = 2.3±0.6
					Q. phellos = 1.5±0.3	T = 1.4±0.4
Control	2	20.9±1.9	98.5±0.9	3079±64	*L. styraciflua* = 3.5±2.5	I = 12.7±0.3
					Q. nigra = 3.3±3.2	MI = 1.5±0.5
					Q. nuttallii = 3.2±0.5	MT = 5.6±0.4
					Fraxinus pennsylvanica = 3.1±0.9	T = 1.2±0.7

BA = basal area; VR-CT = variable-retention clustered-thinning; I = intolerant and very intolerant; MI = moderately intolerant; MT = moderately tolerant; T = tolerant and very tolerant.

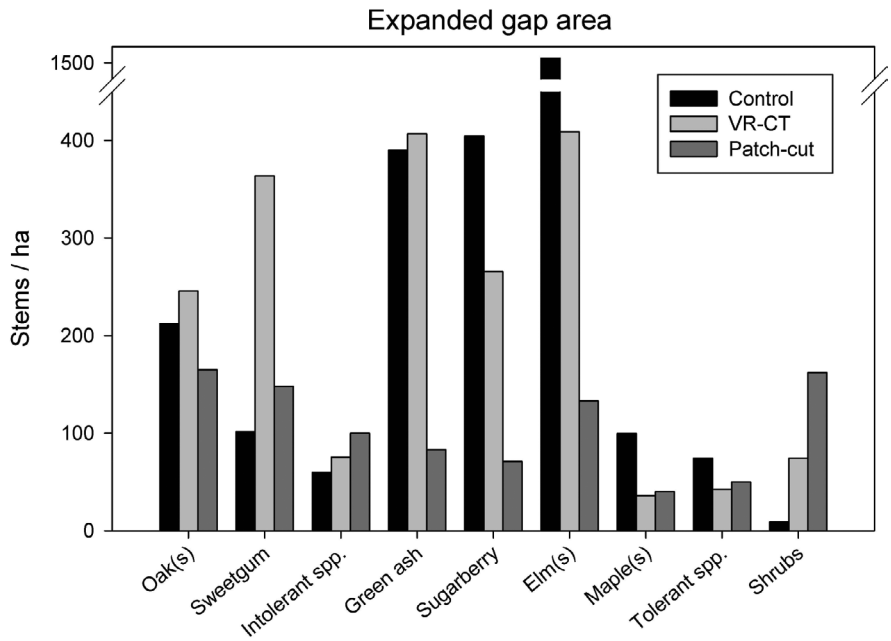

Figure 1—Density of all regenerating stems within expanded canopy gaps of 21 natural gaps (Control), 41 anthropogenic gaps created during variable-retention, clustered thinning silvicultural treatments (VR-CT), and 4 patch cuts (Patch-cut) in bottomland hardwood stands on Tensas River National Wildlife Refuge, northeast Louisiana, 6 years after treatments were initiated.

compared to 374 shade-intolerant stems/ha in natural gaps on control stands.

Because most regenerating stems within natural gaps were of shade-tolerant species, the proportion of stems that were shade-intolerant (0.14±0.04) was significantly less (F = 21.0, P < 0.01) than the proportion of shade-intolerant stems on treated stands (0.36±0.03). Over half (51 percent) of regenerating stems on patch cuts were shade intolerant. Although the proportion of shade-intolerant stems varied widely among canopy gaps (range = 0.0 to 0.78), we found a significant (F = 64.6, P < 0.01) nonlinear relationship with the

expanded gap area (fig. 2). Moreover, shade-intolerant stems tended to be taller (height = 5.5±0.1 m) than their shade-tolerant counterparts (3.9±0.1 m) regardless of treatment (fig. 3).

Differences in density of species between natural and anthropogenic gaps were further amplified by consideration of only interior regenerating stems within the basic canopy area. Densities of shade-intolerant species, especially sweetgum and red oak species, were markedly greater on treated stands than on control stands (fig. 4). Densities of green ash as well as shrub and understory species were also greater within anthropogenic gaps whereas shade-tolerant species, most notably elms, had greater densities within natural canopy gaps (fig. 4).

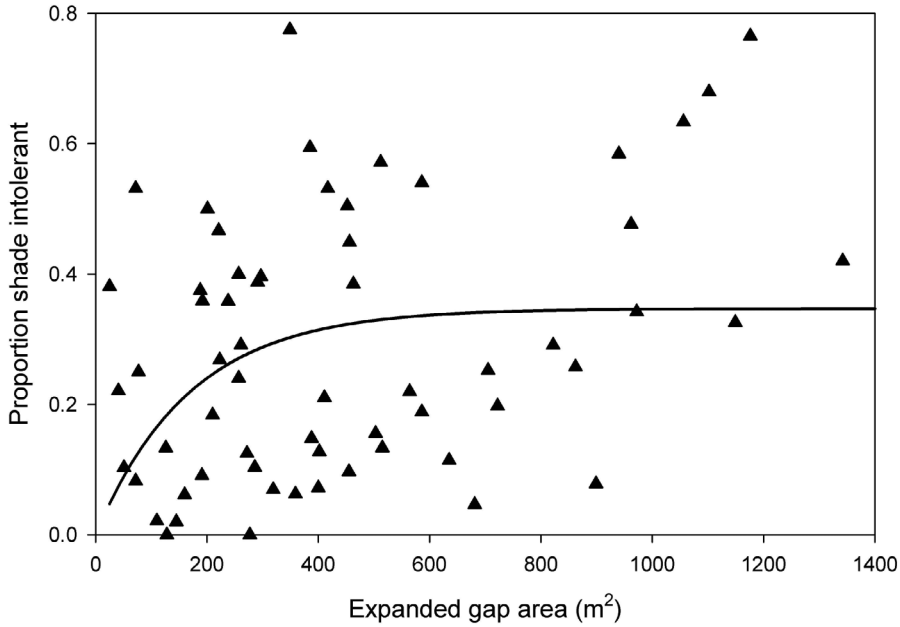

Figure 2—Relationship between the proportion of regenerating stems that were shade intolerant within canopy gaps and expanded gap area for 62 canopy gaps of natural and anthropogenic origin in bottomland hardwood stands on Tensas River National Wildlife Refuge, northeast Louisiana.

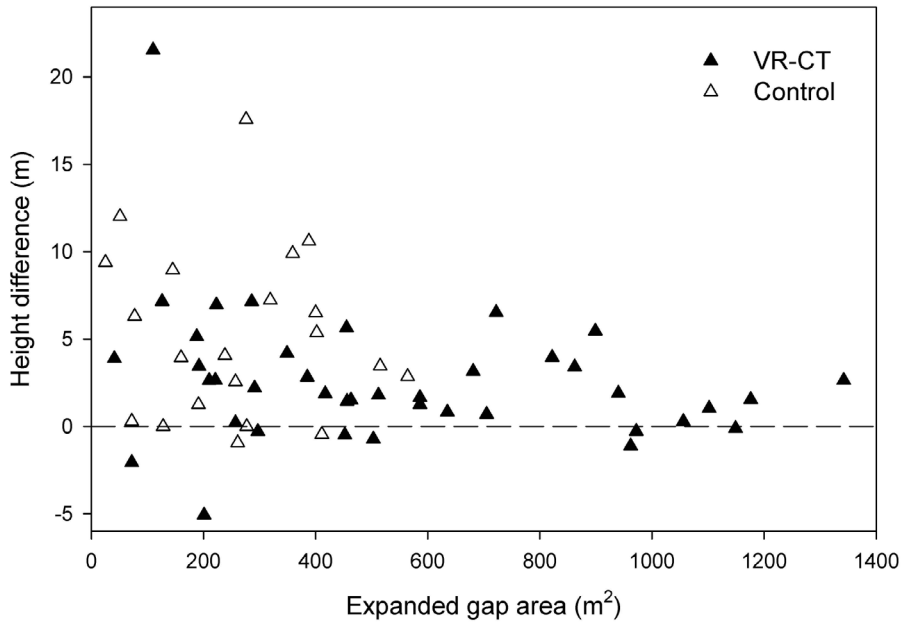

Figure 3—Height difference between shade-intolerant and shade-tolerant regenerating stems within 62 canopy gaps of natural and anthropogenic origin in bottomland hardwood stands on Tensas River National Wildlife Refuge, northeast Louisiana. Height differences >zero indicate shade-intolerant stems were taller than shade-tolerant stems.

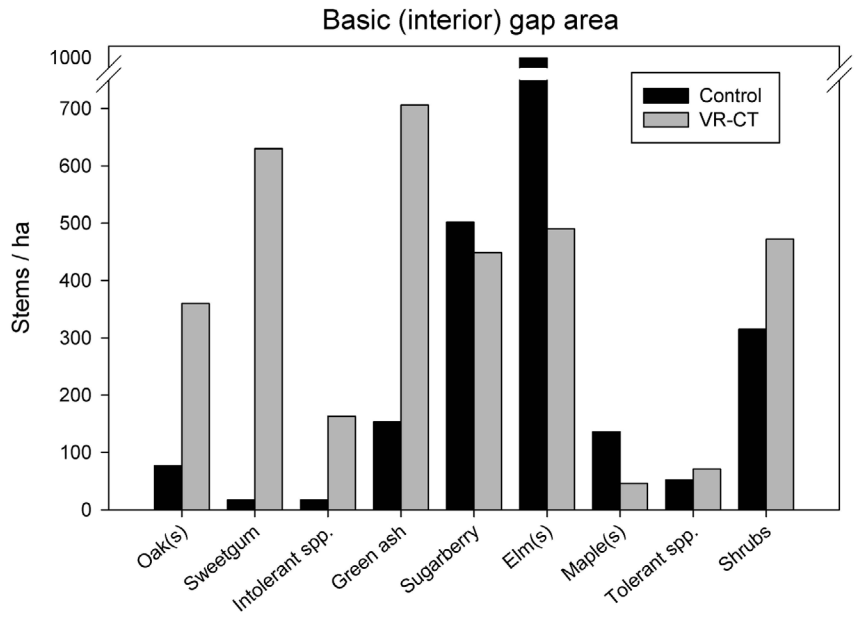

Figure 4—Density of interior regenerating stems within basic canopy gap area (area lacking canopy cover) on 21 natural gaps (Control) and 41 anthropogenic gaps created during variable-retention, clustered thinning silvicultural treatments (VR-CT) in bottomland hardwood stands on Tensas River National Wildlife Refuge, northeast Louisiana, 6 years after treatments were initiated.

Definitive Gap Fillers

We recorded 158 definitive gap fillers within 41 anthropogenic gaps but only 37 within 21 natural gaps. Within patch cuts we indentified 49 definitive gap fillers of which 84 percent were shade intolerant. In contrast, only 54 and 35 percent of definitive gap fillers were shade intolerant within anthropogenic and natural gaps, respectively (fig. 5). The occurrence of a red oak species as a definitive gap filler was independent of gap area ($\chi^2 = 1.06$, $P = 0.30$) and treatment ($\chi^2 = 0.11$, $P = 0.74$).

Definitive gap fillers were markedly taller on untreated stands (11.5±1.1 m) than on treated stands (6.6±0.4 m). Similarly, the age of definitive gap fillers on untreated sites (33.3±4.6 years, $n = 21$) was over twice the age of definitive gap fillers on treated stands (15.2±2.8 years, $n = 46$). Notably, heights of definitive gap fillers within patch cuts (6.1±0.2 m) were similar to their heights within gaps on treated stands, yet their mean age (4.7±0.2 years, $n = 10$) was less than a third the age of definitive gap fillers on treated stands.

DISCUSSION

Most regeneration within canopy gaps was present before gap creation or originated as sprouts from harvested trees. Even so, seeds provided by extant canopy trees, especially boundary trees surrounding canopy gaps may contribute to regeneration within gaps. As such, both treated and untreated stands, with >70 percent of trees surrounding gaps being shade-intolerant species, were well endowed to provision propagules of shade-intolerant trees.

The density of shade-intolerant stems within canopy gaps that resulted from VR-CT harvest (685 stems/ha) was less than the circa 1,000 stems/ha (400 stems per acre) recommended as a desired forest condition (Wilson and others 2007). However, this recommendation was based on stocking levels developed by Hart and others (1995) who incorporated seedling classes <1 m in height (<1 foot and 1 to 3 feet). As we only recorded regeneration ≥1 m in height, we were unable to account for numerous shade-intolerant seedlings within shorter height classes.

We classified green ash among shade-tolerant species but it has been variously reported as moderately shade tolerant (Meadow and Stanturf 1997), intermediate in shade tolerance (Zhao and others 2005), and shade intolerant (King and Antrobus 2005). Green ash was the most common regenerating species within anthropogenic gaps and the third most common regenerating species within natural gaps (fig. 1), as well as being a common canopy tree (table 2). Therefore, had we considered green ash among shade-intolerant species, the proportion of shade-intolerant species would be markedly greater than those we report.

Our data suggest an increased proportion of shade-intolerant regeneration as expanded gap area increased up to circa 500 m^2 (fig. 2). Approximately 35 percent shade-intolerant regeneration occurred within gaps of 500 m^2 with only modest increases thereafter as canopy gap area increased to 2000 m^2 (0.2 ha). Even so, we found the proportion of shade-intolerant regeneration was markedly greater within patch cut openings of 0.5 to 1.8 ha than within canopy gaps. The threshold of

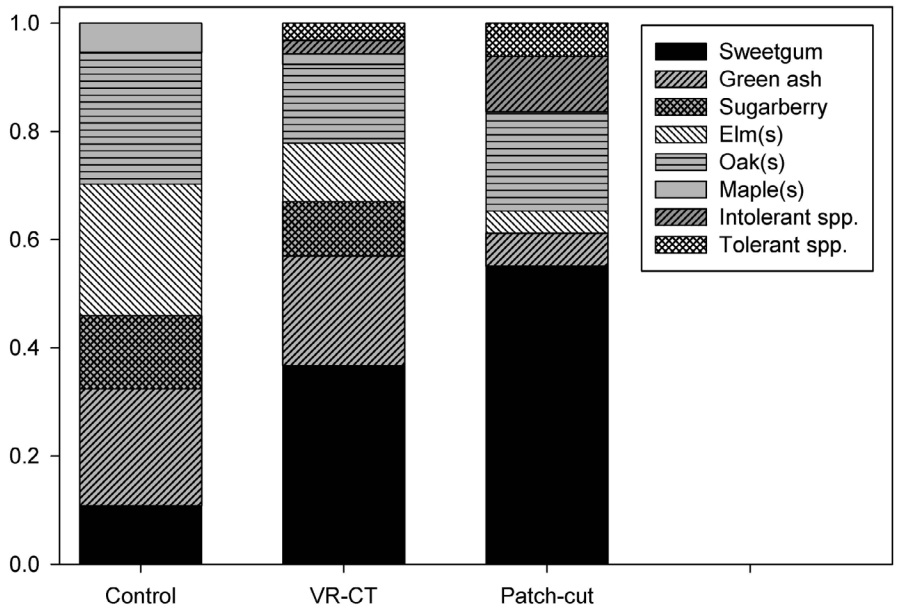

Figure 5—Proportions of definitive gap fillers (individuals deemed likely to ultimately occupy the forest canopy) by species within 21 natural gaps (Control), 41 anthropogenic gaps created during variable-retention, clustered thinning silvicultural treatments (VR-CT), and 4 patch cuts (Patch-cut) in bottomland hardwood stands on Tensas River National Wildlife Refuge, northeast Louisiana.

circa 500 m^2 for increased shade-intolerant regeneration is nearly double the mean expanded gap area (262 m^2) of natural gaps. Indeed, the maximum area of a natural gap in this study was only 564 m^2. Other studies of natural canopy gaps reported similar gap areas—median expanded gap area in Arkansas bottomland forest was 238 m^2 (King and Antrobus 2005), and mean expanded gap area from old-growth mesic forests in Eastern North America was circa 200 m^2 (Runkle 1981). In an east Texas bottomland forest, however, Almquist and others (2002) found mean natural gap area was 657 m^2.

The relatively old age (35±6 years, max = 60 years) of shade-intolerant definitive gap fillers in natural gaps suggests that shade-intolerant species persist for decades in the understory of bottomland stands. Similarly, average age (18±5 years, max = 48 years) of shade-intolerant definitive gap fillers in anthropogenic gaps predated gap creation by a dozen years. Persistence of shade-intolerant species in the understory is consistent with findings in upland hardwood forests of Eastern North America where four species of oak (*Q. alba*, *Q. rubra*, *Q. velutina*, and *Q. prinus*) had understory residence times of 89, 54, 50, and 38 years, respectively, before being released by a canopy disturbance (Rentch and others 2003).

With mean age of definitive gap fillers on patch cuts of <5 years, it appears that advanced regeneration within patch cuts was greatly reduced during treatment, and most regeneration within patch cuts occurred after harvest. This was evidenced by the prominence of stems of stump sprout origin among definitive gap fillers in patch cuts. Even so, height (6.2±0.2 m, n = 41) of shade-intolerant definitive gap fillers within patch cuts nearly equaled the mean height (6.9±0.5 m, n = 85) of considerably older shade-intolerant definitive gap fillers within silviculturally created gaps. The rapid growth of stems originating from stump sprouts is consistent with mean height (circa 4.5 m) of dominant water oak (*Q. nigra*) root sprouts 5 years after a heavy-thinning silvicultural treatment (Gardiner and Helmig 1997). Notably, regeneration of very shade-intolerant species, such as eastern cottonwood (*Populus deltoides*) and black willow (*Salix nigra*) was only recorded in patch cuts.

As shade-intolerant stems comprised only 35 percent of definitive gap fillers in natural gaps and 54 percent in anthropogenic gaps, it is likely that maintaining 67 percent (basal area) shade-intolerant canopy trees, as currently found on unharvested stands, may require use of patch cuts. Moreover, the benefits we observed from inclusion of patch cuts within the matrix of a VR-CT treated stand is not unique to bottomland hardwood forest systems, as Pinard and others (1999) found harvest of "even-aged groups of trees within an uneven-aged matrix" was necessary to achieve their multiple goals of maintaining biodiversity and ecological integrity of the forest while maintaining viable timber harvest in seasonally dry forests of Bolivia.

MANAGEMENT RECOMMENDATIONS

Because oak regeneration is largely dependent on establishing advanced regeneration and creating canopy openings that provide sufficient light to the forest floor

(Clatterbuck and Meadows 1993), we believe that unlike light-thinning or single-tree selection treatments, silvicultural treatments prescribed to promote desired forest conditions will provide for regeneration of shade-intolerant trees (particularly oaks). Indeed, we found that following wildlife-forestry based VR-CT harvest, 54 percent of definitive gap fillers were shade-intolerant species, but the proportion of shade-intolerant species within silviculturally created gaps increased until gap area exceeded 500 m^2. Thus, managers should strive to ensure prescribed treatments create canopy gaps with diameter >25 m. Even so, including patch cut areas of 0.5 to 1.5 ha within VR-CT harvested stands is likely required to achieve >60 percent shade-intolerant regeneration and these patch cut openings may be required to perpetuate very shade-intolerant species within the forest canopy.

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LITERATURE CITED

Almquist, B.E.; Jack, S.B.; Messina, M.G. 2002. Variation of the treefall gap regime in a bottomland hardwood forest: relationship with microtopography. Forest Ecology and Management. 157: 155–163.

Battaglia, L.L.; Collins, B.S.; Weisenhorn, P.B. 2004. *Quercus michauxii* regeneration in and around aging canopy gaps. Canadian Journal of Forest Research. 34: 1359–1364.

Battaglia, L.L.; Sharitz, R.R. 2005. Effects of natural disturbance on bottomland hardwood regeneration. In: Fredrickson, L.H.; King, S.L.; Kaminski, R.M., eds. Ecology and management of bottomland hardwood systems: the state of our understanding. Spec. Publ. 10. Puxico, MO: University of Missouri-Columbia, Gaylord Memorial Laboratory: 121–136.

Clatterbuck, W.K.; Meadows, J.S. 1993. Regenerating oaks in the bottomlands. In: Loftis, D.; 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: 184–195.

Collins, B.S.; Battaglia, L.L. 2002. Microenvironmental heterogeneity and *Quercus michauxii* regeneration in experimental gaps. Forest Ecology and Management. 155: 279–290.

Dickinson, M.B.; Whigham, D.F.; Hermann, S.M. 2000. Tree regeneration in felling and natural treefall disturbances in a semideciduous tropical forest in Mexico. Forest Ecology and Management. 134: 137–151.

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

Fiala, A.C.S.; Garman, S.L.; Gray, A.N. 2006. Comparison of five canopy cover estimation techniques in the western Oregon Cascades. Forest Ecology and Management. 232: 188–197.

Gardiner, E.S.; Helmig, L.M. 1997. Development of water oak stump sprouts under a partial overstory. New Forests. 14: 55–62.

Gardiner, E.S.; Stanturf, J.A.; Schweitzer, C.J. 2004. An afforestation system for restoring bottomland hardwood forests: biomass accumulation of Nuttall oak seedlings interplanted beneath eastern cottonwood. Restoration Ecology. 12: 525–532.

Hart, C.P.; Hodges, J.D.; Belli, K.; Stanturf, J.A. 1995. Evaluating potential oak and ash regeneration on minor bottoms in the Southeast. In: Edwards, M.B., ed. Proceedings of 8th biennial southern silviculture research conference. Gen. Tech. Rep. SRS-1. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 434–442.

King, S.L.; Antrobus, T.J. 2005. Relationships between gap makers and gap fillers in an Arkansas floodplain forest. Journal of Vegetation Science. 16: 471–480.

Meadows, J.S.; Stanturf, J.A. 1997. Silvicultural systems for southern bottomland hardwood forests. Forest Ecology and Management. 90: 127–140.

Mitchell, S.J.; Beese, W.J. 2002. The retention system: reconciling variable retention with the principles of silvicultural systems. Forestry Chronicle. 78: 397–403.

Nix, L.E. 2006. Crop trees and quality in bottomland hardwoods ten years after an early thinning in a young sprout-origin stand in South Carolina. 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: 620–622.

Paquette, A.; Bouchard, A.; Cogliastro, A. 2006. Survival and growth of under-planted trees: a meta-analysis across four biomes. Ecological Applications. 16: 1575–1589.

Pinard, M.A.; Putz, F.E.; Rumiz, D. [and others]. 1999. Ecological characterization of tree species for guiding forest management decisions in seasonally dry forests in Lomerio, Bolivia. Forest Ecology and Management. 113: 201–213.

Putnam, J.A.; Furnival, G.M.; McKnight, J.S. 1960. Management and inventory of southern hardwoods. Agric. Handb. 181. Washington, DC: U.S. Department of Agriculture Forest Service. http://www.srs.fs.usda.gov/pubs/28908. [Date accessed: October 1, 2008].

Rentch, J.S.; Fajvan, M.A.; Hicks, R.R. 2003. Oak establishment and canopy accession strategies in five old-growth stands in the central hardwood forest region. Forest Ecology and Management. 184: 285–297.

Runkle, J.R. 1981. Gap regeneration in some old-growth forests of the Eastern United States. Ecology. 62: 1041–1051.

Runkle, J.R. 1982. Guidelines and sample protocol for sampling forest gaps. Gen. Tech. Rep. PNW-GTR-283. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 44 p.

Telewski, F.W.; Lynch, A.M. 1991. Measuring growth and development of stems. In: Lassoie, J.P.; Hinckley, T.M., eds. Techniques and approaches in forest tree ecophysiology. Boca Raton, FL: CRC Press: 503–555.

Twedt, D.J.; Wilson, R.R. 2007. Management of bottomland forests for birds. In: Shupe, T.F., ed. Proceedings of Louisiana natural resources symposium. Baton Rouge, LA: Louisiana State University AgCenter: 49–64.

Wilson, R.R.; Ribbeck, K.; King, S.L.; Twedt, D.J., eds. 2007. Restoration, management and monitoring of forest resources in the Mississippi Alluvial Valley: recommendations for enhancing wildlife habitat. Vicksburg, MS: LMVJV Forest Resource Conservation Working Group, Lower Mississippi Valley Joint Venture. 88 p. http://www.lmvjv.org/bookshelf.htm. [Date accessed: October 1, 2008].

Zhao, D.; Borders, B.; Wilson, M. 2005. A density-dependent matrix model for bottomland hardwood stands in the Lower Mississippi Alluvial Valley. Ecological Modeling. 184: 381–395.

RED MAPLE (*ACER RUBRUM*) RESPONSE TO PRESCRIBED BURNING ON THE WILLIAM B. BANKHEAD NATIONAL FOREST, ALABAMA

Stacy L. Clark and Callie Jo Schweitzer[1]

Abstract—Prescribed burning is used as a management tool on national forests in the Southeastern United States to maintain oak (*Quercus* spp.) -dominated forest or woodland habitat. Few studies have examined response to burning at the stand, plot, and tree level. We documented red maple (*Acer rubrum*) response to dormant-season prescribed burns and the relationship with fire characteristics. At the stand level, large seedling density increased in burned stands by 930 trees per acre, significantly higher than the increase of 10 trees per acre in control stands. Prescribed burning had no effect on red maple sapling mortality, sprouting, or diameter growth. Maximum temperature did not predict red maple density changes or sprouting occurrence at the plot level. At the tree level, char height was a significant predictor of sprouting, and saplings could produce up to 97 sprouts following burning. We conclude that burning as applied in this study was not an effective tool to change species composition and stand structure in this ecosystem.

INTRODUCTION

The concern that hazardous fuel buildup and overstocked stands will increase the risk of unnatural and catastrophic wildfires has led to implementation of large-scale Federal programs to reduce fuels, which include prescribed burning (Veblen 2003). The U.S. Forest Service National Forest System (NFS) implemented prescribed burning treatments on approximately 1.3 million acres per year from 2006 to 2008 for the purposes of hazardous fuel reduction through the auspices of the Healthy Forest Initiative and the National Fire Plan (Healthy Forests and Rangelands 2009a). While much of the media and research attention related to fuels reduction has been concentrated in the Western United States, the NFS conducted 71 percent of prescribed burn acres in 2008 on lands in the Southern Region (Healthy Forests and Rangelands 2009b). National Forests in Alabama burned approximately 99,000 acres in 2008, more than the national forests in Colorado and California combined.

In addition to fuels reduction, prescribed burning is often included in NFS Land Resource and Management Plans to achieve landscape restoration goals to improve wildlife habitat, to conduct site preparation for regeneration, and to restore native species. However, adequate site-specific research examining if fuels reduction treatments can adequately achieve ecological restoration goals is lacking (Veblen 2003), particularly in the Southern United States.

For many years, researchers and natural resource managers have promoted fire as a tool that will restore the oak (*Quercus* spp.) component to upland hardwood forests (Moser and others 2006, Nyland and others 1983, Van Lear and Waldrop 1989). However, empirical evidence is often conflicting due to differences in study design among experiments, and long-term studies are rare. Results from several studies suggest that prescribed burning alone, without additional disturbances involving overstory tree harvesting, will not significantly promote oak regeneration over the short term (Blankenship and Arthur 2006, Hutchinson and others 2005, Signell and others 2005). These studies indicate prescribed burning is an inefficient tool for altering species composition in the understory; however, site-specific research needs to be conducted before broad recommendations can be made regarding the applicability of prescribed burning as a management tool to enhance the oak component in upland hardwood forests. Studies that examine large-scale vegetation responses, e.g., determining differences in vegetation response between silvicultural treatments at the stand level, are useful for determining overall effectiveness of management tools but do not help explain the mechanisms controlling the vegetation response.

A large-scale replicated study was initiated in 2005 on the William B. Bankhead National Forest (BNF) in Alabama to examine the effects of thinning and prescribed burning on fuel loading and residual oak tree health (Schweitzer and Wang, this proceedings). The overall goals of the prescribed burning treatments are to improve forest health and restore native upland hardwood forests, particularly oak, by reducing fuel loading and oak competitors in the midstory and understory. The BNF conducted the prescribed burning for this study under the auspices of their Land and Resource Management Plan (U.S. Department of Agriculture Forest Service 2003). The objective of this study was to examine if prescribed burning treatments were effective at reducing competition from a primary oak competitor, red maple (*Acer rubrum*). We examined response of red maple at the stand level, and we measured fire and tree characteristics at a smaller scale to help determine the mechanisms controlling response of this species to dormant-season prescribed burns.

METHODS

Study Area

We implemented this study within the BNF, in the Southern Cumberland Plateau Physiographic Province (Smalley 1979).

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Average annual precipitation is approximately 57 inches per year, average annual temperature is 60 °F, and elevation averages 870 feet. Soils are Typic Hapludults, consisting of moderately deep, well-drained soils that are weathered from sandstone with a thin strata of siltstone or shale. Stands are on gently sloping to moderately steep side slopes and ridgetops with slopes ranging from zero to 30 percent.

Stands are 15- to 45-year-old loblolly pine (*Pinus taeda*) plantations that have not been entered for harvesting since establishment and have not been burned during the last 10 years. Basal area averages approximately 100 square feet per acre consisting of 75 percent pine and 25 percent hardwood (Schweitzer and Wang, this proceedings). The BNF's Land and Resource Management Plan (U.S. Department of Agriculture Forest Service 2003) details a proposed action for restoring existing pine plantations to oak-dominated forests using treatments that include thinning and prescribed burning.

Data Collection and Experimental Design

This study was conducted within a larger silvicultural research study (Schweitzer and Wang, this proceedings). The original study was designed as a two-factor, randomized, complete block with sampling to examine vegetation response to different silvicultural treatments. Experimental units were 20- to 30-acre forest stands. Treatments included three levels of thinning (no thinning control, residual basal area of 50 square feet per acre, and residual basal area of 75 square feet per acre); three levels of prescribed burn treatments (no burning control, infrequent burn every 8 to 10 years, and frequent burn every 3 to 5 years); and all combinations of burning and thinning treatments. The resulting overall study constitutes nine treatments, each replicated four times, and blocked by year of implementation (block 1 implemented in 2006; blocks 2 and 3 implemented in 2007; block 4 implemented in 2008).

For the purposes of this study, we only examined prescribed burn and control treatments, and we did not examine treatments that involved thinning. The first implementation of the two prescribed burn treatments was conducted during the same year within each of the four blocks; therefore, we had eight replications of the prescribed burn treatment and four replications of the control treatment.

Prescribed burns were conducted between December 20 and March 4 using a combination of flanking, backing, and strip-head fires with hand ignition. Fire behavior was recorded by ocular estimation by BNF personnel during prescribed burns, flame lengths were estimated to average 1.2 feet (range 0.6 to 1.8 feet), and rate of spread was estimated to average 1.4 chains per hour (range 1.1 to 2.0 chains per hour) across the eight prescribed burns. Fires were characterized by BNF personnel as cool burns with low risk of escaping across fire line boundaries. Objectives of burning included fuels reduction, enhancement of wildlife habitat, and enhancement of hardwood natural regeneration (U.S. Department of Agriculture Forest Service 2003).

Five measurement plots consisting of two concentrically nested circular plots were systematically arranged within each stand relatively equidistant to each other. We used a 0.025-acre plot to record large saplings (1.5 to 5.5 inches d.b.h.), and we tallied understory tree regeneration on a 0.01-acre plot by enumerating vegetation into three size classes—small seedlings (<1 foot height), large seedlings (≥1 foot, <4 feet height), and small saplings (≥4 feet height, <1.5 inch d.b.h.). We measured and recorded vegetation in plots prior to and just after each treatment implementation.

To more accurately determine the mechanisms controlling red maple response to fire, we documented fire characteristics within stands. We installed fire-monitoring devices at each of the five vegetation plots within the frequent burn regime treatment units (stands to be burned every 3 to 5 years). We could not measure fire behavior in the infrequent burn regime treatments due to lack of resources. In block 1, maximum fire temperature was measured using aluminum tags painted with five levels of temperature-sensitive paints (Tempil®) that melt at 175, 200, 300, 400, and 575 °F. We positioned aluminum tags horizontally on an aluminum pin flag 10 inches above the ground so that each of the five paints had equal opportunity to melt during the prescribed burn. One tag was placed at plot center, and 4 tags were placed 12 feet in each cardinal direction from plot center, resulting in 25 tags per burn unit in block 1. Temperature-sensitive paints are a relatively inexpensive and efficient method for monitoring fire behavior when resources for research monitoring are low. For blocks 2, 3 and 4, additional resources became available and we used type-K thermocouple probes attached to HOBO dataloggers to record fire behavior data. The probe tips record temperature, and they were placed in an upward position 10 inches above the ground. A 6-foot cable extending from the probe to the datalogger was buried in a 3-inch trench of mineral soil. Prior to transportation to the field, the datalogger was covered in an antistatic bag to prevent a buildup of static electricity (Iverson and others 2004). In the field, the datalogger was placed inside a PVC casing and buried in a 6-inch hole. We installed dataloggers and probes on the morning of the burns, and we programmed them to record temperature every 2 seconds. Care was taken to minimize disturbance to fuels around the probe tip, and the litter layer was repositioned over the trench of the buried cable. We positioned three probes 12 feet from plot center in the north, east, and west cardinal direction. We recorded a plot as burned if one of the paint tags or one of the dataloggers obtained a minimum temperature of at least 175 °F or 90 °F, respectively. If none of the paint tags or dataloggers reached this minimum temperature, we labeled the plot as unburned.

We examined response to fire at the tree-level scale by recording maximum fire temperature and red maple responses for 42 red maple saplings (1.5 to 5.4 in d.b.h.) in the block 4 frequent burn regime unit. We placed two aluminum tags painted with temperature sensitive paint (as described above) just above the litter layer on opposite sides of the base of each tree. Prior to the burns, we measured d.b.h. and tallied number of sprouts (>1 foot in height, <1.5 inch d.b.h.) for each red maple sapling. We also recorded

volume (height, width, and depth to nearest 0.1 inch) of existing cambium wounds on the tree. The day following the burns, we measured maximum height of the charred surface, i.e., char height, on each red maple sapling to the nearest 0.1 foot and recorded the minimum temperate recorded from the paint tags. A tree was labeled as unburned if none of the levels of temperature-sensitive paints on the two tags melted. In late May following the prescribed burn, we documented tree mortality, and if the tree was alive, we documented dieback to the main stem and counted number of sprouts.

Data Analysis

We used the statistical program, SAS (SAS Institute Inc. 2000), to conduct all statistical analyses, and we chose an error level of 0.05 to indicate significance in all tests. We examined fire effects at the stand level by using PROC MIXED to conduct an analysis of variance (ANOVA) (table 1). The ANOVA was conducted on pretreatment measurements to determine differences between burn and control treatments in density of red maple regeneration in each size class prior to treatment implementation. We then conducted the ANOVA to determine if there were differences among burn and control treatments in stem density changes following treatment implementation. Stem density change was calculated as the difference in number of stems from before to after treatment implementation. For the ANOVA, we included data from all vegetation plots in the burn treatments, whether they were recorded as burned or unburned, because we wanted to determine overall effect of management practices currently used by the BNF on the density of red maple at the stand level. Normality and equality of variance assumptions for residuals were checked and dependent variables were transformed when needed.

We examined the mechanisms controlling red maple response to fire by measuring fire behavior at the plot and tree level in all of the frequent burn regime treatment units. We conducted a linear regression (PROC REG) to determine if mean maximum fire temperature recorded in the plot could be used to predict red maple density changes of regeneration (<1.5 inches d.b.h.), mortality of red maple large saplings (1.5 to 5.5 inches d.b.h.), and occurrence of large sapling sprouting. We removed plots that were recorded as unburned from the linear regression analysis because it is impossible to determine the relationship between vegetation response to fire characteristics if no burning occurred on the plot itself. Temperatures recorded in block 1 using temperature-sensitive paints were adjusted to be comparable with the temperatures recording using probes in blocks 2 through 4. We adjusted the temperature by multiplying by 0.815 and adding 49.5 °F (Iverson and others 2004). This adjustment is necessary because temperature-sensitive paints record lower temperatures than probes.

We examined fire effects at the tree level by conducting a multiple linear regression using data from the 42 large saplings tallied in block 4. The regression was conducted using a stepwise elimination with mean maximum fire temperature, char height, wound volume, and d.b.h. as independent variables and differences in number of sprouts before and following prescribed fire as the dependent variable. Logistic regression (PROC LOGISTIC) was used to determine if mean maximum fire temperature, char height, wound volume, and d.b.h. could be used to explain the probability of sapling mortality and occurrence of crown dieback. For the multiple regression and logistic regression analysis, we removed trees that were labeled as unburned, as described above.

RESULTS

The prescribed burns in the frequent burn regime treatment units were relatively similar in mean maximum temperature across blocks but were highly variable within stands. The overall mean of fire temperature was 181 °F (table 2), but plots ranged in mean maximum temperature from 98 °F to 308 °F across all blocks. Block 1 in the frequent burn regime treatment units had the most consistent burn pattern with all plots burning (n = 5), and this block had higher maximum fire temperatures than all other blocks. Block 2 had the lowest

Table 1—Analysis of variance used to determine differences in changes in density pre- and posttreatment between prescribed burn and control treatment units

Source	Effect	Degrees of freedom	Error term
Treatment	Fixed	1	Block*treatment
Block	Random	3	
Block*treatment	Random	3	
Plot (block*treatment)	Random	9	

Table 2—Fire mean maximum temperature, associated number of plots (n), standard error, and associated range for each block and across blocks. Fire temperature was recorded using temperature-sensitive paints in block 1 and was recorded using thermocouple probes attached to dataloggers in blocks 2, 3, and 4.

Block	n	Mean maximum temperature (°F)	Standard error	Range (°F)
1[a]	5	269	12	240–308
2	2	110	12	98–121
3	4	188	21	130–228
4	4	157	5	143–164
Overall	4	181	34	109–269

[a] Temperature values for block 1 have been adjusted as described in the methods.

recorded temperatures and the fewest number of plots that burned (n = 2).

We did not detect any pretreatment differences in red maple density between control and prescribed burn treatments for any size class ($P > 0.70$). Prior to treatment implementation, stands had an average of 1,840, 380, 280, and 208 stems per acre of small seedlings, large seedlings, small saplings, and large saplings, respectively. Small seedling density changes following treatment implementation did not differ between control and burn treatment ($P = 0.91$). Large seedlings increased 930 trees per acre after prescribed burning and increased 10 trees per acre in the control treatment (fig. 1), and the difference between treatments was significant ($P = 0.04$). Prescribed burn units had a reduction of small saplings by 120 trees per acre following treatment implementation, while the control treatments had an increase of 95 small saplings per acre; however, the difference between treatments was not statistically significant ($P = 0.11$). Large sapling density changes were not different between treatments ($P = 0.23$).

Maximum fire temperature was not a good predictor of variation in density changes for red maple regeneration in any size class ($R^2 < 0.22$, $P > 0.28$), and it did not explain variation in large sapling mortality and sprouting occurrence ($R^2 = 0.06$ and 0.0005, respectively; $P = 0.41$ and 0.94, respectively). Char height was selected as the only significant variable that explained variation in number of new sprouts from saplings after prescribed burning ($R^2 = 0.24$, $P = 0.01$). The model predicts that number of new sprouts following burning was positively related to char height (inches).

$$\hat{y} = -7.3 + 20.2b \quad (1)$$

Of the 42 saplings that we tallied, 20 percent had crown dieback and 43 percent produced new sprouts following the burn. Average number of new sprouts following prescribed burning was 10 per tree, but could be as high as 97. Using logistic regression, sapling d.b.h., wound volume, char height, and maximum fire temperature around the base of the tree could not be used to explain the probability of sapling mortality ($P > 0.45$) or occurrence of crown dieback ($P > 0.34$).

DISCUSSION

Fire is an ecological process that has helped maintain oak-dominated forests of the Eastern United States for millennia (Delcourt and Delcourt 1997, Van Lear and Waldrop 1989). Determining the effects of fire on species composition and stand structure through empirical research is difficult due to the variable nature of ecosystem function and processes, and the difficulty in replicating fire as a testable treatment effect. Our research indicates that the first application of dormant-season prescribed burning was highly variable in intensity, as measured from fire temperature, and did not have the desired effect of decreasing red maple density to favor recruitment of hard-mast species like oak. Prescribed burn units showed an increase of 930 trees per acre in the abundance of red maple large seedlings, likely due to basal sprouting following

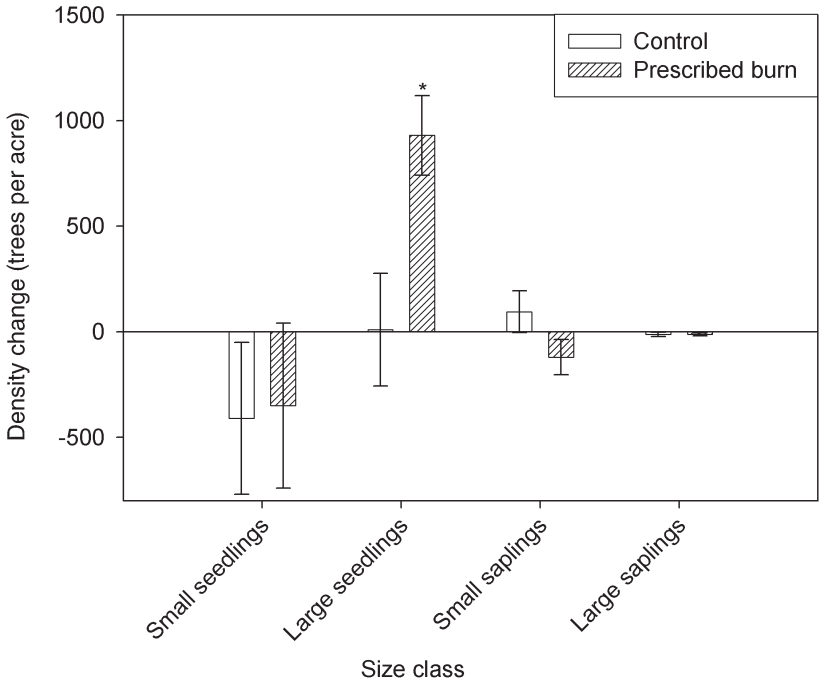

Figure 1—Stem density changes in red maple (*Acer rubrum*) seedlings and saplings following control and prescribed burn treatments. Error bars represent standard error of the mean. Asterisk indicates significant differences between control and prescribed burn treatments for respective size class.

topkill of the main stem. This increase in stem density of large seedlings represents an important biological change that may affect the recruitment of advanced oak regeneration (Lorimer and others 1994). While small saplings did decrease by 120 trees per acre in prescribed burn units, the difference between burn and control units in stem density changes was not statistically different. The reduction in small saplings in burn units may be significant to managers, however, as this represented a 46-percent density decrease from pretreatment levels.

We predict that additional prescribed burn treatments alone will not reduce red maple density of large seedlings in the short term. Blankenship and Arthur (2006) found that three prescribed burns increased the density of red maple stems in a similar size class to the large seedling size class in this study. Decreases in red maple seedling densities due to fire will probably be temporary as sprouts mature and new seedlings establish after the disturbance (Hutchinson and others 2005, Waldrop and others 2008).

Examination of smaller scale mechanisms that might affect red maple response was difficult due to the variable nature of the fire and response of red maple at the plot and tree level. The use of maximum fire temperature measured from temperature-sensitive paints placed at the base of maple saplings was not useful for predicting individual tree response to prescribed burning. If we repeat this study, we will measure temperature at positions farther up the bole of the tree to increase correlations between tree response and fire temperature. Char height was the only variable that was a significant predictor of tree response to fire, and this variable was correlated positively to number of sprouts produced following a prescribed burn. Char height has been shown to be a good indicator of mortality in boreal species (Hely and others 2003) and in slash pine (*Pinus elliottii*) (Menges and Deyrup 2001), but has not been adequately tested for red maple.

CONCLUSIONS

At a stand-level scale, prescribed burning did not have a significant effect on reducing red maple density. In fact, density of large seedlings significantly increased compared to the control treatment. Small sapling density decreased by 46 percent in burn treatments. Although this reduction was not statistically different than the control, it may hold some value for management practitioners in these ecosystems. Without additional treatment, however, basal sprouts from these topkilled small saplings will eventually contribute to overall recruitment of red maple into the midstory. The failure of correlation between vegetation response and fire temperature at the plot and tree level was due to the high variability in fire behavior and in the spatial patchiness of the burns. We predict that repeated burning alone will not be an effective tool to reduce the abundance of red maple due to the ability of this species to sprout prolifically following topkill of the main stem.

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LITERATURE CITED

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.

Delcourt, H.R.; Delcourt, P.A. 1997. Prehistoric human use of fire on Southern Appalachian landscapes. Conservation Biology. 11: 1010–1014.

Healthy Forests and Rangelands. 2009a. Healthy forests report, FY 2008 accomplishments. http://www.forestsandrangelands.gov/resources/reports/documents/healthyforests/2008/healthy_forests_report_fy2008.pdf. [Date accessed: January 15].

Healthy Forests and Rangelands. 2009b. National fire plan. http://www.forestsandrangelands.gov/NFP/index.html. [Date accessed: January 15].

Hely, C.; Flannigan, M.; Bergeron, Y. 2003. Modeling tree mortality following wildfire in the Southeastern Canadian mixed-wood boreal forest. Forest Science. 49: 566–576.

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.; 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: 1–12.

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.

Menges, E.S.; Deyrup, M.A. 2001. Postfire survival in south Florida slash pine: interacting effects of fire intensity, fire season, vegetation, burn size, and bark beetles. International Journal of Wildland Fire. 10: 53–63.

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.

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.

SAS Institute Inc. 2000. SAS. Version 8.01. Cary, NC: SAS Institute Inc.

Schweitzer, C.J.; Wang, Y. X. Overstory tree status following thinning and burning treatments in mixed pine-hardwood stands on the William B. Bankhead National Forest, Alabama. This proceedings.

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.

U.S. Department of Agriculture 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. 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.

Veblen, T.T. 2003. Key issues in fire regime research for fuels management and ecological restoration. In: Omi, P.N.; Joyce, L.A., tech. eds. Fire, fuels treatments and ecological restoration: conference proceedings. Proceedings RMRS-P-29. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 259–276.

Waldrop, T.A.; Yaussy, D.A.; Phillips, R.J. [and others]. 2008. Fuel reduction treatment affect stand structure of hardwood forests in western North Carolina and southern Ohio, USA. Forest Ecology and Management. 225: 3117–3129.

PREDICTING THE REGENERATION OF APPALACHIAN HARDWOODS: ADAPTING THE REGEN MODEL FOR THE APPALACHIAN PLATEAU

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Abstract—The difficulty of achieving reliable oak (*Quercus* spp.) regeneration is well documented. Application of silvicultural techniques to facilitate oak regeneration largely depends on current regeneration potential. A computer model to assess regeneration potential based on existing advanced reproduction in Appalachian hardwoods was developed by David Loftis of the U.S. Forest Service, Bent Creek Experimental Forest. REGEN is a competition-based, expert system which predicts dominant and codominant species composition at the onset of stem exclusion. A knowledge base containing competitive rankings for each species and size combination of advance reproduction is used to make predictions. REGEN was initially developed for hardwood forests in the Blue Ridge Physiographic Province of the Southern Appalachians and is only applicable for predicting regeneration following a heavy disturbance, such as a clearcut. We have developed preliminary REGEN knowledge bases for hardwood forests in the Appalachian Plateau Province.

INTRODUCTION

Hardwood Regeneration

Successfully regenerating oak (*Quercus* spp.) is difficult in eastern hardwood forests when using traditional regeneration systems (Loftis and McGee 1993). Oak is often displaced by more aggressive shade-intolerant species following clearcutting or by more shade-tolerant species in partial harvesting systems. Site quality has been identified as a primary driver of upland oak regeneration potential (Weitzman and Trimble 1957). The influence of site quality on regeneration potential is perhaps most evident in the Southern Appalachians, resulting in increased site specificity in techniques prescribed to foster oak regeneration (Loftis 1990b). The most promising techniques are modifications of the shelterwood system (Brose and Van Lear 1998, Loftis 1990a). Efforts to foster development of large oak advance reproduction often require an extended regeneration period, perhaps up to 20 years or longer (Sander 1972). Complex relationships between species and site further hinder implementation of landscape level oak regeneration improvement efforts, and the extended planning horizon required by those techniques can only increase management uncertainty. Therefore, identification of individual stands where oak regeneration will be inadequate and their potential for improvement should assist land managers in allocating limited resources. Estimates of regeneration potential may be achieved preharvest using regeneration prediction models. Several regeneration models have been developed for eastern hardwoods (Gould and others 2006, 2007; Loftis 1990b; Sander and others 1984).

Regeneration Models

Regeneration models are generally categorized as either qualitative or quantitative (Rogers and Johnson 1998). Qualitative models are often presented as decision charts or guidelines and usually offer interpretation and prescriptions. A qualitative model has been published for the central Appalachians (Steiner and others 2008). Quantitative models are often computer based and have historically provided estimates of regeneration potential for a single species or perhaps a species group. Quantitative models are often presented as equations and are typically more limited in adaptability. Regional quantitative models such as the ACORn model for the Ozarks are available in some areas (Dey 1991). Currently a multispecies regional quantitative regeneration prediction model for the Southern Appalachians has not been published. Interest in such a tool has led to the development of the REGEN model by Research Forester David Loftis of the U.S. Forest Service, Bent Creek Experimental Forest (Loftis 1989).

REGEN Model

The REGEN model is an expert system designed to predict dominant and codominant species composition following heavy disturbance. The expert user assigns competitive rankings based on species and size of advance reproduction in a predefined scenario. A key feature of REGEN is the categorization of advance reproduction species-size combinations, which are separated as—germinant, small (<2 feet tall), medium (2 to 4 feet tall), large (4 feet and taller), and potential stump sprouts (taller than 4 feet and >2 inches d.b.h.). Each species-size combination is given a ranking, thereby increasing the specificity of the model. Rankings range from 1 to 20 decreasing in competiveness. Given that sources of successful regeneration are often not present as advance reproduction, REGEN allows for the probability of establishment for these unobserved sources, as well as vegetative forms of reproduction, to be included as a constant or logistic parameter. Stems are rewarded based on relative rankings of the population of propagules in each sample plot. The stochastic feature of the model permits numerous runs of the input data thus allowing summary statistics to be included with the results.

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A REGEN knowledge base (RKB) is used to process input data collected from field sample plots. Competition is simulated at the plot level and "winning" stems per plot are later scaled to stems per acre. REGEN populates the predicted plot by adding up to six "winning" stems based on competitive ability per 0.01-acre plot. If one stump sprout is chosen, the number of "winners" per plot is reduced to four; if two or more sprouts are chosen there can be only three "winners" per plot. These rules are intended to compensate for the increased space requirements of stump sprouts as opposed to seed origin regeneration.

RKB's are modular, allowing for the expansion of REGEN to different scenarios by creating an RKB unique for that scenario. The original RKB was developed for the Blue Ridge Physiographic Province of the Southern Appalachians. The objectives of this study were to evaluate the adaptability of the REGEN model framework to the Appalachian Plateau, create RKBs for that region, and field test the predictions from REGEN against data collected across the Appalachian Plateau.

METHODS

Recognizing the impact site quality can have on the species composition of Appalachian hardwoods, three preliminary RKBs were developed for the Appalachian Plateau in an attempt to capture species variability resulting from site differences. The delineation for application of the three RKBs is based on upland oak site index (base age 50) along the following breaks: site index <65 feet = low quality RKB, site index 65 to 75 feet = medium quality RKB, site index >75 feet = high quality RKB. Rankings were subjective but were based on general silvics and trends reported in available literature as much as possible. Yellow-poplar (*Liriodendron tulipifera*) and black cherry (*Prunus serotina*) were the highest ranked species in all size categories and site qualities, with sweet birch (*Betula lenta*) and red maple (*Acer rubrum*) also being ranked highly. Oaks generally increased in rank with size and decreasing site quality. The more mesic species generally decreased in rank as site quality decreased, while the more xeric species increased in rank. Stump sprouts were ranked as the most competitive source of regeneration when applicable, and rankings generally decreased with smaller size classes.

The RKBs were field tested using a paired stand approach. This approach required that sample sites have a mature hardwood stand relatively free from disturbance for the past 25 years immediately adjacent to a regenerating stand which was harvested via clearcut at least 5 years earlier. Efforts were made to ensure similar site characteristics existed on each component of the paired sample sites; however, a wide range of site quality was desired across sample sites. A total of 41 paired sample sites were located throughout West Virginia on the eastern edge of the Appalachian Plateau in Fayette, Greenbrier, Nicholas, Tucker, and Webster Counties. Of these 41 stands, 7 had a majority of plots in the high-site quality group (site index >75 feet), 23 stands were in the medium-quality group (site index 65 to 75 feet), and 11 stands were in the low-quality group (site index <65 feet).

Sample plots were located using a systematic random grid. A total of up to twenty 0.025-acre regeneration plots were installed at a density of 1 plot per acre on each mature stand to assess regeneration potential. Stand size was variable. At each sample plot in the mature stand, advance reproduction was tallied by species and REGEN height class. Harvested stands were sampled using 0.001-acre plots unless stand development had progressed such that the plots were frequently not populated, in which case sample plots were reestablished as 0.025-acre plots. Sampling was conducted at a density of one plot per acre on a systematic random grid. Stand sizes were variable. Stems were tallied by species, stem origin (seed or sprout), and crown class. At each sample plot on both components of the paired stand slope, aspect, and landscape position were measured to obtain an estimate of site quality using the forest site quality index (Meiners and others 1984).

Although advance reproduction was sampled on 0.025-acre plots, an error was discovered with the scaling algorithm that REGEN uses to adapt data from various plot sizes to the native 0.01-acre plot size. For the purpose of this paper, field data were manually scaled premodel by multiplying each individual advance reproduction propagule by 2.5 and rounded up due to the selection process of the model.

RESULTS AND DISCUSSION

Due to the high floristic diversity found in the Southern Appalachians, species are combined into 10 groups based on either frequency or similarity. The 10 groups created are as follows: black cherry, red maple, sugar maple (*A. saccharum*), sweet birch, yellow-poplar, oaks, mixed mesophytic, subcanopy, pioneer, and miscellaneous. The oaks group consisted of black oak (*Q. velutina*), chestnut oak (*Q. prinus*), northern red oak (*Q. rubra*), scarlet oak (*Q. coccinea*), and white oak (*Q. alba*). Mixed mesophytic species included basswood (*Tilia americana*), American beech (*Fagus grandifolia*), buckeye (*Aesculus octandra*), cucumbertree (*Magnolia acuminata*), Fraser magnolia (*M. fraseri*), and yellow birch (*Betula alleghaniensis*). Subcanopy species found were American chestnut (*Castanea dentata*), American holly (*Ilex opaca*), blackgum (*Nyssa sylvatica*), flowering dogwood (*Cornus florida*), eastern hophornbeam (*Ostrya virginiana*), sassafras (*Sassafras albidum*), serviceberry (*Amelanchier arborea*), sourwood (*Oxydendron arboreum*), and striped maple (*Acer pensylvanicum*). Pioneer species included black locust (*Robinia pseudoacacia*), pin cherry (*Prunus pensylvanica*), and American sycamore (*Platanus occidentalis*). The miscellaneous species group included those that were too few to justify a single group and did not fit well into any other established groups. Species in the miscellaneous group include the ashes (*Fraxinus* spp.), hickories (*Carya* spp.), eastern hemlock (*Tsuga canadensis*), red spruce (*Picea rubens*), and eastern white pine (*Pinus strobus*). Predictions from REGEN were summarized postcomputation into the same 10 species groups.

Model results using advance reproduction in the mature stand as input were compared to the species composition of the harvested component under the assumption that the mature

stand would regenerate very similarly to the developing stand if it were to be harvested today. Comparisons of the model results using the three preliminary RKBs with field-collected data are presented. The average species composition for the seven high-quality harvested stands and the predicted species composition from the high-quality RKB are presented in table 1. The high-quality RKB had the largest discrepancies with the observed composition. The most notable difference was for sugar maple which was overestimated considerably; other species in this RKB were reasonably predicted. The average species composition of the 23 medium-quality harvested stands and the predicted species composition from the medium-quality RKB are displayed in table 2. For the 11 low-quality stands, the average species composition is shown in table 3 along with the predicted species composition from the low-quality RKB. In the medium- and low-quality RKBs, red maple was predicted in higher proportions than what was actually observed, while sugar maple was again overestimated in the low-quality RKB. Given the frequency in which maple is referred to as a potential benefactor to decreasing oak regeneration in the literature, both red and sugar maples were given highly competitive rankings in all three RKBs. Sugar maple was ranked most strongly in the high-quality RKB, and red maple was ranked highly throughout all three RKBs. Results from the field testing suggest that the maples are not as competitive as currently ranked.

Yellow-poplar was observed in similar proportions across all three site quality groups and was predicted similarly in all three RKBs. This is expected as the ranking for yellow-poplar is identical for all three RKBs. Sweet birch was more prevalent in the medium- and high-site quality stands but was underrepresented in the REGEN predictions. Sweet birch, a shade-intolerant species, was very sparse as advance reproduction, and although it was ranked highly, the absence of advance reproduction suggests that the stochastic addition should be increased. Sweet birch is known to have the ability to regenerate very aggressively; however, dominance is often short-lived, and the higher levels found in the medium- and

Table 2—Mean dominant and codominant species composition of 23 medium-site quality stands

Species group	Species composition	
	Observed	Predicted
	---------- *percent* ----------	
Black cherry	12.41	13.02
Miscellaneous	1.73	0.98
Mixed mesophytic	16.53	22.30
Oaks	9.73	3.35
Pioneer	9.02	5.03
Red maple	15.96	38.54
Subcanopy	3.57	4.11
Sugar maple	1.44	2.01
Sweet birch	15.08	3.44
Yellow-poplar	14.53	7.22
Total	100.00	100.00

Table 1—Mean dominant and codominant species composition of seven high-site quality stands

Species group	Species composition	
	Observed	Predicted
	---------- *percent* ----------	
Black cherry	27.54	22.44
Miscellaneous	0.00	1.56
Mixed mesophytic	17.85	9.12
Oaks	2.12	0.08
Pioneer	7.55	2.85
Red maple	12.33	13.50
Subcanopy	8.54	3.43
Sugar maple	0.53	33.48
Sweet birch	14.90	4.14
Yellow-poplar	8.64	9.40
Total	100.00	100.00

Table 3—Mean dominant and codominant species composition of 11 low-site quality stands

Species group	Species composition	
	Observed	Predicted
	---------- *percent* ----------	
Black cherry	4.73	0.81
Miscellaneous	3.12	3.86
Mixed mesophytic	3.77	7.43
Oaks	17.27	4.85
Pioneer	17.53	4.15
Red maple	24.38	36.95
Subcanopy	9.73	15.93
Sugar maple	0.62	7.68
Sweet birch	5.46	6.67
Yellow-poplar	13.39	11.67
Total	100.00	100.00

high-quality groups are expected to gradually decrease as the stand continues through stem exclusion. The pioneer species are expected to perform similarly. This anticipated trend somewhat discourages higher ranking for these species even though the predictions are likely inaccurate at the targeted onset of stem exclusion. Oaks were underestimated by all RKBs but did follow published trends of increasing postharvest competitiveness with decreasing site quality. Other species groups were considered reasonably accurate across the three RKBs for preliminary results.

CONCLUSION

Literature suggests that species composition of postharvest regeneration may react differently somewhere near the aforementioned site quality delineations for the three RKBs in the Southern Appalachians (Smith 1994). For regional modeling purposes it may only be realistic to approximate general breaks in site quality, perhaps within an accuracy of 5 to 10 feet of site index, due to natural uncertainty of forest systems and unreliable methods of estimating site quality. Further investigation of the ranges at which site quality can be assumed to impact regeneration similarly is warranted.

An evaluation of the feasibility of adapting the REGEN model to the Appalachian Plateau based on the results of the preliminary knowledge base tests indicates potential, given refinements and amendments, for REGEN to be developed into a useful tool for the region. Considering that the preliminary rankings are largely subjective, as there are no published numerical competitive rankings compatible with this type of system for the region, future work to amend the now existing knowledge bases for the Appalachian Plateau seems a worthwhile endeavor.

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LITERATURE CITED

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.

Dey, D.C. 1991. A comprehensive ozark regenerator. Columbia, MO: University of Missouri-Columbia. [Number of pages unknown]. Ph.D. dissertation.

Gould, P.J.; Fei, S.; Steiner, K.C. 2007. Modeling sprout-origin oak regeneration in the central Appalachians. Canadian Journal of Forest Research. 37: 170–177.

Gould, P.J.; Steiner, K.C.; McDill, M.E.; Finley, J.C. 2006. Modeling seed-origin regeneration in the central Appalachians. Canadian Journal of Forest Research. 36: 833–844.

Loftis, D.L. 1989. Species composition of regeneration after clearcutting Southern Appalachian hardwoods. In: Miller, J.H., ed. Proceedings: 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. 1990a. A shelterwood method for regenerating red oak in the Southern Appalachians. Forest Science. 36(4): 917–929.

Loftis, D.L. 1990b. Predicting post-harvest performance of advanced red oak reproduction in the Southern Appalachians. Forest Science. 36(4): 908–916.

Loftis, D.L.; McGee, C.E., eds. 1993. Oak regeneration: serious problems, practical recommendations. Proceedings of a symposium. Gen. Tech. Rep. SE-84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 319 p.

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.

Rogers, R.; Johnson, P.S. 1998. Approaches to modeling natural regeneration in oak-dominated forests. Forest Ecology and Management. 106: 45–54.

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. 9 p.

Sander, I.L.; Johnson, P.S.; Rogers, R. 1984. Evaluating oak advance reproduction in the Missouri Ozarks. Res. Pap. NC-251. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 16 p.

Smith, D.W. 1994. The Southern Appalachian hardwood region. In: Barrett, J.W., ed. Regional silviculture of the United States. New York: John Wiley: 173–225.

Steiner, K.C.; Finley, J.C.; Gould, P.J. [and others] 2008. Oak regeneration guidelines for the central Appalachians. Northern Journal of Applied Forestry. 25(1): 5–16.

Weitzman, S.; Trimble, G.R. 1957. Some natural factors that govern the management of oaks. Stn. Pap. 88. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 44 p.

MODELING THE LONG-TERM EFFECTS OF OAK SHELTERWOOD REGENERATION TREATMENTS ON SPECIES DIVERSITY AND OAK ABUNDANCE IN SOUTHERN APPALACHIAN FORESTS OF NORTH CAROLINA

Tara L. Keyser and Chad E. Keyser[1]

Abstract—In April 2008, the Upland Hardwoods Ecology and Management Research Work Unit of the U.S. Forest Service, Southern Research Station began a long-term cooperative study to describe forest ecosystem response to three oak (*Quercus* spp.) shelterwood regeneration treatments in the central hardwoods region of the United States. Pretreatment inventory data from 10 mature, mixed-oak forest stands on North Carolina Wildlife Resources Commission Game Lands were input into the Forest Vegetation Simulator (FVS) to analyze the long-term forest ecosystem response to the following oak shelterwood regeneration treatments: (1) shelterwood followed by prescribed fire and overstory removal, (2) shelterwood via herbicide followed by overstory removal, (3) repeated prescribed fire followed by overstory removal, and (4) control. In this study, FVS growth forecasts were used to analyze alternative oak shelterwood regeneration treatment effects on species diversity and oak abundance over the next 50 years.

INTRODUCTION

Historically, disturbance events such as low-intensity surface fires (both natural and human-caused), timber harvesting, grazing, loss of American chestnut (*Castanea dentata*), and land clearing for agriculture promoted overstory and understory conditions conducive to the establishment, development, and recruitment of midtolerant oak species (*Quercus* spp.) across the upland hardwood forest ecosystem (Abrams 1992, Lorimer 1993). As a result of repeated disturbance events, oak gained dominance in forest stands at the expense of competitors such as shade-tolerant red maple (*Acer rubrum*) and shade-intolerant yellow-poplar (*Liriodendron tulipifera*). Abundant evidence indicates that changing disturbance regimes are promoting the conversion of mixed-oak forests to forests dominated by shade-tolerant species such as red maple (e.g., Orwing and Abrams 1994), or by shade-intolerant species such as yellow-poplar (e.g., Beck and Hooper 1986, Rodewald 2003). For example, Aldrich and others (2005) observed an increase in the abundance of shade-tolerant sugar maple (*A. saccharum*) from ~1 percent of total stand density in 1926 to ~25 percent in 1992 in an old-growth forest in Indiana that had no active management since 1917. Similarly, in the same forest, Spetich and Parker (1998) reported a decrease in the biomass of small-diameter (10 to 25 cm) oak trees from 14 percent in 1926 to only 1 percent in 1992. The authors note that during the same time period, biomass of sugar maple in the same size class increased from 12 to 43 percent.

Ecologically, mixed-oak forests are among the most productive terrestrial ecosystems (Whittaker and Likens 1975), store substantial amounts of carbon (Greco and Baldocchi 1996), and are considerable sources of wildlife habitat, food resources, and overall biodiversity. The decline of oak as an overstory tree species, coupled with regeneration failures (Aldrich and others 2005), could have cascading ecological effects. Silvicultural practices are utilized to achieve numerous resource management objectives; including the creation and maintenance of wildlife habitat, habitat restoration, timber production, and maintenance of landscape-level biodiversity. Within the upland hardwood ecosystem, numerous silviculture prescriptions have been developed to specifically regenerate oak in mid- to high-quality stands (e.g., Brose and others 1999, Loftis 1990). In this paper, we examine efficacy of three recommended, but not widely tested, oak shelterwood regeneration treatments by modeling their impact on oak abundance and overall species composition over a 50-year period using available regeneration and growth-and-yield models.

METHODS

Study Site

The data from this study were collected from the North Carolina Wildlife Resources Commission, Cold Mountain Game Lands (CMGL) and serve as pretreatment data for the U.S. Forest Service's Research Work Unit 4157 (RWU-4157) Regional Oak Study. The CMGL, which lie within the Blue Ridge Physiographic Province of the Southern Appalachian Mountains, consists of mature, second-growth upland mixed-oak forests. Terrain is mountainous with steep slopes. Elevations range from 980 to 1200 m. Oak SI_{50} in the 10 units ranged from 19 to 31 m. Oaks [red (*Q. rubra*), white (*Q. alba*), chestnut (*Q. prinus*), and black (*Q. velutina*)], hickory (*Carya* spp.), black cherry (*Prunus serotina*), and yellow-poplar are the dominant overstory trees. Species composition in the midstory consists primarily of shade-tolerant species including sourwood (*Oxydendrum arboreum*), blackgum (*Nyssa sylvatica*), silverbell (*Halesia tetraptera*), flowering dogwood (*Cornus florida*), and red maple.

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Shelterwood Regeneration Treatments

Oak shelterwood regeneration treatments were designed to test the effectiveness of silvicultural methods currently suggested to regenerate oak in upland hardwood forests in the Eastern United States. Treatments were: (1) shelterwood/burn (SWB), (2) oak shelterwood via herbicide (OSW), (3) prescribed fire (RXF), and (4) control (CON). The prescription for the SWB treatment followed the guidelines outlined in Brose and others (1999). The initial step was to perform an establishment cut leaving approximately 7 m^2/ha of residual basal area (BA). Three years following the establishment cut, a prescribed fire was performed. Ten years after the establishment cut, the residual overstory trees were removed down to a target BA of ~2 to 3.5 m^2/ha. The prescription for the OSW treatment followed the guidelines presented in Loftis (1990). The initial step, e.g., establishment cut, was the removal of the competing midstory [trees ≥5.0 cm and <25.0 cm diameter at breast height (d.b.h.)] using herbicide. The goal of the herbicide treatment was to reduce total BA by 25 to 30 percent. Ten years following the herbicide treatment, the residual overstory trees were removed down to a target BA of ~2 to 3.5 m^2/ha. In the RXF treatment, prescribed fire was performed three times (return interval of 4 years). Ten years following the initial prescribed fire, the residual overstory trees were removed down to a target BA of ~2 to 3.5 m^2/ha. No silvicultural manipulation occurred in the CON treatment throughout the duration of this study.

Experimental Design and Data Collection

During the summer of 2008, we established twenty 5-ha treatment units on the CMGL. Treatments were randomly assigned to each treatment unit. Treatment units contained mature (>70 years old), fully-stocked, closed-canopied stands in which oak comprised at least 10 percent of the overstory tree (≥25.0 cm d.b.h.) BA, contained approximately 2 m^2/ha of BA beneath the main canopy, and contained at least ~1,000 oak seedlings/ha. Within each treatment unit six 0.05-ha (12.6 m radius) permanent vegetation plots were systematically located along a grid.

Within each 0.05-ha vegetation plot, all overstory trees (≥25.0 cm d.b.h.) were tagged and measured. Midstory trees (≥5.0 cm and <25.0 cm d.b.h.) were tagged and measured within a 0.01-ha (5.6 m radius) subplot nested within each of the larger vegetation plots. For each tagged tree, the data recorded included species and d.b.h. to the nearest 0.1 cm. Tree regeneration was sampled using two 0.004-ha (3.6 m radius) circular regeneration subplots originating 8 m from plot center at bearings of 45° and 225°. All arborescent regeneration sources were tallied by species in four height/diameter classes: (1) <0.6 m, (2) 0.6 to <1.2 m, (3) ≥1.2 but <3.8 cm, and (4) ≥3.8 cm.

Modeling

Ten of the twenty treatment units were selected for modeling the effects of the aforementioned treatments. To address the variability associated with the inventory sample of these units, a bootstrapping technique described by Hummel and Cunningham (2006) was implemented through the FVSBoot computer program (Gregg and Hummel 2002) to resample the plots within each unit. Each unit was resampled 500 times, which resulted in the creation of 5,000 boostrapped stands plus the original 10 stands, for a total of 5,010 stands.

Each stand was modeled under each treatment alternative (SWB, OSW, RXF, and CON) using the Southern Variant of the Forest Vegetation Simulator (FVS-SN) (Crookston and Dixon 2005, version 6.21, revision date 9-19-08). FVS-SN is the U.S. Forest Service nationally supported growth-and-yield modeling system that is used to forecast stand development with and without management or other disturbance events. Variants of FVS-SN have been calibrated to most forest types in the United States and can be downloaded through the U.S. Forest Service, Forest Management Service Center Web site (www.fs.fed.us/fmsc/). Each FVS-SN variant is a distance-independent, individual tree growth model that has the capability of including silvicultural, fire, and insect and disease impacts on forest stands. Users are able to track outputs of individual tree characteristics and stand characteristics such as density, volume, wildlife habitat, fire and health related indices, and carbon stocks.

Two control variables were adjusted in the FVS-SN model runs. First, default site index for each stand was estimated using methodology developed by McNab and Loftis (this proceedings) and entered into the FVS-SN forecasts. Secondly, Reineke's Stand Density Index maximums were reset by species based on FVS-SN estimates of forest-type maximums and an article by Schnur (1937).

A known constraint of the FVS-SN is the regeneration model. In FVS-SN terminology, the regeneration model is a partial establishment model, meaning only sprouts are estimated when a tree is cut or killed by fire. All other regeneration must be entered by the user. By default, you get two sprouts per tree cut or killed by fire; however, FVS-SN allows the user to turn off this automatic sprouting, modify the sprouting routine, or enter regeneration by species and size directly at any time during the growth forecast. Given the importance of regeneration estimates in forecasting stand development under the proposed treatments, we deemed the partial establishment model's sprouting routine insufficient[2] and decided to enter regeneration estimates based on literature and a local regeneration model. We provided regeneration estimates for two conditions: (1) following prescribed fire and (2) following substantial overstory removal. We entered regeneration estimates following prescribed fire based on Alexander and others (2008).

Regeneration estimates following substantial overstory removal were provided by the REGEN (version 1.0.2) model (Loftis 1988, Schweitzer and others 2004). REGEN is a model used to predict species composition of regeneration after

[2] Personal communication. David Loftis.

overstory removal in mixed-species stands. The model is driven by a pretreatment inventory of all existing regeneration sources enumerated by species and size class. Based on probabilities, the model adds sprouts as well as seedlings and root suckers to the regeneration plots. Seedlings and root suckers are only added for species that are capable of either producing root suckers or establishing new seedlings shortly after substantial disturbance. In the Southern Appalachians, these species include American beech (*Fagus grandifolia*), black locust (*Robinia pseudoacacia*), and sassafras (*Sassafras albidum*) for root suckers and sweet birch (*Betula lenta*), yellow birch (*B. alleghaniensis*), yellow-poplar, black cherry, and yellow pines (*Pinus virginiana*, *P. echinata*) for new seedlings. REGEN picks the dominant/codominant trees on each regeneration plot at crown closure based on a ranking of the postharvest performance of different regeneration sources which include new seedlings, various sizes of advance reproduction, and stump sprouts. Probabilities and rankings used in this study were provided by David Loftis (U.S. Forest Service, Southern Research Station, Bent Creek Experimental Forest). FVS-SN preharvest tree lists were entered into REGEN in the year of the simulated overstory removal. REGEN then predicted the type and amount of regeneration to input back into the FVS-SN growth forecasts. Details of the exact timing of treatments in FVS-SN and interactions between FVS-SN and REGEN are provided in table 1. Outputs tracked by FVS-SN over the growth period of 50 years included density by species group (table 2) within each stand. Metrics reported throughout the paper were calculated on the bootstrapped sample (n = 5010).

RESULTS

Pretreatment Data

The stands used in this study were dominated by oak and hickory species with 48 percent of the BA of dominant/codominant trees being in the oak-hickory (OH) species group, 35 percent in the intolerants (IN) species group, and 11 percent in the other (OT) species group (fig. 1A). Species within the midstory (MS) group contributed only 7 percent of the dominant/codominant BA. Within the OH species group, oak accounted for an average of 10.8 m^2/ha of BA or 71 percent of the dominant and codominant BA within the OH species group. Similarly, 51, 28, 9, and 12 percent of the dominant/codominant stems/ha were within the OH, IN, MS, and OT species groups, respectively (fig. 1B). Within the OH species group, oak accounted for an average of 86 stems/ha or 63 percent of the dominant/codominant BA within the OH species group. Prior to the implementation of treatments in FVS-SN or REGEN, the stand BA, density, and quadratic mean diameter averaged (±1 standard deviation) 37.1 (8.0) m^2/ha, 635 (139) trees/ha, and 27.5 (4.3) cm.

Posttreatment Species Diversity and Oak Abundance

Treatments targeted towards oak regeneration had a substantial impact on overall species composition compared to pretreatment species composition. After 50 years, the distribution of the BA by species was most similar to pretreatment levels in the OSW treatment with 41, 40, 1, and 17 percent of the BA in dominant/codominant stems in the OH, IN, MS, and OT species groups, respectively (fig. 2A). The SWB treatment resulted in the greatest departure from pretreatment species composition. A substantial proportion of the BA in dominant/codominant stems, especially within the OH group, was the result of the ~2 to 3.5 m^2/ha of residual overstory (with preference given to oak species) left during the overstory removal that FVS-SN simulated in 2018. When examining the number of dominant and codominant stems/ha resulting from treatments, the departure from pretreatment conditions was more visible. However, the OSW treatment, again, best approximated the distribution of dominant and codominant stems/ha prior to treatment with 22, 48, 2, and 32 percent of the dominant/codominant stems/ha in the OH, IN, MS, and OT species groups followed by the RXF and SWB treatments (fig. 2B).

Despite substantial variability in model outcomes (fig. 3), little difference was observed in the 50-year model projections regarding the regeneration of oak species into dominant/codominant canopy positions. The BA of dominant/codominant oak stems in the OSW, RXF, and SWB treatments was between of 3.5 and 4.5 m^2/ha (fig. 3A). The similarity in oak BA among the treatments, again, is likely due to the ~2 to 3.5 m^2/ha of residual overstory left during the overstory removal that occurred in 2018. Despite substantial variability, after 50 years, the number of dominant/codominant oak stems/ha regenerated by the OSW and RXF treatments were most similar, averaging ~81 stems/ha whereas the SWB treatment resulted in an average of 24 dominant/codominant oak stems/ha. By the end of the modeling forecast, oaks accounted for 99 percent of the dominant/codominant stems within the OH species group in the OSW and RXF treatments and 92 percent in the SWB treatment.

DISCUSSION

Results from the model forecasts show that the efficacy of these treatments in regenerating oak and maintaining species diversity in upland forests of the Southern Appalachians varied by treatment. Oak shelterwood regeneration methods using the treatments modeled in this study have been suggested to regenerate oak: however, the success of these methods likely varies in response to the ecological differences, e.g., soils, climate, and species composition, that exist across the central hardwoods region (CHR) of the United States. For example, using a method similar to the OSW treatment, Loftis (1990) reported significantly higher oak regeneration on sites where BA was reduced by 30 percent the decade prior to overstory removal in highly productive sites in northern Georgia (oak SI_{50}). However, this method, which was highly successful in northern Georgia, has not been tested across the upland hardwood ecosystem. Similarly, repeated prescribed fire, which has been suggested as critical to successful oak regeneration (e.g., Abrams 1992), has shown promise as a method to develop large advance oak regeneration on dry to intermediate sites in southern Ohio (Iverson and others 2008) but has not been tested across the broad range of ecosystems that occur in the CHR. Before fire is used to regenerate oaks in the Southern Appalachians, more studies examining the effects of repeated burning in

Table 1—Description of modeling timeframe and linkages between FVS-SN and REGEN

Treatment and year/years	Description
OSW	
2008	- Read stand data into FVS-SN and create tree list file
2009	- Simulate establishment cut via herbicide treatment in FVS-SN
2018	- FVS-SN passes tree list to REGEN - REGEN estimates regeneration response to overwood removal - REGEN passes regeneration composition back to FVS-SN - FVS-SN simulates overwood removal and inputs regeneration
2018–58	- FVS-SN continues to grow stand
SWB	
2008	- Read stand data into FVS-SN and create tree list file
2009	- FVS-SN passes tree list to REGEN - REGEN estimates regeneration response to establishment cut - REGEN passes regeneration composition back to FVS-SN - FVS-SN simulates shelterwood harvest and inputs regeneration
2012	- FVS-SN simulates prescribed fire/mortality and sprouting from fire
2018	- FVS-SN simulates overwood removal
2018–58	- FVS-SN continues to grow stand
RXF	
2008	- Read stand data into FVS-SN and create tree list file
2009	- FVS-SN simulates prescribed fire/mortality and sprouting from fire
2012	- FVS-SN simulates prescribed fire/mortality and sprouting from fire
2016	- FVS-SN simulates prescribed fire/mortality and sprouting from fire
2018	- FVS-SN passes tree list to REGEN - REGEN estimates regeneration response to overwood removal - REGEN passes regeneration composition back to FVS-SN - FVS-SN simulates overwood removal and inputs regeneration
2018–58	- FVS-SN continues to grow stand
CON	
2008	- Read stand data into FVS-SN and create tree list file
2008–58	- FVS-SN continues to grow stand

FVS-SN = Southern Variant of the Forest Vegetation Simulator; REGEN = a model used to predict species composition of regeneration after overstory removal in mixed-species stands; OSW = oak shelterwood via herbicide treatment; SWB = shelterwood/burn treatment; RXF = prescribed fire treatment; CON = control treatment.

various forest types throughout the Southern Appalachians are required. Surprisingly, the model projections did not show the SWB treatment to be as successful as the RXF and OSW treatments at regenerating oak despite a study by Brose and others (1999) where a technique similar to the SWB treatment simulated in this paper showed the treatment to be highly successful at regenerating oak on intermediate-quality (oak SI_{50} 23 m) sites in northern Virginia. The stands used in this study were, on average, higher quality sites than used by Brose and others (1999) with oak SI_{50} averaging 25 m creating a more competitive environment where oak may not outcompete faster growing species in the INT species group like black

Table 2—Designation of species groups

Species group	Species included
Oak-hickory	Oaks (*Quercus* spp.) and hickories (*Carya* spp.)
Intolerants	Yellow-poplar (*Liriodendron tulipifera*), black cherry (*Prunus serotina*), sweet birch (*Betula lenta*), black locust (*Robinia pseudoacadia*)
Midstory	Flowering dogwood (*Cornus florida*), red maple (*Acer rubrum*), blackgum (*Nyssa sylvatica*), sourwood (*Oxydendrum arboreum*), striped maple (*Acer pensylvanicum*), silverbell (*Halesia tetraptera*)
Other	Sugar maple (*Acer saccharum*), black walnut (*Juglans nigra*), magnolia (*Magnolia* spp.), others

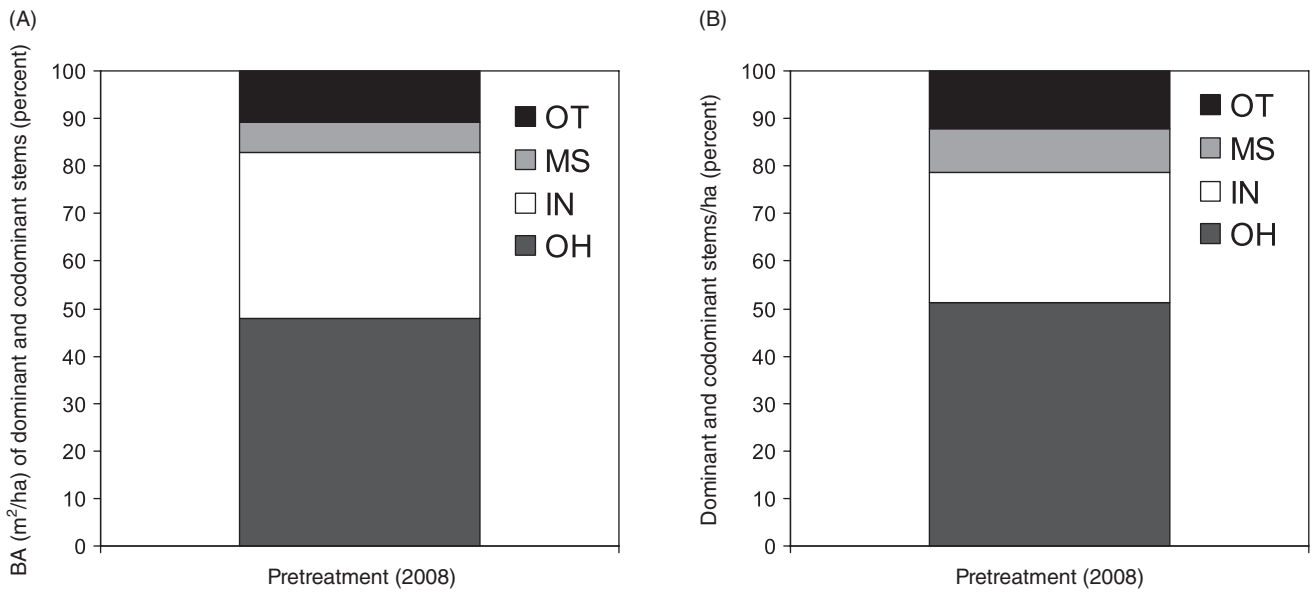

Figure 1—Pretreatment distribution of (A) basal area (BA) (m^2/ha) and (B) stems/ha within the oak-hickory, intolerant, midstory, and other species group within the 5,010 stands.

cherry and yellow-poplar, emphasizing, again, the need to test these treatments across a broad range of ecosystems before applying a one-size-fits-all oak regeneration treatment across the landscape.

CONCLUSIONS

Forestry models provide land managers a means to assess potential effects of alternative treatments in forested stands and are especially useful when site-specific information regarding the potential effects of a treatment is lacking. In research, model outcomes can help stimulate new research ideas and formulate hypotheses. Using FVS-SN alone in this study was not acceptable given the inability of the southern variant of FVS to sufficiently predict regeneration success following overstory removals in the proposed treatments. Alternatively, using REGEN alone would not have allowed us to sufficiently predict the effects of the intermediate treatments, e.g., prescribed burn, on the regeneration pools or predict the long-term stand development patterns and tradeoffs in proposed treatments. By combining the two models, we were able to diminish weaknesses in both models, thus, allowing for multiscale comparisons of treatment alternatives.

Stage (1973) noted in his first publication on Prognosis, the predecessor of FVS-SN, that our ecological and silvicultural knowledge is incomplete and as such our forestry models are incomplete. With this in mind it is important that forestry models allow land managers the ability to adjust model relationships as needed. We found FVS-SN to be flexible with respect to modifying regeneration inputs. It is also essential that forestry models are consistently maintained to facilitate the incorporation of new ecological findings; such is the case with FVS-SN, which reflects over two decades of scientific development. To meet the modeling needs of mixed-oak forests in the Southern Appalachians, we recommend to the FVS-SN staff that the REGEN model be fully integrated within the FVS-SN.

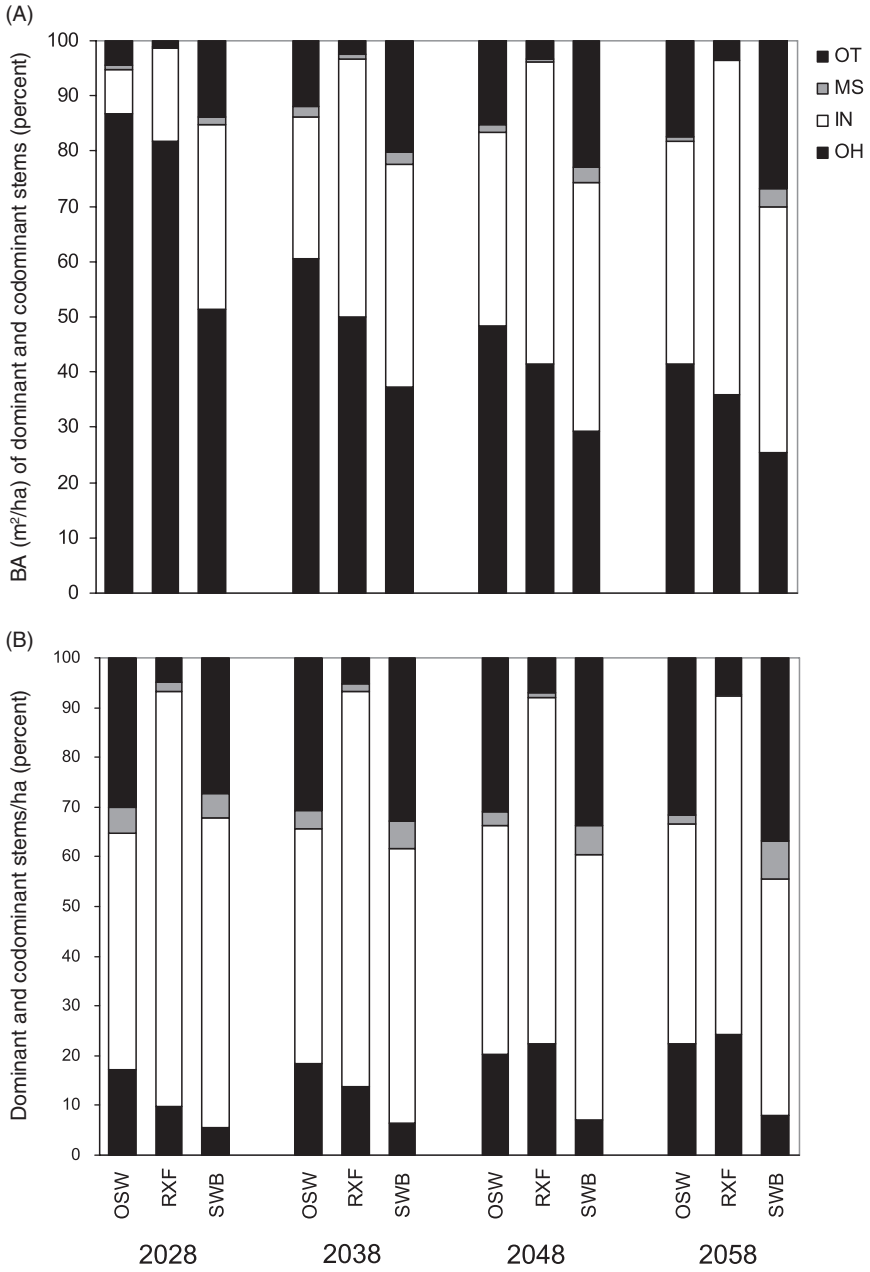

Figure 2—Predicted posttreatment distribution of (A) basal area (BA) (m^2/ha) and (B) stems/ha within the oak-hickory, intolerant, midstory, and other species group within the 5,010 stands. (OSW = oak shelterwood via herbicide, RXF = prescribed fire, SWB = shelterwood/burn).

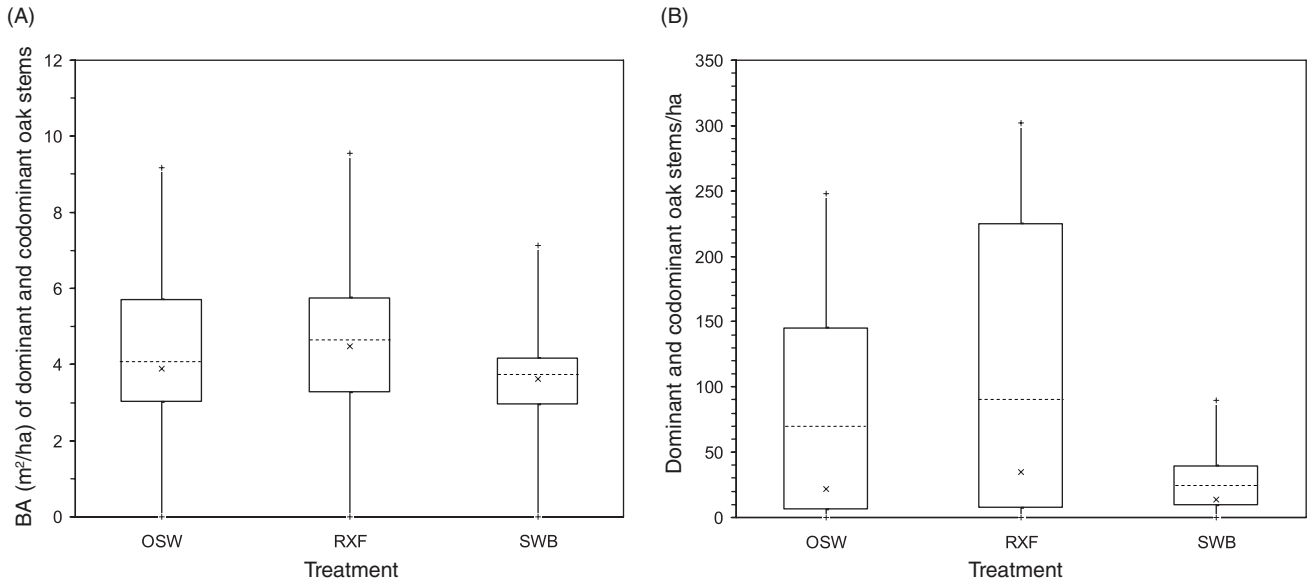

Figure 3—Predicted posttreatment (A) basal area (BA) (m^2/ha) and (B) stems/ha of oak species at the end of the 50-year forecast (year 2058) for the 5,010 stands. For each treatment, the + represents the minimum and maximum predicted values; x represents the median predicted value; the dashed line represents the mean predicted value; and the top and bottom lines of the boxes represent the upper and lower quartiles of the predicted variables, respectively. (OSW = oak shelterwood via herbicide, RXF = prescribed fire, SWB = shelterwood/burn).

LITERATURE CITED

Abrams, M.D. 1992. Fire and the development of oak forests. Bioscience. 42: 346–353.

Aldrich, P.R.; Parker, G.R.; Romero-Severson, J.; Michler, C.H. 2005. Confirmation of oak recruitment failure in Indiana old-growth forest: 75 years of data. Forest Science. 51: 406–416.

Alexander, H.D.; Arthur, M.A.; Loftis, D.L.; Green, S.R. 2008. Survival and growth of upland oak and co-occurring competitor seedlings following single and repeated prescribed fires. Forest Ecology and Management. 256: 1021–1030.

Beck, D.E.; Hooper, R.M. 1986. Development of a Southern Appalachian hardwood stand after clearcutting. Southern Journal of Applied Forestry. 10: 168–172.

Brose, P.H.; van Lear, D.H.; Keyser, P.D. 1999. A shelterwood-burn technique for regenerating productive upland oak sites in the Piedmont region. Southern Journal of Applied Forestry. 16: 158–163.

Crookston, N.; Dixon, G. 2005. The forest vegetation simulator: a review of its structure, content, and applications. Computers and Electronics In Agriculture. 49: 60–80.

Greco, S.; Baldocchi, D.D. 1996. Seasonal variations of CO2 and water vapour exchange rates over a temperate deciduous forest. Global Change Biology. 2: 183–197.

Gregg, T.F.; Hummel, S. 2002. Assessing sampling uncertainty in FVS projections using a bootstrap resampling method. In: Proceedings of the second forest vegetation simulator conference. RMRS-P-25. Ogden, UT: U.S Department of Agriculture Forest Service, Rocky Mountain Research Station: 164–167.

Hummel, S.; Cunningham, P. 2006. Estimating variation in a landscape simulation of forest structure. Forest Ecology and Management. 228: 135–144.

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.

Loftis, D.L. 1988. Regenerating oaks on high-quality sites, an update. In: Proceedings of the guidelines for regenerating Appalachian hardwood stands. Publ. 88-93. Bethesda, MD: Society of American Foresters: 199–209.

Loftis, D.L. 1990. A shelterwood method for regenerating red oak in the Southern Appalachian Mountains. Forest Science. 36: 917–929.

Lorimer, C.G. 1993. Causes of the oak regeneration problem. In: Proceedings of the oak regeneration: serious problems, practical solutions symposium. Gen. Tech. Rep. SE -84. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 14–39.

McNab, W.H.; Loftis, D.L. X. A preliminary test of estimating forest site quality using species composition in a Southern Appalachian watershed. Proceedings of the 15th Southern Biennial Silviculture Research Conference.

Orwing, D.A.; Abrams, M.D. 1994. Land-use history (1720-1992), composition, and dynamics of oak-pine forests within the Piedmont and Coastal Plain of northern Virginia. Canadian Journal of Forest Research. 24: 1216–1225.

Rodewald, A.D. 2003. Decline of oak forests and implications for forest wildlife conservation. Natural Areas Journal. 23: 368–371.

Schnur, G.L. 1937. Yield, stand, and volume tables for even-aged upland oak forests. Tech. Bull. 560. Washington, DC: U.S. Department of Agriculture. 88 p.

Schweitzer, C.J.; Lofits, D.L.; Wang, Y.; Janzen, G.C. 2004. Regeneration potential of selected forested stands on the Cumberland Plateau of north Alabama: Proceedings of the upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 269–274.

Spetich, M.A.; Parker, G.R. 1998. Distribution of biomass in an Indiana old-growth forest from 1926 to 1992. American Midland Naturalist. 139: 90–107.

Stage, A.R. 1973. Prognosis model for stand development. Res. Pap. INT-137. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 32 p.

Whittaker, R.H.; Likens, G.E. 1975. The biosphere and man. In: Lieth, H.; Whittaker, R.H., eds. Primary productivity of the biosphere. Berlin, Germany: Springer-Verlag: 305–328.

A COMPARISON OF CANOPY STRUCTURE MEASURES FOR PREDICTING HEIGHT GROWTH OF UNDERPLANTED SEEDLINGS

John M. Lhotka and Edward F. Loewenstein[1]

Abstract—The study compares the relationship between 15 measures of canopy structure and height growth of underplanted yellow-poplar (*Liriodendron tulipifera* L.) seedlings. Investigators used 4 midstory removal intensities to create a structural gradient across fifty 0.05-ha experimental plots; removals resulted in a range of canopy cover between 51 to 96 percent. Twelve 1-year-old containerized yellow-poplar seedlings were planted within each plot. Height growth was monitored through two growing seasons (2004 to 2005). Investigators used regression analysis (n = 50) to predict 2-year height growth using measures of tree size and density, canopy openness, and vertical structure. Model of best-fit included height to the forest canopy and canopy cover estimated using crown width models (R^2 = 0.78). Results emphasize the potential importance of quantifying horizontal and vertical canopy characteristics when evaluating the relationship between forest structure and growth of underplanted seedlings.

INTRODUCTION

Underplanting involves the establishment of nursery grown tree seedlings under an existing forest canopy. The purpose of underplanting is to establish advance reproduction prior to harvest. Underplanting can help supplement natural pools of reproduction or establish high-value species in degraded stands or in stands lacking sufficient seed sources. Unlike the artificial establishment of seedlings following a complete overstory removal, survival and development of underplanted seedlings are influenced by the mitigating effect of the forest canopy on the understory environment (Paquette and others 2006). Existing research shows that field performance of underplanted seedlings is also linked to planting stock quality and size (Dey and Parker 1997a, Spetich and others 2002).

Silvicultural treatments affect the development of underplanted seedlings by altering the understory environment through canopy manipulation. To provide sufficient resources for seedlings, underplanting generally coincides with silvicultural treatments like midstory removal or shelterwood harvest (Dey and Parker 1997b, Johnson 1984, Teclaw and Isebrands 1993). Successful design of these silvicultural treatments is contingent upon understanding interactions between forest structure, understory environment, and physiologic response of forest reproduction. Without considering how the overwood influences growth and mortality of underplanted seedlings as well as their competitors, the success of underplanting operations may be limited. Quantitative approaches linking seedling growth to stand structure are important because they can help silviculturists develop appropriate residual density recommendations that can be practically applied by field foresters.

Our objective is to identify measures of canopy structure that can be used to predict initial height growth of underplanted seedlings along a gradient of partial harvest conditions. We present height growth models for underplanted yellow-poplar (*Liriodendron tulipifera* L.) developed using two groups of predictor variables: (1) all measures of canopy structure evaluated and (2) only measures of canopy structure that can be derived from tree inventory data. We hypothesize that models will incorporate measures of canopy structure and seedling size. Based upon published relationships between forest structure and the understory environment (Lhotka and Loewenstein 2006), we further hypothesize that the seedling growth models will include variables describing horizontal and vertical characteristics of the forest canopy.

METHODS

Site Description

The study was conducted within the riparian forest corridor of a 450-ha watershed in Harris County, GA (approximately 32° N, 85° W). The site is within the lower Piedmont physiographic region. The overstory was primarily composed of yellow-poplar and sweetgum (*Liquidambar styraciflua* L.). Water oak (*Quercus nigra* L.), green ash (*Fraxinus pennsylvanica* Marsh.), and boxelder (*Acer negundo* L.) are minor components of the stand. A dense midstory was present across much of the area, dominated by flowering dogwood (*Cornus florida* L.), two-wing silverbell (*Halesia diptera* Ellis), musclewood (*Carpinus caroliniana* Walt.), and ironwood [*Ostrya virginiana* (Mill.) K. Koch]. The understory was primarily composed of Japanese honeysuckle (*Lonicera japonica* Thunb.), Nepalese browntop [*Microstegium vimineum* (Trin.) A. Camus], and blackberry (*Rubus* spp.). No flooding occurred during the duration of this study.

Study Design

In August 2003, fifty 0.05-ha circular plots were established within portions of the riparian forest corridor that were at least 38 m wide. Plots were systematically located along a transect bisecting this corridor and a minimum of 38 m separated each plot center. To ensure that all plots were located under closed canopy conditions, establishment criteria ensured that all plot centers were not less than 19 m from the edge of the riparian corridor and not less than 12.6 m from a forest gap.

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Investigators created a canopy cover gradient across the 50 plots by randomly assigning 1 of 4 midstory removal treatments: (1) no midstory trees removed, (2) one-third of all midstory trees removed, (3) one-half of all midstory trees removed, and (4) complete midstory removal. Midstory trees were defined as stems not present in the dominant/codominant canopy layer. Structural manipulations were completed using directional chainsaw felling between August and October of 2003. Vegetation <1.4 m tall was not removed unless it created a safety hazard during felling operations. No trees were removed from the site but were cut up to facilitate underplanting and to speed decomposition. Following midstory treatments, twelve 1-year-old containerized yellow-poplar seedlings were planted within each plot. Seedlings were planted in a systematic grid located within the inner portion (6.31 m from plot center) of each plot. This planting area was selected so that the outer half of the plot could help buffer the effects of plot edge on the response of planted seedlings. All planting occurred between October and November of 2003. Readers should note that the yellow-poplar seedlings sustained wind damage at the nursery during the middle of the growing season and that the nursery manager ameliorated damage by top clipping seedlings to a uniform height. By fall planting, seedlings had new stem growth and a fully developed terminal bud.

Data Collection and Analysis

Seedling growth was monitored over two growing seasons (2004 and 2005), and seedling inventories were completed prior to budbreak in the spring of 2004 and after final terminal bud formation in the fall of 2004 and 2005. At each inventory, basal diameter (mm), height (cm), and survival status were recorded for the planted seedlings. To link growth of underplanted seedlings to canopy structure, metrics of canopy openness, stand density, tree size, and vertical structure were quantified.

Following midstory treatment, overstory tree inventories were completed for each 0.05-ha plot. All trees >5 cm d.b.h. were measured and total height (m), height (m) to the base of the live crown (HBLC), d.b.h. (cm), and species were recorded. Tree inventory data were summarized to determine density (trees/ha), basal area (m^2/ha), and quadratic mean diameter (cm). Measures of vertical structure were derived from tree inventory data including average HBLC, average tree height, top height, and average canopy depth (e.g., average tree height – average HBLC). Vertical structure of each plot was also characterized by measuring height to the forest canopy (m) above each seedling. Height-to-canopy was defined as the vertical distance (m) from a seedling to the nearest overhead tree crown. Vertical distances were measured using a Vertex III digital hypsometer.

In the summer of 2004, investigators quantified canopy openness using measures of percent canopy cover and canopy closure for each plot. A GRS Densitometer (Geographic Resource Solutions, Arcata, CA) was employed to estimate canopy cover using the vertical sighting tube method (Johansson 1985). Observations were taken on 2- by 2-m grid with a total of 113 points located on each plot. The instrument was leveled at every sample point and the presence or absence of canopy was tallied. Percent cover was calculated by dividing the number of points where canopy was present by the total number of sample points. Canopy cover was also estimated using tree inventory data and species-specific crown width models. Calculating canopy cover using crown area projection involved three computational steps. Allometric crown width models were used to estimate each tree's horizontally projected crown area (Bechtold 2003). These estimated crown areas were then summed to determine a plot's total projected crown area (CA_{tot}). Finally, percent canopy cover was determined by inputting CA_{tot} into the crown overlap correction function (equation 1) presented by Crookston and Stage (1999).

$$\text{Percent canopy cover} = 100\left(1 - \exp\left(-0.01 \times 100 \times \frac{CA_{tot}}{10000}\right)\right) \qquad (1)$$

where
exp = exponential function
CA_{tot} = plot's total projected crown area

Canopy closure was estimated using a convex spherical densiometer. Readings were taken directly over plot center in each cardinal direction and average closure was recorded (Buckley and others 1999). Because research suggests that observer effect can introduce bias into densiometer readings (Vales and Bunnell 1988), a single individual collected the data. Hemispherical photography was also used to quantify canopy closure (Jennings and others 1999). One photograph was taken 1.25 m above each plot center using a Nikon Coolpix 5700 (5 megapixel) digital camera and fisheye converter (183° view angle). Although research suggests that digital and film hemispherical photography can yield comparable results (Englund and others 2000, Hale and Edwards 2002), factors such as digital image size, compression, quality, and saturation can influence the analysis of digital fisheye photos. To minimize these issues, the following camera settings were used: (1) image quality—1 to 4 compression JPEG format, (2) saturation—black and white, and (3) image size—full (2,560 by 1,920 pixels) (Frazer and others 2001). Additionally, all photos were taken during overcast conditions when the solar disk was completely obscured. The camera was leveled and the fisheye lens oriented toward magnetic north using a compass prior to each shot. Visible sky proportion was obtained from the hemispherical photographs by using Hemiview software (Delta-T Devices Ltd., Cambridge, UK) and canopy closure (1—visible sky proportion) was calculated. Threshold pixel classification of "sky" vs. "canopy" was set individually for every photo; one operator completed all analyses. Photo analysis was completed at four view angles 180°, 120°, 90°, and 60° by constraining the proportion of the photo processed by Hemiview.

The goal of our analysis was to determine the relationship between the measures of forest canopy structure and the 2-year growth of underplanted yellow-poplar seedlings. Analysis was completed at the plot level and used average 2-year (2004 to 2005) height growth by plot (n = 50) as the response variable. The predictor variables evaluated included metrics of tree size and density, canopy openness, and vertical structure (table 1). Simple linear regression was

Table 1—Descriptive statistics for plot-level canopy structure and underplanted seedling data (n = 50)

Plot-level variables	Mean	Minimum	Maximum	Standard deviation	R-square[a]
Quadratic mean diameter (cm)	31.60	14.38	60.67	10.58	0.37
Density (trees/ha)	569.20	120.00	1860.00	371.30	0.44
Basal area (m^2/ha)	34.85	12.91	62.85	9.20	0.09
Top height (m)	32.24	25.94	39.57	3.89	0.12
Tree height (m)	21.43	11.44	38.76	6.77	0.40
Height to the base of live crown (m)	10.53	4.76	20.94	3.90	0.32
Canopy depth (m)[b]	21.72	13.12	31.27	3.93	0.35
Height to the forest canopy (m)	14.23	2.35	30.92	6.32	0.68
Percent cover—vertical sight tube	84.04	51.32	95.57	8.76	0.36
Percent cover—crown width models	77.24	56.20	92.72	9.55	0.60
Closure—spherical densiometer	0.91	0.75	0.96	0.05	0.45
Closure—photo angle 180	0.92	0.90	0.95	0.01	0.31
Closure—photo angle 120	0.87	0.80	0.92	0.03	0.19
Closure—photo angle 90	0.82	0.71	0.89	0.05	0.16
Closure—photo angle 60	0.79	0.60	0.91	0.07	0.11
Mean seedling diameter at planting (cm)	8.01	6.34	9.52	0.78	0.01
Seedling 2-year height growth (cm)	68.98	17.50	177.60	35.84	1.00

[a] Coefficient of determination (R^2) for relationship between the given variable and 2-year (2004–05) height growth (cm) of underplanted yellow-poplar seedlings.

[b] Canopy depth = (average tree height – average height to base of live crown).

used to quantify the relationship between each canopy structure measure and 2-year height growth. Next, multiple regression was used to construct best-fit models from two groups of variables. The first set of models evaluated each of the forest structural metrics reviewed by the study and the second incorporated only variables derived from tree inventory data. Given known relationships among canopy structure, understory microclimate, and tree ecophysiology (Assenac 2000), we hypothesized that best-fit models would include measures of canopy openness, vertical structure, and a measure of seedling size at planting. Average initial basal diameter was used as the measure of seedling size. Goodness-of-fit was evaluated using the coefficient of determination, commonly referred to as R^2 (Neter and others 1996). A Box-Cox power transformation (equation 2) was used to meet homogeneity of variance and normality of residuals assumptions (Ott 2005). Box-Cox transformation power (λ) was determined in SAS using PROC Transreg.

$$Y_t = \frac{Y_i^{\lambda} - 1}{\lambda} \qquad (2)$$

where
Y_t = Box-Cox power transformed observation
Y_i = observed value
lambda (λ) = Box-Cox transformation power

For models of best-fit, variance inflation factor (VIF) was used to evaluate multicollinearity. Any variable with a VIF greater than 10 was removed from the model (Neter and others 1996).

RESULTS AND DISCUSSSION

Our goal was to determine the relationship between the measures of forest canopy structure and the 2-year growth of underplanted yellow-poplar seedlings across a gradient

of partial harvest conditions. The random application of four midstory removal intensities was successful at creating a canopy structure gradient across the experimental plots. The canopy cover gradient was between 51 and 96 percent. Height to the forest canopy (height-to-canopy) ranged from 2 to 31 m and residual basal area was between 12 and 63 m^2/ha.

Of the variables evaluated in this study, height-to-canopy (R^2 = 0.68) and canopy cover estimated using crown area projection (R^2 = 0.60) were most strongly related to height growth of the underplanted yellow-poplar seedlings. Other variables that explained >30 percent of the variation in 2-year height growth included: (1) spherical densiometer estimates of canopy closure, (2) stand density, (3) average tree height, (4) vertical sighting tube estimates of canopy cover, (5) average canopy depth, (6) average HBLC, and (7) hemispherical photo derived canopy closure (180° view angle) (table 1).

Models of best-fit were developed using two groups of predictor variables: (1) all measures of canopy structure evaluated and (2) only measures of canopy structure that can be derived from tree inventory data. The model developed for each group of variables explained at least 70 percent of the variance in 2-year height growth and included canopy cover estimated using crown area projection and a measure of vertical structure (equations 3 and 4). The presented models support our hypothesis that variables describing both horizontal and vertical canopy structure are needed to adequately predict seedling growth. Unlike our hypothesized model structure, average seedling size (basal diameter) was not a significant predictor of height growth at the plot level. This may have been due to the relative uniformity of the planting stock.

Model of best-fit (R^2 = 0.77)

$$\text{height growth}_{\text{Trans}} = 11.8542 + 0.1541\ (\text{height-to-canopy}) - 0.0753\ (\text{canopy cover}_{\text{CA}}) \quad (3)$$

Tree inventory based model (R^2 = 0.70)

$$\text{height growth}_{\text{Trans}} = 15.5557 - 0.1190\ (\text{canopy cover}_{\text{CA}}) + 0.1714\ (\text{average canopy depth}) \quad (4)$$

where

height growth$_{\text{Trans}}$ = Box-Cox transformed 2-year height increment with a lambda transformation power of 0.30

canopy cover$_{\text{CA}}$ = percent canopy cover estimated using crown area projection

height-to-canopy = average height (m) to the forest canopy above underplanted seedlings

average canopy depth = average tree height – average height to the base of live crown

Models suggest that average 2-year height growth increases as height-to-canopy increases and canopy cover decreases (fig. 1). Because midstory removal and/or shelterwood harvests decrease canopy cover and increase the vertical distance between the forest floor and the canopy (Loftis 1990), results of this study support the application of these treatments as a method for enhancing height growth of underplanted yellow-poplar seedlings. Trends presented in figure 1 could be used to determine the average height growth response that may result from any given residual height-to-canopy and canopy cover combination. While models explained more than 70 percent of the variance in 2-year height growth, lack of site replication across the landscape limits the applicability of the presented models. However, the outlined methodology may serve as a framework for the development of quantitative approaches that link growth of underplanted seedlings to variables describing the stand structure. A model based solely on metrics derived from tree inventory data could potentially be linked with a stand development model, e.g., Forest Vegetation Simulator, to evaluate how residual structure affects seedling response. This linkage may allow managers to evaluate how a suite of silvicultural practices affect growth of underplanted seedlings, to identify a target residual structure, and to produce stand structure-based marking prescriptions that can be implemented by field foresters. Finally, results emphasize the potential importance of quantifying horizontal and vertical canopy characteristics when evaluating the relationship between forest structure and growth of underplanted seedlings.

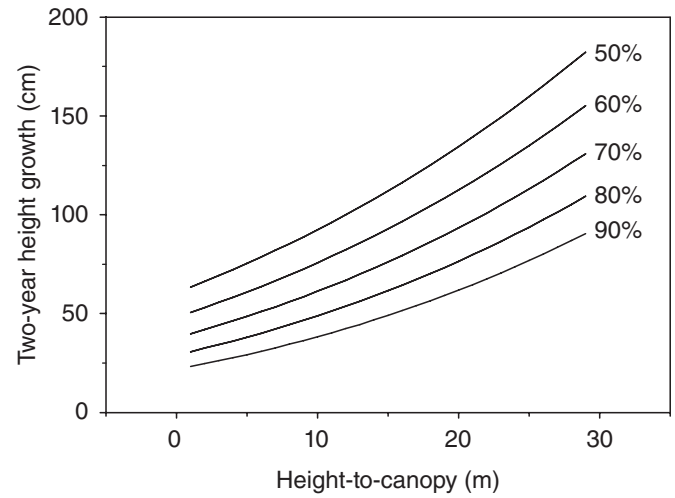

Figure 1—Generalized relationship between height-to-canopy and 2-year (2004 and 2005) height growth of underplanted yellow-poplar seedlings at five levels of canopy cover. The height-to-canopy and canopy cover array used to estimate height growth trends fall within the study's observed data range.

LITERATURE CITED

Assenac, G. 2000. Interactions between forest stands and microclimate: ecophysiological aspects and consequences for silviculture. Annals of Forest Science. 57: 287–301.

Bechtold, W.A. 2003. Crown-diameter prediction models for 87 species of stand-grown trees in the Eastern United States. Southern Journal of Applied Forestry. 27(4): 269–278.

Buckley, D.S.; Isebrands, J.G.; Sharik, T.L. 1999. Practical field methods of estimating canopy cover, PAR, and LAI in Michigan oak and pine stands. Northern Journal of Applied Forestry. 16(1): 25–32.

Crookston, N.L.; Stage, A.R. 1999. Percent canopy cover and stand structure statistics from the Forest Vegetation Simulator. Gen. Tech. Rep. GTR-RMRS-024. Ogden, UT: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 11 p.

Dey, D.C.; Parker, W.C. 1997a. Morphological indicators of stock quality and field performance of red oak (*Quercus rubra* L.) seedlings underplanted in a central Ontario shelterwood. New Forests. 14(2): 145–156.

Dey, D.C.; Parker, W.C. 1997b. Overstory density affects field performance of underplanted red oak (*Quercus rubra* L.) in Ontario. Northern Journal of Applied Forestry. 14(3): 120–125.

Englund, S.R.; O'Brien, J.J.; Clark, D.B. 2000. Evaluation of digital and film hemispherical photography and spherical densiometry for measuring forest light environments. Canadian Journal of Forest Research. 30: 1999–2005.

Frazer, G.W.; Fournier, R.A.; Trofymow, J.A.; Hall, R.J. 2001. A comparison of digital and film fisheye photography for analysis of forest canopy structure and gap light transmission. Agricultural and Forest Meteorology. 109: 249–263.

Hale, S.E.; Edwards, C. 2002. Comparison of film and digital hemispherical photography across a wide range of canopy densities. Agricultural and Forest Meteorology. 112(1): 51–56.

Jennings, S.B.; Brown, N.D.; Sheil, D. 1999. Assessing forest canopies and understory illumination: canopy closure, canopy cover and other measures. Forestry. 72(1): 59–73.

Johansson, T. 1985. Estimating canopy density by the vertical tube method. Forest Ecology and Management. 11: 139–144.

Johnson, P.S. 1984. Responses of planted northern red oak to three overstory treatments. Canadian Journal of Forest Research. 14(4): 536–542.

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(1/3): 310–318.

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

Neter, J.; Kutner, M.H.; Nachtshein, C.J.; Wasserman, W. 1996. Applied linear statistical models. New York: McGraw-Hill, Inc. 1,408 p.

Ott, P. 2005. The Box-Cox transformation. Biom. Inf. Pam. 62. British Columbia Ministry of Forests, Research Program. 5 p. http://www.for.gov.bc.ca/hre/biopamph/pamp62.pdf. [Date accessed: July 13, 2011].

Paquette, A.; Bouchard, A.; Cogliastro, A. 2006. Survival and growth of under-planted trees: a meta-analysis across four biomes. Ecological Applications. 16(4): 1575–1589.

Spetich, M.A.; Dey, D.C.; Johnson, P.S.; Graney, D.L. 2002. Competitive capacity of *Quercus rubra* L. planted in Arkansas' Boston Mountains. Forest Science. 48(3): 504–517.

Teclaw, R.M.; Isebrands, J.G. 1993. An artificial regeneration system for establishing northern red oak on dry-mesic sites in the Lake States, USA. Annals of Forest Science. 50(6): 543–552.

Vales, D.J.; Bunnell, F.L. 1988. Comparison of methods for estimating forest overstory cover. I. Observer effects. Canadian Journal of Forest Research. 18: 606–609.

EFFECT OF DIRECTED-SPRAY GLYPHOSATE APPLICATIONS ON SURVIVAL AND GROWTH OF PLANTED OAKS AFTER THREE GROWING SEASONS

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Abstract—Thousands of acres of oak (*Quercus* spp.) plantations are established across the South annually. Survival and growth of these plantings have been less than desirable. Several techniques have been utilized in attempts to achieve improved success in these areas. One such technique that has been recommended is the application of directed-spray herbicide treatments. In this study, 3,240 bare-root Nuttall oak (*Q. nuttallii* Palmer) and white oak (*Q. alba* L.) seedlings were planted in February 2005 on Malmaison Wildlife Management Area near Carrollton, MS. One-third of the seedlings were subjected to a pre-emergent Oust® XP application in March 2005. One-third of the seedlings were treated with a pre-emergent Oust® XP application in March 2005 and directed-spray glyphosate applications throughout the 2005 growing season. One-third of the seedlings were subjected to pre-emergent Oust® XP applications in March of 2005, 2006, and 2007, and received directed-spray glyphosate applications once a month during the growing season each year. The effect of repeated directed-spray applications was evaluated, and significant differences were noted. Multiple examples of adverse seedling conditions were observed in the 3-year directed-spray plots. Seedling survival and total height were appreciably lower in 3-year directed-spray areas compared to nondirected-spray or 1-year directed-spray areas.

INTRODUCTION

Thousands of acres are being converted to oak (*Quercus* spp.) plantations across the South annually. The performance of these plantings has been less than desirable in many attempts. Several factors are paramount in achieving the establishment of successful plantations in these areas. These factors include matching species to the site, proper handling of seedlings both on and off site prior to planting, and using proper planting techniques. Even when proper site-species relationships have been considered and seedlings of the proper species have been planted, vegetative competition is still a concern. Much of the acreage on which new hardwood plantations are being established is in the form of retired agricultural fields (Stanturf and others 2004). In these settings, herbaceous competition will often be intense, and the possibility of exotic or aggressive weed species is of serious concern. If these problematic species are present and not eliminated before planting, some form of postplant competition control may be necessary to establish satisfactory hardwood plantations. One such form is the application of directed-spray herbicide treatments.

If competing vegetation is not controlled with site preparation efforts, and aggressive weed species resistant to broad spectrum herbaceous chemical control are present, directed-spray herbicide applications are often prescribed. Typically, an application of a foliar active herbicide is applied to competing vegetation around the seedling. The seedling is either shielded, or herbicide is carefully applied immediately adjacent to the seedling without the use of shielding. In both cases, extreme caution is taken to avoid wind drift or vegetative drip of the herbicide onto the seedling. Most research involving the use of directed-spray herbicide applications has been at an experimental level with special precautions being given to precise application of herbicides. Under these settings, an extreme amount of time and effort is spent in an attempt to keep herbicide off of planted seedlings. If weather conditions are not optimal for application, the researcher may be able to delay spraying until more favorable conditions exist. This level of discretion and caution may not be possible in operational settings using migrant workers, contractors, or field personnel. Directed spraying in operational settings often takes place on a much more massive scale than most research efforts. Also, contractors performing operational directed-spray applications work under various time and expense constraints that the scientist does not have to consider.

Observing 25 seedlings each of green ash (*Fraxinus pennsylvanica* Marsh.) and sweetgum (*Liquidambar styraciflua* L.), Hopper and others (1992) found significantly greater 4-year survival in plots undergoing a directed application of Roundup® than in control plots (77 and 61 percent, respectively). Seedling growth was not affected by herbicide treatment. Moree and others (2010) indicated that white oak (*Q. alba* L.) survival was negatively impacted on a Delta site in Mississippi when repeated directed glyphosate applications came into contact with seedlings. This comparison was made while observing 1,080 seedlings in two different directed-spray regimes and 540 seedlings in control areas. Initial vegetation was high and some seedlings were affected by herbicidal drift and/or dripping from overhanging vegetation.

Ware and Gardiner (2004) observed that 600 3-year-old Nuttall oak (*Q. nuttallii* Palmer) seedlings planted in a cutover were 44 percent more likely to be in a free-to-grow crown position when treated with directed-spray applications of Roundup® Ultra. While this treatment did not statistically impact survival or growth, the increase in free-to-grow

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position could have significant impacts in later measurements. Zutter and others (1987) performed two studies in South Carolina which indicated that sweetgum seedlings undergoing a directed glyphosate application exhibited significantly greater 5-year height than seedlings in untreated control plots (12.0 and 7.9 feet, respectively). Diameter growth followed a similar pattern with directed-spray treated seedlings exhibiting 0.43 inches more growth compared to untreated seedlings, and seedlings in disked plots were significantly smaller than seedlings in directed-spray plots. There were no observed survival differences between disking, directed-spray application, and a combination of both. Directed-spray herbicide application appeared to increase seedling growth through competition reduction. Disking did not appear to be as beneficial since weed roots were not killed, resulting in continued competition with seedlings. Both studies utilized 40 trees or less per treatment.

Postplanting control of woody species in hardwood plantations is typically performed using directed-spray applications of glyphosate or triclopyr (Stanturf and others 2004). Use of these potentially damaging materials is dictated by the tolerance of woody species to chemical compounds typically used in herbaceous vegetation control. When using directed-spray application of herbicides the compound sprayed must not be soil active, thus foliar active products containing triclopyr, glyphosate, or 2, 4-DP have been recommended (Miller 1987).

OBJECTIVES

The objectives of the study were:

1. To evaluate effects of directed-spray glyphosate applications on 3-year survival of oaks
2. To evaluate effects of directed-spray glyphosate applications on 3-year height growth of oaks

MATERIALS AND METHODS

Site Description

The tract selected for this study was formerly in row crop production and is located approximately 14 miles north of Greenwood, MS (90.0531° W, 33.6876° N) in Grenada County. The site was retired from row crop production sometime in the late 1990s. It has been maintained as an opening for wildlife through mowing and disking from agricultural retirement until the present. The study area encompasses approximately 7 1/2 acres on Waverly silt loam (coarse-silty, mixed, acid, thermic Cumulic Normaquept) and Falaya silt loam (coarse-silty, mixed, acid, thermic Aeric Cumulic Normaquept) soil series with slopes between zero and 2 percent. These soils are poorly drained and somewhat poorly drained, respectively. Average yearly temperature is 63.9 °F, and average yearly precipitation is 52.30 inches (Soil Conservation Service 1967). Soil tests indicate that the site has silt loam texture with pH ranging from 6.3 to 7.0.

There was a well-established herbaceous ground cover with a scattered woody component at the time of site selection. The dominant herbaceous species onsite included: Johnsongrass [*Sorghum halepense* (L.) Pers.], Bermudagrass [*Cynodon dactylon* (L.) Pers.], broomsedge bluestem (*Andropogon virginicus* L.), yellow nutsedge (*Cyperus esculentus* L.), blackberry (*Rubus argutus* Link), poison ivy [*Toxicodendron radicans* (L.) Kuntze], Brazilian vervain (*Verbena brasiliensis* Vell.), and Carolina horsenettle (*Solanum carolinense* L.). There were small scattered components of American sycamore (*Platanus occidentalis* L.), black willow (*Salix nigra* Marsh.), and boxelder (*Acer negundo* L.) across the entire site.

Study Design and Plot Establishment

A three-split strip-plot in a randomized complete block design with subsampling was used in this experiment. Three replications consisting of one of each possible treatment combination were established. Several of these combinations were created for the study of factors not reported in this paper. Each replication consisted of 72 plots approximately 150 feet by 10 feet and contained 15 seedlings. A total of 216 treatment plots were established. Plots were marked at each end with 4-foot sections of 3/4-inch polyvinyl chloride (PVC) pipe. Individual seedling locations were determined and marked with 36-inch pin flags color specific to species. All boundary lines were delineated using a compass and a 100-foot surveyor's tape.

Seedling Establishment

Nuttall oak and white oak seedlings were chosen for this study because they are two of the most commonly planted species in the South. Seedlings were purchased from Molpus Timberlands Nursery near Elberta, AL. On February 19, 2005, Mississippi State University personnel planted 1,620 1-0 bare-root seedlings of each species using a 10-feet by 10-feet spacing. Seedlings were transported to and stored onsite in a walk-in cooler until the time of planting.

Treatments

Three competition control treatments were used in this study. These treatments included a pre-emergent application only, a control for one full growing season, and a total competition control treatment. The pre-emergent application only treatment consisted of one application of Oust® XP applied over the top of seedlings at a rate of 2 ounces per acre in March 2005. The one growing season and total herbicide control treatments were the same for the first growing season. These treatments also included a pre-emergent application of Oust® XP in March 2005. Additionally, the one growing season and total herbicide control treatments also consisted of directed-spray applications of glyphosate (1.5 percent v/v) once a month from June to October 2005 for control of forbs and other plants not controlled by the earlier applications. An herbicide regime identical to that used in year 1 in the one growing season and total herbicide control treatment areas was utilized in years 2 and 3 on plots in total herbicide control treatment areas.

Field Data Collection

Initial seedling height measurements were taken April 15 to 25, 2005. Height was measured to the nearest tenth of

a centimeter using a height stick. Seedling survival was based on ocular evaluation and was recorded at the end of the growing season in 2005, 2006, and 2007. All missing seedlings were considered dead. If a seedling was observed as a resprout in later observations, it was reinstated into earlier survival estimations. The cambium was checked on seedlings which appeared dead to ensure survival status.

Statistical Analysis

Statistical analyses were performed using a mixed procedure in Statistical Analysis System (SAS) software version 9.1 and 9.1.3. Analyses were separated by species. A mixed model analyses of variance (ANOVA) was used to test for effects and interactions. Data were analyzed using least square means (LSMEANS). Survival percentages were arcsine square root transformed to normalize the data. However, actual means are presented for ease of data interpretation. Means were considered significant if $P < 0.05$.

RESULTS

Survival

Overall, survival was 78.1 percent for Nuttall oak and 57.8 percent for white oak. The 20.3 percent difference in overall survival was thought to result from the wet site conditions prevalent on this tract. The site was saturated for much of the winter and early growing season for all 3 years of the study. Inundation and/or soil saturation could have had negative impacts on white oak survival and growth. Survival of Nuttall oak was appreciably greater than survival observed in comparably treated white oak (table 1). These survival differences ranged between 5.8 and 30.6 percent among identical herbicide regimes.

The lowest seedling survival was observed in total herbicide control areas (table 1). Nuttall oak survival in this treatment (61.7 percent) was significantly lower compared to survival in the one growing season and pre-emergent only treatments (87.6 and 83.1 percent, respectively). White oak survival did not differ significantly among any of the three treatments. Treatment effects could have been masked due to much lower overall survival exhibited by white oak as a result of detrimental site conditions.

Total Height

Average total height was 5.71 feet for Nuttall oak and 4.09 feet for white oak. Substantially greater total height was observed in Nuttall oak than in white oak treated comparably (table 2). Significantly lower total height was observed for both species in total herbicide control and one growing season treatment areas compared to pre-emergent only areas. Average tree heights of both species in the total herbicide control and one growing season treatment areas were not statistically different.

SUMMARY AND DISCUSSION

Nuttall oak exhibited greater survival and height compared to white oak in each herbicide regime. Nuttall oak survival was significantly greater for seedlings in the one growing season and pre-emergent only treatments compared to seedlings in the total herbicide control treatment. No notable difference between treatments was observed for white oak. While white oak can survive as well as Nuttall oak under more suitable site conditions, lack of statistical difference between treatments was probably the result of the low overall survival of white oak due to wet site conditions. The significantly lower survival in total herbicide control areas indicates a negative effect of repeated glyphosate applications.

Total height was significantly greater for both species in the pre-emergent only areas compared to the other two treatments. Multiple examples of adverse treatment effects on seedlings were observed in the total herbicide control and one growing season plots. Repeated herbicidal contact of the seedlings resulted in lower overall height growth of trees in these areas. There were also some discrepancies between calculated height averages and realistic evaluations of overall treatment effect due to “end tree” influence. “End trees” were located at plot ends and often included the first and sometimes the second tree from either plot end. “End trees” occurred in more intensive treatments and were created through individual trees receiving comparatively little glyphosate contact with respect to other trees in the plot. These trees had substantial influence on height averages and statistical comparisons, effectively serving as outliers pulling height averages upwards.

Table 1—Survival by herbicide regime combination

Herbicide regime	White oak	Nuttall oak
	---------- *percent* ----------	
Pre-emergent only	61.5 a	83.1 a
One growing season	57.0 a	87.6 a
Total herbicide control	55.9 a	61.7 b

Values within a column followed by different letters are significantly different at $\alpha = 0.05$ according to Duncan’s Multiple Range Test.

Table 2—Total height by herbicide regime combination

Herbicide regime	White oak	Nuttall oak
	---------- *feet* ----------	
Pre-emergent only	4.60 a	6.20 a
One growing season	3.99 a	5.55 a
Total herbicide control	3.85 a	5.53 b

Values within a column followed by different letters are significantly different at $\alpha = 0.05$ according to Duncan’s Multiple Range Test.

CONCLUSIONS

Overall, best survival and growth results were observed in areas not undergoing or undergoing less directed-spray applications. Precautions were taken to ensure that seedlings were not accidentally contacted through either wind drift or vegetative drip. However, when performing 5 to 6 applications per year (15 to 18 applications over 3 years) contact inevitably occurs. While herbicidal contact might have only a slight effect if contact occurs once or twice, repeated monthly contact over several years has a substantial cumulative impact. This is of paramount concern when using migrant workers, contractors, or field personnel to perform directed spraying. Nonresearch oriented personnel may or may not have the expertise and/or knowledge to perform this technique satisfactorily. Directed spraying on an operational scale usually occurs on considerably more area at once than most research efforts. Contractors performing directed spraying under various time and expense constraints may not be able to postpone applications until ideal field conditions are present before spraying.

While plausible in research settings, directed spray applications may not be biologically or economically feasible for large-scale operations on retired agricultural areas across the South. Studies testing seedling survival and growth in areas treated with pre-emergent applications of broad spectrum herbicides before the first growing season have shown excellent results (Ezell 1999, Ezell and Catchot 1997, Ezell and Hodges 2002, Ezell and others 2007, Russell and others 1997). The costs of each treatment used in this study are as follows: pre-emergent only = $33.80 per acre, one growing season = $267.55 per acre, and total herbicide control = $802.65 per acre (at year 3). Economically, directed glyphosate applications are very cost prohibitive compared to a pre-emergent application of a broad spectrum herbicide, and their usage at this regime's intensity would not be possible for most landowners.

LITERATURE CITED

Ezell, A.W. 1999. Crop tolerance of Nuttall, water, and willow oaks to preemergent applications of DPX-6447: second year results. In: Proceedings tenth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-30. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 142–144.

Ezell, A.W.; Catchot, A.L., Jr. 1997. Competition control for hardwood plantation establishment. In: Proceedings ninth 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: Proceedings eleventh 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.

Hopper, G.M.; Buckner, E.R.; Mullins, J.A. 1992. Effects of weed control and fertilization on plantation establishment and growth of green ash, sweet gum, and loblolly pine: four-year results. In: Proceedings seventh biennial southern silvicultural research conference. Gen. Tech. Rep. SO-93. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 357–360.

Miller, J.H. 1987. The use of herbicides in hardwood forestry. In: Proceedings annual meeting of the Hardwood Research Council: 31-52. http://www.srs.fs.usda.gov/pubs/ja/ja_miller048.pdf. [Date accessed: July 15, 2011].

Moree, J.L.; Ezell, A.W.; Hodges, J.D. [and others]. 2010. Evaluating the use of enhanced oak seedlings for increased survival and growth: first-year survival. In: Stanturf, J.A., ed. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-121. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 165–169.

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: Proceedings ninth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 272–276.

Soil Conservation Service. 1967. Soil survey of Grenada County Mississippi. Washington, DC: U.S. Government Printing Office. 152 p.

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.

Ware, B.P.; Gardiner, E.S. 2004. Partial cutting and establishment of artificial Nuttall oak regeneration in the Mississippi Alluvial Plain. In: Proceedings twelfth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 587–591.

Zutter, B.R.; Nelson, L.R.; Minogue, P.L.; Gjerstad, D.H. 1987. Hardwood plantation growth following weed control using herbicide and cultivation. Southern Journal of Applied Forestry. 11(3): 134–138.

NINE-YEAR PERFORMANCE OF A VARIETY OF *POPULUS* TAXA ON AN UPLAND SITE IN WESTERN KENTUCKY

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Abstract—A variety of hybrid poplars have been planted on upland sites throughout the Midwest and Midsouth regions of the United States. Very few of these clones have proven to be worthwhile due to susceptibility to a variety of diseases. Five different *Populus* taxa were planted on an upland site in western Kentucky as a means of assessing resistance to local diseases, especially *Septoria musiva*. These taxa included combinations of *P. trichocarpa* crossed with *P. deltoides*, *P. maximowiczii*, and *P. nigra*, as well as backcrosses to *P. deltoides* and *P. maximowiczii*. Age 9 results indicated that survival for all five taxa was rather low with the *P. trichocarpa* × *P. deltoides* (TD) taxon being highest at 53.9 percent and the *P. trichocarpa* × *P. nigra* the lowest at 8.8 percent. In addition, the TD taxon also exhibited the best volumetric performance for age 9 diameter and height at 4.7 inches and 27.9 feet, respectively. Clone 24, a TD clone, exhibited the best age 9 survival, diameter, height, volume, and a good disease index rating at 96 percent, 7.2 inches, 37.4 feet, 4.34 cubic feet, and 2.20, respectively. Although, the TD taxon was the overall best performing taxa through age 9, a tremendous amount of variability exists among clones dictating testing of numerous clones prior to recommendation of large-scale plantings.

INTRODUCTION

Performance of fast-growth hardwood species established under plantation culture in the South, except for those planted on highly fertile alluvial and bottomland sites, has been rather poor. The ability to identify a specific hardwood species that combines rapid growth, ability to sustain rapid juvenile growth when grown over a wide geographic area and numerous soil types, disease resistance, and wood characteristics needed for an array of products has never been obtained. However, by limiting the geographic area, soil type, and the product, it may be possible to identify a fast-growth hardwood. The work reported in this paper was such an attempt made by Westvaco Corporation for upland sites in western Kentucky.

In general, eastern cottonwood (*Populus deltoides* Bartr.) plantations perform exceptionally well when grown on alluvial sites along the Mississippi River (Krinard 1985, Krinard and Johnson 1984, Rousseau 1987). Westvaco's Central Region had established thousands of acres of successful Mississippi River alluvial cottonwood plantations beginning in the mid-1970s but had not been successful in duplicating this performance outside of the alluvial area. The rapid growth of alluvial cottonwood plantations resulted in shorter rotation lengths and favorable economic returns. Unfortunately, alluvial cottonwood plantations are inaccessible during much of the winter and spring of each year thus dictating an additional source of this type of fiber. One approach to this problem was to investigate the possibility of fertigated plantations, i.e., plantations that are both fertilized and irrigated, and upland sites as sources of such fiber (Rainwater 1999). While the fertigated plantations concentrated on growing cottonwood plantations, the upland sites were more focused on hybrid poplars and aspen (*P. tremuloides* Michx.).

The reasoning behind this approach was partially taken from earlier studies that involved *P. trichocarpa* × *P. deltoides* hybrids on alluvial sites, Oxford Paper Company clones (also known as NE or OP clones) on upland sites, and a limited number of *P.* ×*canadensis* (synonym *P.* ×*euramericana*) hybrids (Dickmann and others, 2001) all of which were shown to be susceptible to Septoria leaf spot and stem canker (*Septoria musiva* Peck). This was not unexpected as work by Newcombe and Ostry (2001) has shown similar results in much of the Mississippi and St. Lawrence River drainages. The canker stage of Septoria causes the most damage, eventually ending in death of the infected tree (Ostry and others 1989). Hybrid poplars grown on the Mississippi River alluvial sites in western Kentucky quickly succumbed to Septoria cankers. Although some hybrid poplars survived on upland sites in western Kentucky when located in the Ohio River drainage, their growth was not suitable for large-scale plantation establishment. In addition, the majority of these clones were bred for survival and performance in either Europe or the Northeastern United States. It was hoped that planting the hybrid poplars on the uplands would remove them far enough from the main source of Septoria inoculum to prevent infection.

Hybrids resulting from the breeding of black cottonwood (*P. trichocarpa* Torr. & Gray) of the Northwestern United States and Canada have performed very well at various locations in the United States and Europe and warrant further investigation in experimentally designed tests. However, many F_1 hybrid poplars, especially black cottonwood hybrids, are susceptible to Septoria stem canker that renders them useless for plantation forestry. Eastern cottonwood is highly resistant and Japanese poplar (*P. maximowiczii* A. Henry) shows some resistance to Septoria stem canker. Backcrossing *P. trichocarpa* F_1 hybrids to their resistant parent may provide resistant offspring with the rooting and growth potential of the F_1 hybrids.

The Poplar Molecular Genetics Cooperative (PMGC), which was headquartered at the University of Washington, was

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formed in 1995. Westvaco became a member the following year. PMGC was established to increase the understanding of molecular genetic mechanisms causing variation in productivity and quality traits in hybrid poplar and to use the research results to accelerate progress in poplar breeding. Species and hybrid types created through controlled breeding that are used in their research are referred to individually as a "taxon." A byproduct of the cooperative's work is the generation of thousands of pedigreed progeny from a wide variety of interspecific and backcross hybrids. The breeding program primarily uses black cottonwood as the female parent with eastern cottonwood, black poplar (*P. nigra* L.), and Japanese poplar as the pollen parents. Balsam poplar (*P. balsamifera* L.) has been used to a lesser extent. F_1, backcross, and F_2 hybrids have been formed using most of the possible combinations. To take advantage of these diverse hybrid types, a series of studies were established on upland sites in Kentucky, Tennessee, West Virginia, and Virginia from 1999 through 2001. The western Kentucky test site will be reported on in this paper. The purpose of this study was to select hybrid poplar clones that exhibit excellent survival, rapid growth, and disease resistance making them suitable candidates for large-scale plantings on upland sites in western Kentucky.

METHODS

Planting Stock

Clones for the study originated from the breeding program of the PMGC. Three taxa were used as the female parents including pure *P. trichocarpa* (T) selections native to the Pacific Northwest and southwestern Canada, *P. trichocarpa* × *P. maximowiczii* (TM) hybrids, and *P. trichocarpa* × *P. deltoides* (TD) hybrids. Pollen parents included *P. deltoides* (D), *P. nigra* (N), and *P. maximowiczii* (M). Selection of clones for testing was based on parental pedigree information (table 1) and 1- or 2-year nursery growth data supplied by the cooperative. The decision was made to primarily use clones from progeny with parents from southern portions of their respective ranges, but other clones were included to provide diverse genetic backgrounds. No hybrids using *P. balsamifera* were selected. The PMGC was notified about the desired selections approximately 14 months prior to study establishment. They provided 1-year-old, dormant whips to Broadacres Nursery, Inc.,(Hubbard, OR) for propagation in the spring of the year preceding study establishment. Stock plants were propagated from dormant single bud cuttings, and additional ramets for the study were produced from in-leaf cuttings. The plants were grown throughout the summer in 24-cell trays with approximately 2.5- by 2.5- by 3-inch cell dimensions. After attaining dormancy, the plants were trimmed to 8-inch tops, removed from the containers, and stored in plastic bags at 28 °F. The dormant plants were shipped to Wickliffe, KY, in February 1999. In late March, the plants were sorted into replications and taxon blocks for the study site, repackaged in plastic bags, and stored at 35 °F until planted.

Experimental Design

The experimental design for the study is a compact family (split-plot) design with 8 replications, 5 taxa, and 10 clones per taxon. Main plots within a replication represent one taxon, and the subplots include the clones within a taxon. Each clone is represented by a two-tree plot. Certain clones were in limited supply or unavailable because of propagation difficulties; therefore, not every taxon was represented by 10 different clones. To insure a full subplot some clones are duplicated, but only a single designated two-tree measurement plot was included in the analysis. Thus, for the 1999 test site in western Kentucky there are 5 taxa tested, totaling 48 clones. The TD and TM subplots in Kentucky contain only nine clones (table 1).

Site Preparation and Establishment

The study site is located on the Cullom Tract in Livingston County, KY. The site is on a broad ridge and was previously planted in pine (*Pinus* spp.). Site preparation included shearing, raking, piling, and burning during the fall of 1998. The area was disked then row marked and slit at 10 by 10 feet. Study trees were planted on May 3, 1999. A one-row border of NM-6 (hybrid poplar known for its resistance to Septoria) surrounds the study (fig. 1). The area was disked twice during the summer to control weeds, and fertilizer was applied to each tree in June 1999 at a rate of 150 pounds nitrogen (ammonium nitrate) and 75 pounds phosphorous (triple super phosphate).

Measurements and Analysis

Total height, survival, and crown score measurements were taken at age 2. The crown score included both leaf retention and leaf color (table 2). Survival, total height, diameter, and disease ratings were measured at age 9. The age 9 disease rating system focused on canker development and included both limb and stem (table 2). Volume was calculated using an equation developed by Krinard (1988) for plantation grown eastern cottonwood, which was $0.06+0.00221(D^2H)$. Percent survival data was calculated and transformed using arcsine transformation followed by analysis of variance (ANOVA). ANOVA for all traits was performed using PROC GLM along with the RANDOM option to accommodate the unbalance in the design.

RESULTS

Overall survival in the study at age 2 was 92.4 percent. Among the five taxa, the TD hybrids had the highest survival at 97.7 percent while the TDD hybrids were the lowest at 88.8 percent. The TD and TM hybrids (95.8 percent) were the only two that did not differ significantly at age 2. The TDD, TMM (90 percent), and TN (91.3 percent) hybrids exhibited the lowest age 2 survival but were still acceptable.

By age 9, survival of the study dropped to 23.9 percent. Survival of all five taxa would be considered unacceptable, with the TD hybrids again exhibiting the highest survival at 53.9 percent. Age 9 survival of the TDD, TM, and TMM taxa was very similar at 20.6, 20.6, and 21.5 percent, respectively. The TN taxa had the lowest age 9 survival at 8.8 percent. Age 9 clonal survival was significantly different with two TD clones, i.e., 24 and 25, being the highest at 94 percent. Only one other clone, i.e., 28, which was also a TD hybrid, exhibited an

Table 1—Hybrid poplar clones included in the 1999 study identified by field number, taxa, origin, and parentage

Field		Female			Male		
Number	Taxa	Parent	Origin	Clone	Parent	Origin	Clone
1	TN	*Populus trichocarpa*	Washington	2499	*Populus nigra*	France	2554
2	TN	*P. trichocarpa*	Washington	2498	*P. nigra*	France	2554
3	TN	*P. trichocarpa*	Washington	2498	*P. nigra*	France	2557
4	TN	*P. trichocarpa*	Idaho	2518	*P. nigra*	France	2554
5	TN	*P. trichocarpa*	Washington	2499	*P. nigra*	France	2557
6	TN	*P. trichocarpa*	Washington	2499	*P. nigra*	France	2557
7	TN	*P. trichocarpa*	Washington	5103	*P. nigra*	France	2557
8	TN	*P. trichocarpa*	British Columbia	5092	*P. nigra*	France	2554
9	TN	*P. trichocarpa*	British Columbia	5051	*P. nigra*	France	2554
10	TN	*P. trichocarpa*	British Columbia	5098	*P. nigra*	France	2554
12	TDD	TD F_1	WA × TX	177	*P. deltoides*	Texas	961
13	TDD	TD F_1	WA × TX	177	*P. deltoides*	South Carolina	962
14	TDD	TD F_1	WA × TX	177	*P. deltoides*	Illinois	101
15	TDD	TD F_1	WA × TX	177	*P. deltoides*	Illinois	101
16	TDD	TD F_1	WA × IL	255	*P. deltoides*	South Carolina	982
17	TDD	TD F_1	WA × TX	177	*P. deltoides*	South Carolina	979
18	TDD	TD F_1	WA × MS	29	*P. deltoides*	Mississippi	951
19	TDD	TD F_1	WA × MS	29	*P. deltoides*	Mississippi	953
20	TDD	TD F_1	WA × TX	177	*P. deltoides*	Texas	960
21	TDD	TD F_1	WA × TX	177	*P. deltoides*	Mississippi	951
22	TDD	TD F_1	WA × TX	177	*P. deltoides*	Mississippi	951
24	TD	*P. trichocarpa*	Washington	2499	*P. deltoides*	South Carolina	982
25	TD	*P. trichocarpa*	Washington	2499	*P. deltoides*	South Carolina	982
26	TD	*P. trichocarpa*	Washington	2550	*P. deltoides*	Illinois	101
27	TD	*P. trichocarpa*	Washington	2550	*P. deltoides*	Illinois	101
28	TD	*P. trichocarpa*	British Columbia	5098	*P. deltoides*	South Carolina	982
29	TD	*P. trichocarpa*	Washington	2499	*P. deltoides*	Texas	961
30	TD	*P. trichocarpa*	Washington	2499	*P. deltoides*	Texas	961
31	TD	*P. trichocarpa*	British Columbia	2437	*P. deltoides*	Mississippi	953
33	TMM	TM F_1	F_1	256	*P. maximowiczii*	Hokkaido	5105
34	TMM	TM F_1	F_1	265	*P. maximowiczii*	Hokkaido	5105
35	TMM	TM F_1	F_1	265	*P. maximowiczii*	Hokkaido	5105
36	TMM	TM F_1	F_1	252	*P. maximowiczii*	Hokkaido	5105
37	TMM	TM F_1	F_1	252	*P. maximowiczii*	Hokkaido	5105
38	TMM	TM F_1	F_1	256	*P. maximowiczii*	Hokkaido	5105
39	TMM	TM F_1	F_1	256	*P. maximowiczii*	Hokkaido	5105
40	TMM	TM F_1	F_1	256	*P. maximowiczii*	Hokkaido	5105
41	TMM	TM F_1	F_1	279	*P. maximowiczii*	Hokkaido	5105
42	TMM	TM F_1	F_1	279	*P. maximowiczii*	Hokkaido	5105
44	TM	*P. trichocarpa*	Washington	2499	*P. maximowiczii*	Hokkaido	5105
45	TM	*P. trichocarpa*	Washington	2499	*P. maximowiczii*	Hokkaido	5105
46	TM	*P. trichocarpa*	Washington	2499	*P. maximowiczii*	Hokkaido	5105
47	TM	*P. trichocarpa*	Washington	2499	*P. maximowiczii*	Hokkaido	5105
48	TM	*P. trichocarpa*	British Columbia	2437	*P. maximowiczii*	Hokkaido	5104
49	TM	*P. trichocarpa*	British Columbia	2437	*P. maximowiczii*	Hokkaido	5104
50	TM	*P. trichocarpa*	Washington	2499	*P. maximowiczii*	Hokkaido	5105
51	TM	*P. trichocarpa*	Washington	2499	*P. maximowiczii*	Hokkaido	5105
52	TM	*P. trichocarpa*	British Columbia	2437	*P. maximowiczii*	Hokkaido	5104

NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6	NM6					
NM6																NM6					
NM6	**6**	2	8	1	3	**12**	21	17A	16	20	**F45**	30	29	27	25	NM6					
NM6																NM6					
NM6	**5**	4	10	9	7	**15**	13	19	18	14	**24**	31	F47	26	28	NM6	NM6	NM6	NM6	NM6	NM6
NM6																					NM6
NM6	**6**	10	2	9	4	**52**	44	48	F50	49	**26**	31	28	25	F24	**37**	33	38	41	36	NM6
NM6																					NM6
NM6	**1**	3	7	5	8	**45**	51	46	47	50	**29**	27	30	F25	24	**42**	40	39	34	35	NM6
NM6																					NM6
NM6	**28**	26	30	F45	31	**33**	38	34	40	42	**F52**	52	49	48	51	**4**	1	6	3	9	NM6
NM6																					NM6
NM6	**25**	27	F47	24	29	**37**	41	35	36	39	**46**	50	47	44	45	**2**	5	7	10	8	NM6
NM6																					NM6
NM6	**41**	34	36	39	40	**31**	27	26	F47	F47	**49**	52	F50	51	44	**13**	21	12	19	14	NM6
NM6																					NM6
NM6	**37**	35	38	33	42	**30**	28	29	24	25	**45**	46	47	48	50	**17**	20	15	16	18	NM6
NM6					NP											NM6	NM6	NM6	NM6	NM6	NM6
NM6	**47**	48	46	52	NP	**19**	20	15	14	16	**39**	42	33	35	37	NM6					
NM6																NM6					
NM6	**51**	44	45	50	49	**21**	18	12	17A	13	**38**	30	34	41	36	NM6					
NM6																NM6					
NM6	**21**	20	19	17A	12	**4**	9	7	10	3	**24**	26	27	25	30	NM6					
NM6																NM6					
NM6	**13**	15	16	18	14	**5**	6	2	8	1	**F22**	28	31	29	F22	NM6					
NM6																NM6					
NM6	**F45**	29	28	31	F22	**40**	39	38	37	41	**44**	51	45	52	47	NM6					
NM6																NM6					
NM6	**27**	25	26	30	24	**35**	34	36	33	42	**50**	F49	48	46	49	NM6					
NM6																NM6					
NM6	**18**	21	14	13	12	**13**	12	21	18	20	**2**	10	5	3	8	NM6					
NM6																NM6					
NM6	**17**	15	19	20	16	**19**	15	17	14	16	**9**	4	6	7	1	NM6					
NM6																NM6					
NM6	**33**	42	39	37	35	**30**	27	25	F47	26	**41**	35	40	38	39	NM6					
NM6																NM6					
NM6	**41**	40	38	36	34	**F45**	29	24	28	31	**36**	33	34	37	42	NM6					
NM6																NM6					
NM6	**50**	48	47	49	46	**45**	F50	48	49	50	**18**	19	21	13	20	NM6					
NM6																NM6					
NM6	**51**	45	52	F50	44	**47**	52	44	51	46	**12**	14	15	16	17A	NM6					
NM6																NM6					
NM6	**8**	6	2	7	3	**15**	12	17A	16	21	**6**	3	7	4	1	NM6					
NM6																NM6					
NM6	**1**	5	9	10	4	**13**	18	20	19	14	**9**	5	8	2	10	NM6					
NM6	HP	HP	HP	HP	HP											NM6					
					HP	**7**	6	10	8	2	**45**	47	50	F50	44	NM6					
					HP											NM6					
					HP	**5**	4	9	1	3	**48**	51	52	49	46	NM6					
					HP											NM6					
					HP	**39**	40	42	34	38	**24**	29	26	28	31	NM6					
					HP											NM6					
					HP	**33**	36	35	41	37	**27**	F51	F51	30	25	NM6					
					HP	HP	HP	HP	HP	HP	HP	HP	HP	HP	HP	HP					

Code	Family	Clone	Taxon
1	422	3135	TN
2	423	3150	TN
3	424	3372	TN
4	449	4779	TN
5	543	7526	TN
6	543	7527	TN
7	587	10128	TN
8	646	7659	TN
9	688	11014	TN
10	575	7852	TN
12	430	4955	TDD
13	431	11286	TDD
14	493	8336	TDD
15	493	11248	TDD
16	504	6130	TDD
17	426	3549	TDD
18	480	8301	TDD
19	481	11899	TDD
20	429	4954	TDD
21	425	3502	TDD
22	425	3505	TDD
24	447	4708	TD
25	447	4711	TD
26	584	5133	TD
27	584	5136	TD
28	573	11893	TD
29	446	4645	TD
30	446	4650	TD
31	465	4974	TD
33	526	6844	TMM
34	514	11960	TMM
35	514	11964	TMM
36	518	8855	TMM
37	518	8857	TMM
38	526	6837	TMM
39	526	6848	TMM
40	526	6859	TMM
41	530	11154	TMM
42	530	11155	TMM
44	547	11576	TM
45	547	11579	TM
46	547	11590	TM
47	547	11593	TM
48	537	11518	TM
49	537	11525	TM
50	547	11561	TM
51	547	11591	TM
52	537	11031	TM

R7

	TN	TDD	TD	R6
	TN	TM	TD	TMM
	TD	TMM	TM	TN
R8	TMM	TD	TM	TDD
	TM	TDD	TMM	R5
	TDD	TN	TD	
	TD	TMM	TM	R4
R1	TDD	TDD	TN	
	TMM	TD	TMM	
	TM	TM	TDD	
	TN	TDD	TN	R3
		N	TM	
		TMM	TD	

R2

TAXON LOCATIONS WITHIN RE

BOLD INDICATES ANGLE LOCATION

F INDICATES FILLER

HP INDICATES HYBRID POPLAR BORDER

NM6 INDICATES NM6 BORDER

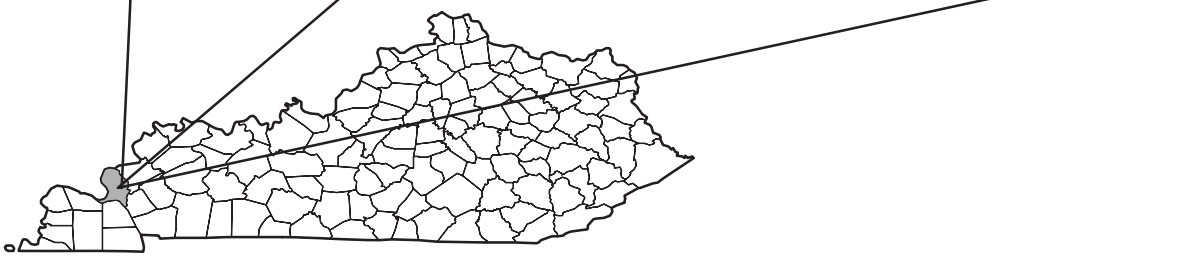

Figure 1—Geographic location of the study in western Kentucky and field design by taxa and clone.

Table 2—Age 2 crown ratings and age 9 Septoria disease index

Age 2 crown scores
1 = Full crown with leaves being green in color
2 = 25 percent defoliation with pale green leaves
3 = 75 percent defoliation with yellowish leaves
4 = >75 percent defoliation with remaining leaves yellow to brown in color
Age 9 Septoria disease rating
1 = No symptoms shown on either limbs or stem
2 = Canker visible on limbs but not on stems
3 = Canker visible on both stem and limbs
4 = Canker damage extensive enough to cause dieback to the groundline

age 9 survival >70 percent. A total of 14 clones had complete mortality. Three of these were TN clones, six TDD clones, and five TMM clones.

ANOVA for age 2 total height showed significant differences among taxa and among clones within taxa as well as the interaction terms of block by taxa and block by clones within taxa. Age 2 mean height for the study was 9.0 feet. The age 2 mean height for the five taxa were 9.5 feet for the TD and TM clones, 8.9 feet for the TMM clones, and 8.5 feet for the TDD and TN clones. Age 2 height of the TD clones ranged from 10.7 feet (clone 29) to 8.6 feet (clone 27). The age 2 height of the TM clones ranged from 10.3 feet (clone 52) to 8.8 feet (clone 46). The age 2 height range of the TMM clones was from 10.3 feet for clone 39 to 8.0 feet for clone 36. Age 2 heights for the clones in the TDD and TN taxa ranged from 9.7 feet (clone 14) to 7.2 feet (clone 20) and 9.6 feet (clone 4) to 6.4 feet (clone 8), respectively.

ANOVA for age 9 diameter and total height showed significant differences among taxa and among clones within taxa. Unlike the age 2 results, the interaction terms of block by taxa and block by clones within taxa were nonsignificant for age 9 diameter and total height. Age 9 mean diameter and height across all taxa in the study was 3.2 inches and 19.7 feet, respectively. The age 9 mean diameters for the five taxa were 4.7, 3.8, 2.5, 4.2, and 3.4 inches for the TD, TM, TMM, TN, and TDD taxa, respectively. The age 9 mean heights of the five taxa were 27.9, 18.5, 14.1, 18.6, and 19.4 feet for the TD, TM, TMM, TN, and TDD taxa, respectively. Clones within the TD taxon ranged from 37.4 (clone 24) to 20.6 (clone 31) feet, for age 9 height and 7.2 (clone 24) to 2.9 (clone 31) inches for age 9 diameter (table 3). Clones within the TDD taxon ranged from 24.0 (clone 16) to 12.5 (clone 18) feet for age 9 height and 4.8 (clone 16) to 2.5 (clone 21) for age 9 diameter (table 3). Age 9 diameter and height performance of the clones within the TM taxon ranged from 5.6 (clone 45) to 1.8 (clone 48) inches and 28.0 (clone 47) to 11.3 feet (clone 49), respectively (table 3). The clones of the TMM taxon showed a range of 3.6 (clone 34) to 1.6 (clone 39) inches and 19.6 (clone 35) to 7.0 (clone 42) feet for age 9 diameter and height, respectively (table 3). Lastly, the TN taxon exhibited clones with a range in values for age 9 diameter and height from 5.6 (clone 8) to 3.0 (clone 10) and 27.1 (clone 8) to 13.0 (clone 10) feet, respectively (table 3).

Volume at age 9 followed along the same trends as shown by the diameters at age 9, with the TD taxon being significantly different than the other four taxa. The mean age 9 volume for the TD, TM, TN, TDD, and TMM taxa were 1.77, 0.82, 0.86, 0.63, and 0.32 cubic feet, respectively. Clones among all five taxa ranged from a high of 4.34 cubic feet for clone 24 in the TD taxon to a low of 0.11 cubic feet for clone 42 in the TMM taxon.

The crown rating scheme that was used at age 2 to provide a sense of disease incidence among the five taxa showed all of the taxa, except the TN taxon, averaged the lowest possible score (crown score of “4”). The TN taxon had the best crown score of all the taxa, yet it was among the shortest in mean height at age 2. The age 9 disease rating scheme among the five taxa showed that the TD and TDD taxa were the most resistant to Septoria, while the TM, TMM, and TN taxa were extremely susceptible. This became even more evident when looking at the range of disease scoring within each taxon. The TD and TDD taxa exhibited clones across the range of values while the TM, TMM, and the TN taxa only exhibited values representative of the severest disease rating.

Clonal performance within taxa indicated that selection should be confined to only the TD taxon. Clones 24 and 25 of the TD taxon were the top performing individuals at age 9. Clone 24 ranked first for every age 9 trait, with 94 percent survival, d.b.h. of 7.2 inches, 37.4 feet in total height, 4.34 cubic feet of volume, and a 2.3 disease rating. Clone 25 performed similarly, exhibiting 94 percent, 6.0 inches, 35.6 feet, 2.89 cubic feet, and 2.7 for survival, d.b.h., total height, volume, and disease rating, respectively. Clone 16 was the top performing clone within the TDD taxon exhibiting age 9 survival, d.b.h., total height, volume, and disease rating of

Table 3—Identification of surviving hybrid poplar clones in the 1999 taxon study located in Livingston County, KY, by taxa and clone number along with the respective age 9 mean performance for survival, d.b.h., total height, volume, and disease resistance

Taxa	Clone	Survival	D.b.h.	Height	Volume	Disease index
		percent	*inches*	*feet*	*cubic feet*	
TN	3	37.5	4.3	16.5	0.73	4.00
TN	4	68.8	3.9	21.1	0.77	4.00
TN	5	31.3	4.2	20.3	0.85	4.00
TN	6	6.3	4.5	17.6	0.85	4.00
TN	7	37.5	3.9	14.8	0.56	4.00
TN	8	18.8	5.6	27.1	1.94	3.67
TN	10	6.3	3.0	13.0	0.32	4.00
TDD	14	62.5	3.1	22.6	0.54	2.70
TDD	15	50.0	3.8	22.4	0.77	1.50
TDD	16	68.8	4.9	24.0	1.33	3.27
TDD	18	6.3	2.6	12.5	0.25	4.00
TDD	21	18.8	2.5	15.4	0.27	3.67
TD	24	93.8	7.2	37.4	4.34	2.20
TD	25	93.8	6.0	35.6	2.89	2.53
TD	26	50.0	6.2	34.0	2.95	3.50
TD	27	25.0	3.7	25.3	0.83	3.75
TD	28	81.3	3.8	20.7	0.72	4.00
TD	29	6.3	3.6	23.6	0.74	2.00
TD	30	31.3	4.5	25.8	1.21	3.00
TD	31	50.0	2.9	20.6	0.44	3.00
TMM	34	12.5	3.6	17.0	0.55	4.00
TMM	35	31.3	3.6	19.6	0.62	4.00
TMM	39	6.3	1.6	12.7	0.13	4.00
TMM	40	6.3	2.0	14.3	0.19	4.00
TMM	42	31.3	1.8	7.0	0.11	4.00
TM	44	6.3	5.0	19.6	1.14	3.00
TM	45	6.3	5.6	23.4	1.68	4.00
TM	46	37.5	3.3	18.3	0.50	4.00
TM	47	56.3	5.2	28.0	1.73	3.78
TM	48	12.5	1.8	13.2	0.15	4.00
TM	49	6.3	2.3	11.3	0.19	4.00
TM	50	25.0	4.6	24.1	1.19	4.00
TM	51	25.0	3.8	13.2	0.48	4.00
TM	52	18.8	2.6	15.1	0.29	4.00

TN = *P. trichocarpa* x *P. nigra*; TDD = TD x *P. deltoides*; TD = *P. trichocarpa* x *P. deltoides*; TMM = TM x *P. maximowiczii*; TM = *P. trichocarpa* x *P. maximowiczii*.

69 percent, 4.9 inches, 24.0 feet, 1.33 cubic feet, and 3.3, respectively. Within the TM taxon clone 47 was the best performer at age 9 with 56 percent survival, d.b.h. of 5.2 inches, 28 feet in height, 1.73 cubic feet of volume, and a disease rating of 3.8. Clone 8 was the top performing clone within the TN taxon, exhibiting 19 percent survival, d.b.h. of 5.6 inches, total height of 27.1 feet, 1.94 cubic feet of volume, and a disease rating of 3.7. The best clone within the TMM taxon was clone 35 which had 31 percent survival, d.b.h. of 3.6 inches, total height of 19.6 feet, volume of 0.62 cubic feet, and a disease rating of 4.0.

DISCUSSION

This test was designed to evaluate the potential of various hybrid poplars based upon their parentage when established on upland sites. The parentage bias toward black cottonwood was simply due to the fact that this material was available and had never been tested in the Midsouth area. In addition, the inclusion of the eastern cottonwood and Japanese poplar germplasm into the hybrid makeup also provided incentive for screening in hopes of finding individuals that would combine hybrid vigor or growth and disease resistance. This was especially true of the backcross TDD taxon, since eastern cottonwood is resistant to Septoria.

As expected, early age performance, in this case age 2, did not adequately predict future performance of the taxa or clones especially in relationship to the Septoria disease resistance. Although there was variability within all taxa for age 2 height, TM was the top performer, with 5 of the top 10 tallest clones. In comparison, the TD taxon had only 2 of the top 10 clones. However, by age 9 this trend was nearly reversed, with 6 of the top 10 tallest clones coming from the TD taxon and only 2 of the tallest clones from the TM taxon. In fact, the top three tallest clones were all from the TD taxon. Surviving clones within the TM, TN, and TMM taxa showed very little resistance to Septoria canker based upon the age 9 disease index. This suggests that within the Midsouth area of the United States, these types of hybrid clones should not be included in plantation culture. Newcombe and Ostry (2001) stated that TD F_1 hybrids were uniformly susceptible to Septoria stem canker and that up to one-half of the backcross TDD taxon would be susceptible. Clones 24, 25, and 29 all of which are in the TD taxon showed good Septoria resistance with disease indices of 2.20, 2.53, and 2.00, respectively, indicating that the disease is limited to only limbs. It has been our experience that the disease seems to affect the limbs first and then move into the stem. If indeed this continues to be the norm, then we would expect none of the clones capable of reaching the size needed for traditional harvest methods. However, if the disease remains limited to the limbs, it may be feasible that a few clones could reach harvest size. Interestingly, clones 24 and 25 share the same male and female parentage, while clone 29 shared the same female parent. The TDD taxon showed similar variability in the disease index rating. It had the only one clone (clone 15) that was nearly free of disease, but this suggests that similar Septoria resistance might be found within this taxon.

Although the test indicated that resistance to Septoria canker disease can be found within the hybrid poplar taxa tested, only two clones, i.e., clones 24 and 25, exhibited survival, growth, and disease resistance sufficient enough to warrant inclusion into an upland plantation program. Even clone 24, which was the fastest growing clone within the test averaged only 4.2 feet and 0.8 inches in diameter per year. While eastern cottonwood was not included in the study, clonal material was established adjacent to the test site. This material showed excellent survival and growth rates superior to the hybrid poplar taxa tested. Development of a *Populus* hybrid that exhibits rapid growth rates and disease resistance is still necessary for plantation culture on upland sites, especially in the arena of biomass production.

LITERATURE CITED

Dickmann, D.I.; Isebrands, J.G.; Eckenwalder, J.E.; Richardson, J, eds. 2001. Poplar culture in North America. Ontario, Canada: NRC Research Press. 397 p.

Krinard, R.M. 1985. Cottonwood development through 19 years in a Nelder's design. Res. Note SO-322. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 4 p.

Krinard, R.M. 1988. Volume equations for plantation cottonwood trees (*Populus deltoides*). Res. Note SO-347. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 8 p.

Krinard, R.M.; Johnson, R.L. 1984. Cottonwood plantation growth through 20 years. Res. Pap. SO-212. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station.11 p.

Newcombe, G.; Ostry, M. 2001. Recessive resistance to Septoria stem canker of hybrid poplar. Phytopathology. 91: 1081–1084.

Ostry, M.E.; Wilson, L.F.; McNabb, H.S., Jr.; Moore, L.M. 1989. A guide to insect, disease, and animal pests of poplars. Agric. Handb. 677. [Washington, DC:] U.S. Department of Agriculture Forest Service. 118 p.

Rainwater, D. 1999. Short rotation eastern cottonwood (*Populus deltoides*) in southeastern Missouri. In: Hartsough, B.R., comp. Proceedings of the short-rotation woody crops operations working group: 17–18. http://www.woodycrops.org/NR/rdonlyres/BF9B2067-FDB0-49B0-9543-8EEA03A415FD/1654/98proceedings.pdf. [Date accessed: July 13, 2011].

Rousseau, R.J. 1987. Genetic variance among selected cottonwood clones: Seven-year results. In: Proceedings of the fifth north central tree improvement conference. Fargo, ND: North Dakota State University: 34–42.

A COMPARISON OF TREE SHELTERS INSTALLED ON GREEN ASH AND CHERRYBARK OAK SEEDLINGS IN ARKANSAS

H. Christoph Stuhlinger[1]

Abstract—Tree shelters can aid hardwood seedling establishment by improving early seedling survival and growth. This study was established in Arkansas to compare three types of tree shelters installed on green ash (*Fraxinus pennsylvanica* Marsh.) and cherrybark oak (*Quercus pagoda* Raf.) seedlings. Seedlings planted in 4 feet tall Blue-X®, Protex® or Tubex® tree shelters were compared to unsheltered controls with respect to survival, browse damage, emergence, groundline diameter, and height. Tubex® shelters cost about twice as much to purchase and establish as the Blue-X® shelters, with the Protex® cost in between. Tree shelters did not affect survival. Diameter growth varied by site and shelter treatment. Height growth and emergence rates were greater for sheltered seedlings than unsheltered seedlings, but shelter type made little difference. Overall growth differed between sites, but sheltered cherrybark oaks grew slightly taller than sheltered green ash seedlings at both sites. Less costly shelters may provide the same growth benefits as more expensive shelters.

INTRODUCTION

The planting of hardwood tree seedlings is becoming more popular in the Southern United States. Cost-share programs such as Conservation Reserve Program and Environmental Quality Incentives Program are used by farmers and nonindustrial forest landowners to convert marginal farmlands and other nonforested lands to trees. Planting hardwood trees offers more opportunities for tree species diversity, adaptability to wetter sites, and more benefits for wildlife. However, hardwood seedlings can be more difficult to establish than pine seedlings. Slower initial growth, weed competition, and herbivory often present challenges for hardwood seedlings during the first several years after planting. Tree shelters have been found to aid hardwood seedling establishment in other regions of the United States (Minter and others 1992, West and others 1999). Faster growth, increased survival, and protection from animal browse are the primary benefits of tree shelters.

Tree shelters were first developed in England during the late 1970s (Potter 1991). Tree shelters usually consist of a translucent plastic tube, about 4 inches in diameter, and from 1 to 6 feet tall. The shelters are installed at planting time, and act like a minigreenhouse with favorable light and humidity conditions (Potter 1991).

This 5-year study was implemented by the Arkansas Forest Resource Center, a Center of Excellence within the University of Arkansas (UA) system. The purpose was to compare the performance of three types of tree shelters on green ash (*Fraxinus pennsylvanica* Marsh.) and cherrybark oak (*Quercus pagoda* Raf.) seedlings grown at two sites in Arkansas. This included comparing shelter costs and establishment times and effects on seedling survival and growth. If the shelters produced similar results, the less expensive shelter might be a more cost-effective investment for landowners than the more expensive shelters.

METHODS

Site Location and Description

Two study sites in Arkansas were used for this study. One site was at the UA Southwest Research and Extension Center near Hope in Hempstead County (Hope study site). This field was previously in hay production, and the soil is a Una silty clay loam. The other site was at the Pine Tree Branch Experiment Station near Forrest City in St. Francis County (Pine Tree site). This field was previously in row crop production, and the soils are Calloway, Loring, and Zachary silt loams. Each site occupied about 2.5 acres.

Study Design and Layout

Seedlings were planted in a randomized complete block design. Four treatments (three shelter types and a control without shelters) were applied to green ash and cherrybark oak seedlings. Each treatment-species plot consisted of 20 seedlings and each plot was replicated 4 times. This resulted in a total of 32 plots containing a total of 640 seedlings at each site.

Materials

Three types of tree shelters were used for this study—Blue-X®, Protex®, and Tubex®. All shelters were about 48 inches tall. The Blue-X® shelter consists of two pieces—an inner poly film is rolled into a cylinder and slipped into a thin plastic tube. The Protex® shelter is shipped flat and must also be rolled into a cylinder. Nine tabs along one edge must be inserted into nine slots along the other edge to maintain the tubular shape. This shelter has prepunched holes for inserting a stake tie. The Tubex® shelter is shipped fully assembled with a stake tie already inserted. Five shelters of slightly smaller diameters are nested inside each other for shipping. Tubex® shelters have a seam along the length of the tube to allow growing trees to break the tube apart. The Blue-X® and Protex® shelters are blue in color, and the Tubex® shelter is green. Bird netting was installed on the top of each shelter

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to prevent birds from falling into the shelters. The netting was removed as each seedling emerged from the tube to prevent damage to the terminal. Four-foot bamboo stakes were used to support each shelter initially. Later, oak stakes were used which lasted longer in the ground.

Study Establishment

The Hope site was disked twice prior to planting, and the Pine Tree site was ripped twice during the fall before planting. The seedlings were handplanted at a 12- by 12-foot spacing in February 2004. Tree shelters were installed immediately after planting. The assembly times were timed for each shelter type, and the installation times required to place each shelter over the planted seedling were recorded. Each site was mowed between tree rows once or twice per growing season to ease access for data collection and for landowner field days.

Field measurements collected during each growing season were survival, browse damage, and emergence. Emergence occurred as each terminal bud broke the plane of the top of each shelter, or when the terminal reached a height of 47 inches for the control seedlings. At least 1 inch of each shelter was buried in the ground to prevent air drafts through the tube and to keep mice out. Total height and basal diameter were measured at the end of each growing season.

RESULTS

Establishment Times and Costs

Establishment times (assembly plus installation) for each tree shelter type at the two study sites are presented in table 1. Assembly times were highest for the Protex® shelters, and zero for Tubex® because no assembly was necessary. Installation times included the time needed to drive a stake next to each seedling, place a shelter over a seedling, attach the shelter to the stake, and place bird netting on top of the shelter. Total establishment times were also highest for Protex® and lowest for Tubex®. Shelters were installed first at Hope and then at Pine Tree, so experience and practice led to shorter times at Hope.

Establishment costs in 2004 are presented in table 2. Establishment times were converted to labor costs and added to materials costs. Total establishment costs for Tubex® were more than twice that of Blue-X® ($1.26 per shelter).

Survival and Browse

Table 3 presents survival and browse percentages after 5 years. There were no significant differences for survival among the four treatments through year 4. A slight difference showed up for the Protex® shelters and the controls at Pine Tree in year 5. In general, shelters did not affect survival in this study. The lower survival rates for cherrybark oak at Hope may have been a function of soil factors, i.e., heavy clay soils. Other studies have shown increased survival for seedlings grown in shelters. Schweitzer and others (1999) reported that shelters improved survival, but Minter and others (1992) found no improved survival.

Green ash seedlings without shelters were heavily browsed, mainly by deer. Cherrybark oak seedlings seem to be less preferred by deer. Deer browse probably contributed to stunted seedling growth but not to mortality. Some seedlings in shelters were browsed after emergence. Taller shelters would have reduced this. An informal comparison of height growth between browsed and unbrowsed green ash at Hope showed that the mean height growth on browsed seedlings was about 3 feet less than for unbrowsed seedlings. However, the population of unbrowsed seedlings was very small.

Diameter Growth

At Hope, there were no significant differences in diameter growth among the four treatments after 5 years for green ash, but there were slight differences for cherrybark oak (fig. 1). Green ash diameters were smaller than cherrybark oak diameters. Blue-X® seedling diameters at Hope were slightly smaller than for the other treatments. Diameter growth for both species was much less the first 2 years than in subsequent years. Tree shelters caused increased seedling height growth until emergence, then normal diameter growth resumed.

Table 1—Establishment times for tree shelters by site and shelter type

Site	Shelter type	Assembly time	Installation time	Total establishment time per shelter
		minutes		
Pine tree	Blue-X	0.92 a	0.94 a	1.86 a
	Protex	1.15 b	1.33 b	2.48 b
	Tubex	0.0 c	1.08 a	1.08 c
Hope	Blue-X	0.51 a	0.80 a	1.31 a
	Protex	1.03 b	1.19 b	2.22 b
	Tubex	0.0 c	0.95 c	0.95 c

Within each site-time group, values followed by different letters indicate significant differences at $\alpha = 0.05$.

Table 2—Purchase and establishment costs for each tree shelter type in 2004

Cost type	Shelter type		
	Blue-X	Protex	Tubex
	---------- *dollars* ----------		
Tree shelter cost	0.89	1.74	2.40
Bird netting	0.05	0.05	0.00
Bamboo stake	0.11	0.11	0.11
Cable tie	0.00	0.06	0.00
Total purchase cost	1.05	1.96	2.51
Assembly cost[a]	0.09	0.15	0.00
Installation cost[a]	0.12	0.17	0.13
Total established cost	1.26	2.28	2.64

[a] Costs based on $8.00 per hour labor cost.

Table 3—Survival and browse by site, species, and treatment

Site	Treatment	Survival		Browsed	
		Ash	Oak	Ash	Oak
		---------- *percent* ----------			
Hope	Blue-X	96 a	74 a	4	1
	Protex	96 a	76 a	1	2
	Tubex	98 a	74 a	2	1
	Control	98 a	83 a	74	15
Pine tree	Blue-X	96 a	90 a	25	5
	Protex	91 ab	96 a	19	7
	Tubex	98 a	95 a	27	0
	Control	85 b	91 a	94	26

Within each site-time group, values followed by different letters indicate significant differences at $\alpha = 0.05$.

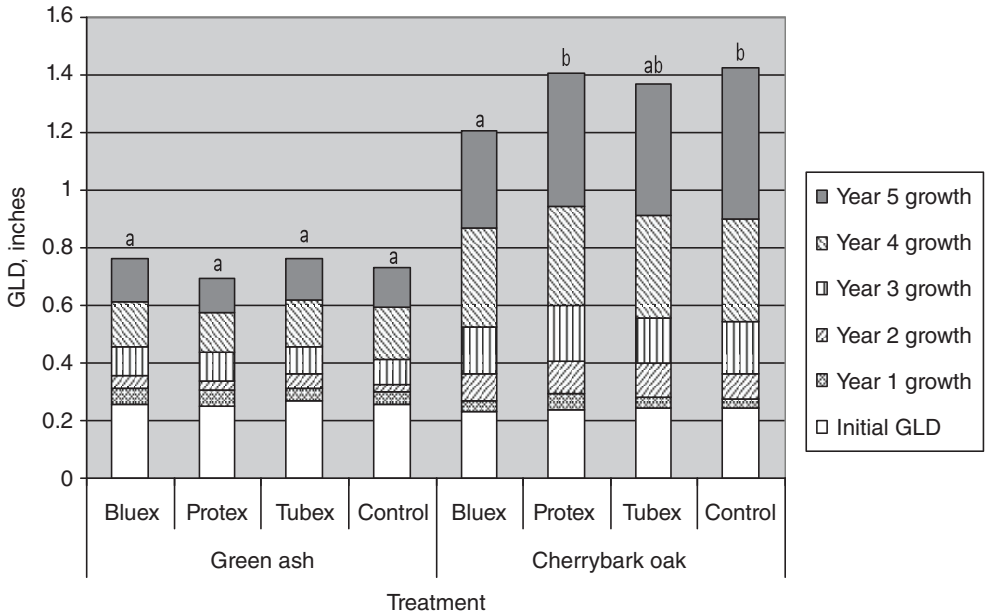

Figure 1—Groundline diameter (GLD) growth at Hope study site. Within each treatment, bars topped by different letters indicate significantly different 5-year growth means at $\alpha = 0.05$.

At Pine Tree, similar trends occurred between the two species, but controls of both species had larger diameters (fig. 2). Perhaps the diameter growth of the sheltered seedlings was still suppressed by the height growth.

Height Growth

At Hope, height growth was higher for all sheltered seedlings compared to control seedlings (fig. 3). Oak seedlings grew taller than the ash seedlings. At Pine Tree (fig. 4), height growth for sheltered seedlings was very fast until emergence, then height growth slowed down. Sheltered ash seedlings grew taller than the controls, but the opposite was true for the oaks. Shelters did not improve height growth over the controls at Pine Tree. Tree shelters generally increased height growth at both sites, but this may be more pronounced only during the early years of growth (Ponder 2003). One advantage of early increased height growth may be to quickly outgrow weed competition.

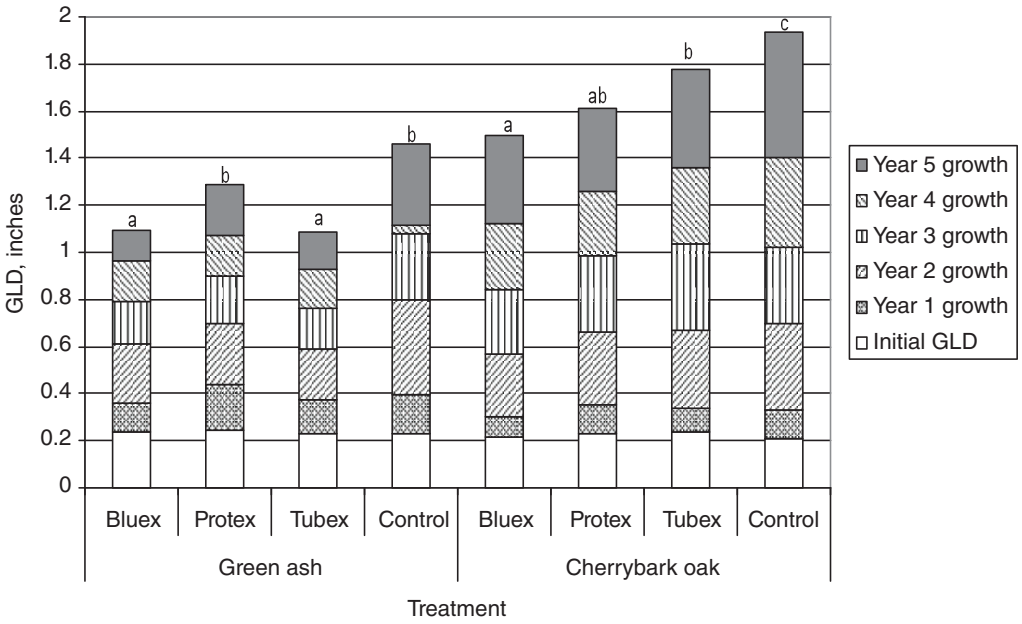

Figure 2—Groundline diameter (GLD) growth at Pine Tree study site. Within each treatment, bars topped by different letters indicate significantly different 5-year growth means at $\alpha = 0.05$.

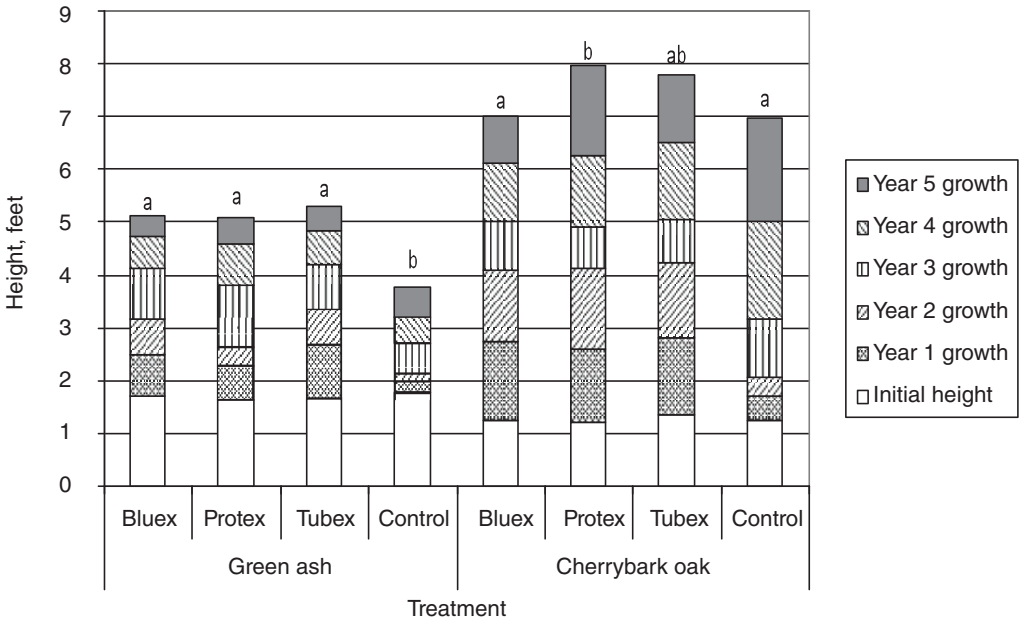

Figure 3—Height growth at Hope study site. Within each treatment, bars topped by different letters indicate significantly different 5-year growth means at $\alpha = 0.05$.

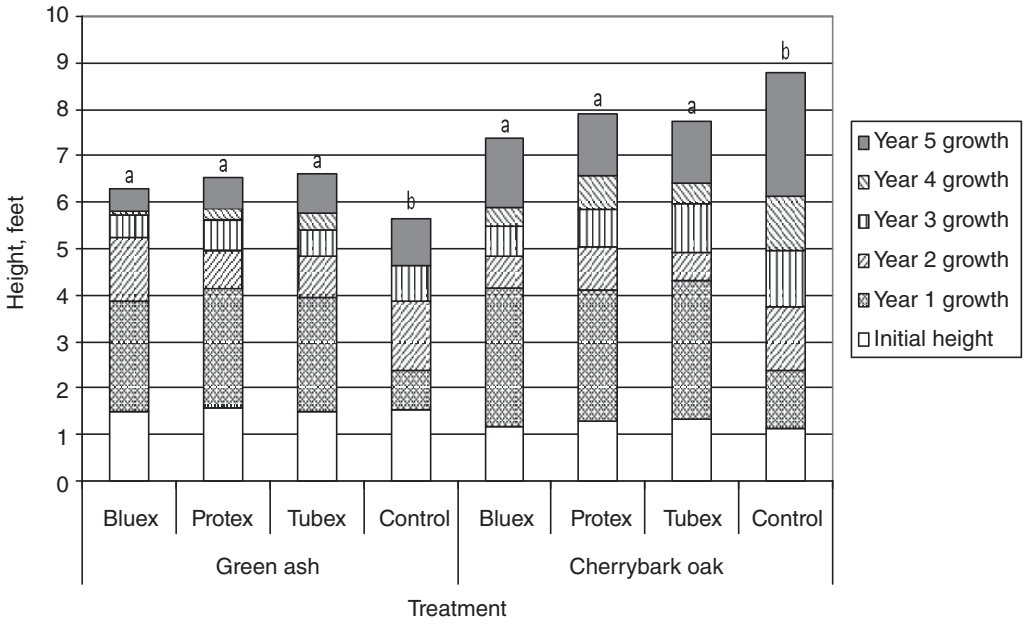

Figure 4—Height growth at Pine Tree study site. Within each treatment, bars topped by different letters indicate significantly different 5-year growth means at $\alpha = 0.05$.

Emergence

The percentages of seedlings that emerged after 5 years are presented in table 4. At Hope, emergence for sheltered ash seedlings was about twice that of controls. There were no large differences for the oaks. At Pine Tree, almost all of the sheltered ash seedlings emerged. Figures 5 and 6 show the cumulative percent emerged by study site for each treatment combination. At Hope, several sheltered seedlings emerged each year, usually during the first few months of each growing season. The controls lagged behind for the green ash, but the unsheltered oaks caught up with the sheltered seedlings during the fifth year. At Pine Tree, most of the sheltered seedlings emerged during the first 2 years. The controls emerged more slowly at first, but their height growth and emergence increased during the last 2 years. The quick emergence at Pine Tree for the sheltered seedlings again shows how the tree shelters force height growth, which is desirable for outgrowing weed competition and herbivory.

Table 4 also presents the emergence rate, or height growth rate, in feet per month, for seedlings inside their shelters. Growth rates at Hope were moderate, and slightly higher for cherrybark oaks. At Pine Tree, sheltered seedlings grew an average of almost 6 inches per month until emergence, which is very fast. At both sites, growth rates of sheltered seedlings were significantly higher than for controls.

Table 4—Percentage of seedlings emerged and mean emergence growth rate by site, treatment, and species

		Emerged		Emerge rate	
Site	Treatment	Ash	Oak	Ash	Oak
		----*percent* ----		*foot per month*	
Hope	Blue-X	89	73	0.17 a	0.26 a
	Protex	86	81	0.13 b	0.25 a
	Tubex	89	75	0.19 a	0.26 a
	Control	48	79	0.08 c	0.11 b
Pine tree	Blue-X	100	98	0.48 a	0.48 a
	Protex	99	98	0.46 a	0.42 b
	Tubex	100	98	0.48 a	0.53 c
	Control	75	90	0.19 b	0.19 d

Within each site-species group, emergence rate means followed by different letters are significantly different at $\alpha = 0.05$.

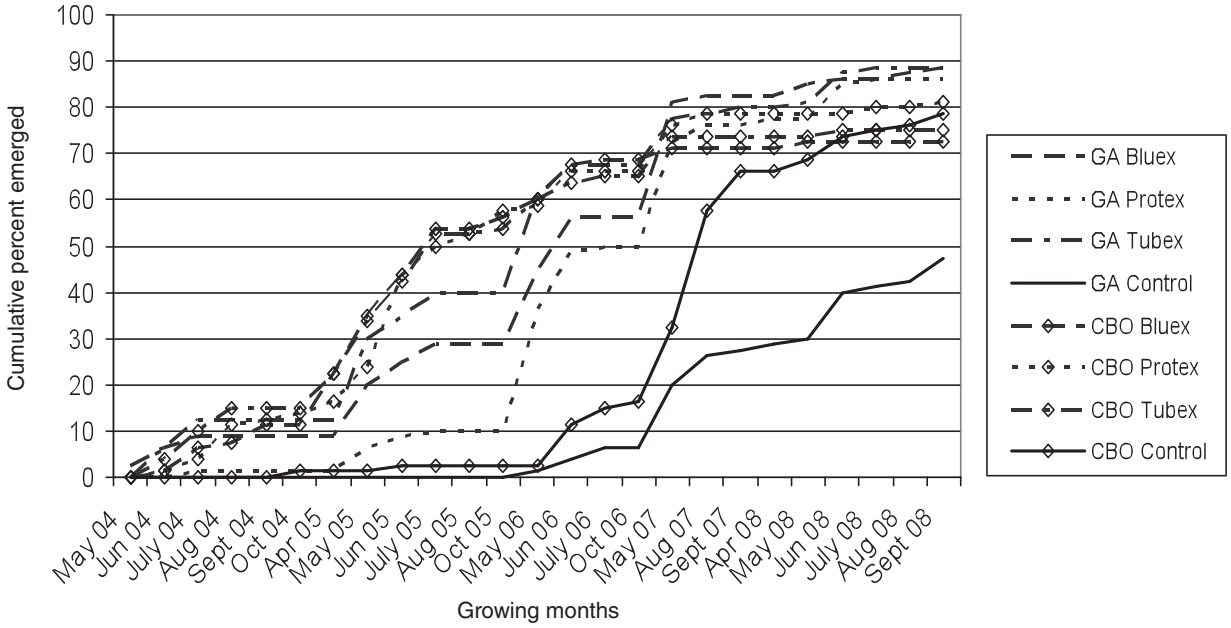

Figure 5—Cumulative percentage of seedlings emerged at Hope study site over 5 years.

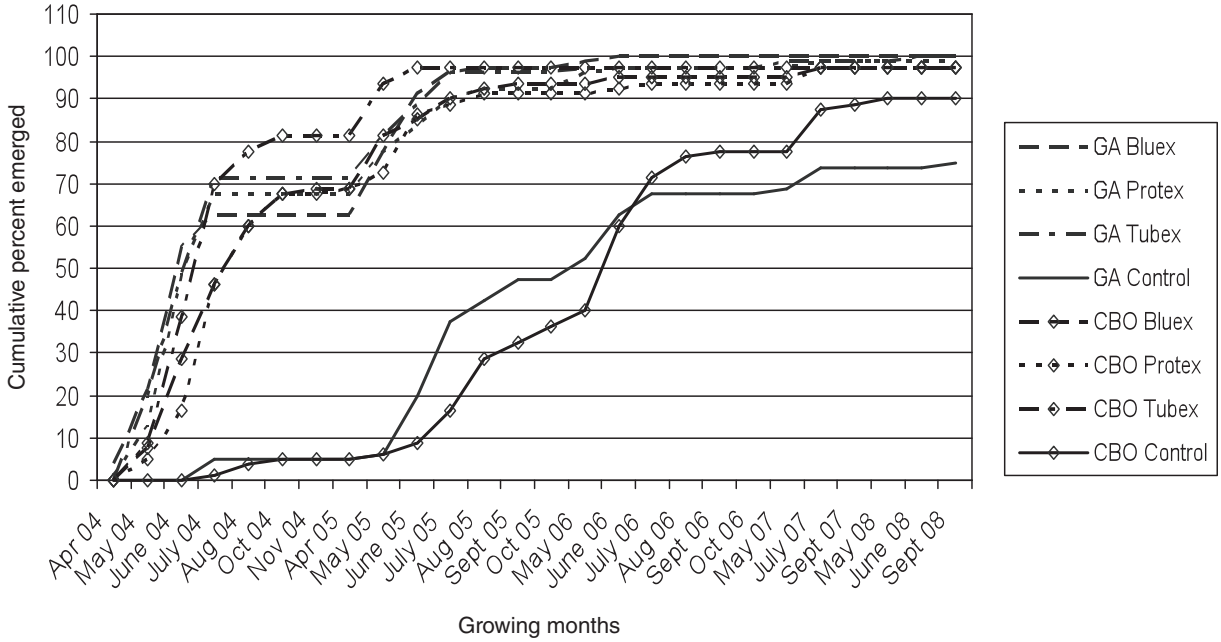

Figure 6—Cumulative percentage of seedlings emerged at Pine Tree study site over 5 years.

CONCLUSIONS

Results of this 5-year study show that Tubex® shelters are the quickest to install but also the most expensive to purchase. Site, not tree shelters, affected survival. Also, the cherrybark oaks at Hope had 15 to 20 percent lower survival than the green ash, probably due to the oaks being less adapted to the heavier soils. Not ripping the Hope site may have also made a difference. Unsheltered green ash seedlings suffered from heavy deer browse. Tree shelters significantly increased height growth for every treatment combination except the oaks at Pine Tree. There were slight growth differences among the shelter types. Even so, Blue-X®, which is quite a bit less expensive than the Tubex®, can produce similar growth advantages. Blue-X® tree shelters might be a cost-effective compromise for landowners. In this study, cherrybark oak seedlings did not benefit significantly from tree shelters, but the green ash did. The advantages provided by tree shelters appear to be somewhat species and site specific.

LITERATURE CITED

Minter, W.F.; Myers, R.K.; Fischer, B.C. 1992. Effects of tree shelters on northern red oak seedlings planted in harvested forest openings. Northern Journal of Applied Forestry. 9: 58–63.

Ponder, F., Jr. 2003. Ten-year results of tree shelters on survival and growth of planted hardwoods. Northern Journal of Applied Forestry. 20(3): 104–108.

Potter, M.J. 1991. Treeshelters. Handb. 7. Forestry Commission. London: Her Majesty's Stationery Office. 48 p.

Schweitzer, C.J.; Gardiner, E.S.; Stanturf, J.A.; Ezell, A.W. 1999. Methods to improve establishment and growth of bottomland hardwood artificial regeneration. In: Stringer, J.W.; Loftis, D.L., eds. Proceedings of the 12th central hardwood forest conference. Gen. Tech. Rep. SRS-24. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 209–214.

West, D.H.; Chappelka, A.H.; Tilt, K.M. [and others]. 1999. Effect of tree shelters on survival, growth, and wood quality of 11 tree species commonly planted in the Southern United States. Journal of Arboriculture. 25(2): 69–75.

CARBON AND BIOENERGY

Slash incorporation and clonal pine biomass for ecosystem carbon management, Berkeley County, South Carolina (photo by D. Andrew Scott)

A TREE BIOMASS AND CARBON ESTIMATION SYSTEM

Emily B. Schultz, Thomas G. Matney, and Donald L. Grebner[1]

Abstract—Appropriate forest management decisions for the developing woody biofuel and carbon credit markets require inventory and growth-and-yield systems reporting component tree dry weight biomass estimates. We have developed an integrated growth-and-yield and biomass/carbon calculator. The objective was to provide Mississippi's State inventory system with bioenergy economic development tools and forest landowners trying to qualify for carbon credit programs with carbon estimates. Biomass equations were collected from many sources in the literature and organized into a uniform software system that provides estimates of volume/biomass/carbon of branches, foliage, and bole. Users may select biomass components for 347 species/region combinations and obtain information on (1) tree profiles, (2) biomass/carbon tables, and (3) model forms and coefficients. Bole biomass and volume can be obtained to any top diameter limit. Pine (*Pinus* spp.) plantation and hardwood stand case studies demonstrate biomass and carbon estimation and pine projections. Alternative financial analyses generated by the growth-and-yield system provide information on which forest landowners may base management decisions for the biofuel and carbon markets if they provide current local stumpage prices.

INTRODUCTION

Chicago Climate Exchange (CCX) members may mitigate carbon dioxide (CO_2) emissions by purchasing carbon offsets generated by the reduction, avoidance, or sequestration of CO_2 (Grebner and others 2007, Rousseau 2008). Protocols for CO_2 emission offsetting using afforestation and managed forest activities have been defined and present a potential benefit to forest landowners trading in the carbon market. CCX protocols include reliable inventories, regional and species-specific carbon sequestration estimates, approved growth-and-yield models, and verification procedures. Elemental carbon (C) is fixed in various parts of a tree such as wood and bark during photosynthesis. The CO_2 equivalent produced by a tree is based on the premise that the oven-dried weight of the bole bark and wood is approximately one-half elemental C. In general, softwoods have more lignin and less hemicellulose than hardwoods and, consequently, softwoods have a higher C content per unit weight. C content also varies by tree species; however, one-half is the accepted conversion percentage for all tree species (Birdsey 1992). The weight ratio of CO_2 to elemental C is 3.67. Therefore, the CO_2 equivalent for the bole wood and bark of a tree is calculated as:

$$3.67\frac{ODWt}{2} \qquad (1)$$

where

ODWt = oven-dried weight of bole wood and bark

A carbon credit is defined as 1 t of CO_2 removed from the atmosphere and, in 2008, CCX C credit offsets traded between $7.40 and $0.95. Forestry offset projects trade-in units of at least 12 500 t/year, but small forest landowners can collectively trade through registered organizations called offset aggregators (Grebner and others 2007, Rousseau 2008).

Biomass and C estimates are needed to support the use of the newly developing C credit exchange and biofuel markets. Decision-making tools are needed by forest landowners trying to qualify for C credit programs and bioenergy industries using Mississippi Institute for Forest Inventory data or other sources to assess the risk on investments in new facilities. Upgrades are necessary for volume calculators, inventory systems, and growth-and-yield models. The objectives of this paper are to: (1) present a comprehensive biomass and C estimation system that can be integrated with traditional volume estimation and prediction models and (2) demonstrate the use of this system by comparing total sequestered bole biomass and C for a cutover site-prepared loblolly pine (*Pinus taeda*) plantation and a natural red oak (*Quercus* section *Lobatae*)-sweetgum (*Liquidambar styraciflua*) bottomland hardwood stand. Although not presented in this paper, the accompanying software also includes updated growth-and-yield models for slash (*P. elliottii*) and longleaf pine (*P. palustris*).

METHODS

The first step in developing the biomass/C estimation system was an exhaustive search of the literature for cubic volume and weight prediction equation systems for all major tree components for Gulf South and Southeastern United States tree species. Other regions of the United States and minor species were included when found. Many of the biomass systems in the literature were for bole wood only, but 41 were complete biomass systems that included at least bole, branches, and foliage. The core biomass/C estimation program was created as a Microsoft Windows® Dynamic Link Library (dll). A dll is a software program module that can be linked to other programs, e.g., Visual Basic, Excel, and FORTRAN. This dll estimates C, biomass by tree component, dry and green weights, and volume in all the conventional units.

The biomass/C dll was incorporated into an existing tree volume/weight table and equation generator called Tree Volume and Weight Calculator (TVolWt). This computer

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program now contains a total of 347 prediction systems. The software interface provides the user with an option of reconciling one of the complete biomass systems with any bole-only profile function. The reconciliation process multiplicatively adjusts volumes and weights of branches and roots by the ratio of total bole volume from the profile system to total bole volume from the complete system. Justification of the procedure is based on the observation that the ratio of the volume of a biomass component to the total bole biomass volume is nearly a constant for all components. If a bole wood only profile equation is reconciled with a biomass system of a similar species, the ratios between total bole volume from the profile equation to total bole volume from the biomass system and biomass component volumes to total biomass bole volume should hold. If the user decides not to reconcile the profile and biomass prediction systems, separate volumes and weights are given. Biomass and C estimates for bark can be obtained by subtracting outside bark (o.b.) volumes/weights from inside bark volumes/weights.

Modifications were made to the TVolWt problem definition screen (fig. 1) to allow biomass equation selection and reconciliation with profile functions if the "Reconcile Biomass System With Profile Function" box is checked. One of three modes is selected from the "Operation" drop-down box to produce standard or local volumes and weight equations and tables or single-tree volumes and weights. Profile functions and biomass systems are chosen from lists. Sources and citations are documented in files that accompany the software. If the user selects "Local Volume Equations," sufficient diameter at breast height (d.b.h.)-merchantable height (HM) pairs (20 or more) must be entered in the interface spreadsheet to predict HM and form class (FC) equations to calculate local volumes (volumes that are a function of d.b.h. alone). Improved estimates of standard equations can also be obtained by providing d.b.h.-HM pairs. These data are used internally to develop guide curves for preparing equations. FC data is only required for those profile functions that involve the FC variable. Possible model forms, fitted to approximate standard and local volume or weight tables, HM, and FC equations, are printed at the end of each problem "Run." Volume tables are printed by checking the "Print Volume Tables" box and can be customized by specifying ranges and increments for d.b.h., HM, and FC. Volume and weight table entries are calculated directly from the profile function or biomass system and not from the fitted volume and weight equations. The tables are exact, and the equations, that only approximate the table entries, are given for the convenience of the user.

Growth-and-yield simulators for cutover site-prepared loblolly pine (Matney and Farrar 1992), cutover site-prepared slash pine (Zarnoch and others 1991), natural longleaf pine (Farrar and Matney 1994), and red oak-sweetgum bottomland hardwood stands (Iles 2008) were also modified to call the biomass/C dll. The red oak-sweetgum bottomland hardwood growth-and-yield model was obtained from a recently completed study in Mississippi funded by the U.S. Forest Service, Center for Bottomland Hardwoods Research. These models allow landowners to explore different management scenarios in comparing the financial benefits of biofuel, C, and traditional markets.

RESULTS

Tree Volume and Weight Calculator (TVolWt)

The TVolWt problem definition screen in figure 1 depicts a standard volume equation operation for a reconciled white oak (*Q. alba*) Southwide profile function and Coastal Plain Gulf South biomass system. Custom inputs were specified for pulpwood and sawtimber d.b.h. thresholds, merchantable tops, volume table formatting, and "pounds per cubic foot (o.b.)." Tables were produced for 44 volume and weight units. Figures 2 and 3 represent volume and biomass system weight tables, respectively. Figure 4 is a representative section of the standard volume equation parameters and fit statistics for models 1 and 2, models that do not involve FC. Printed model forms for standard and local volumes, HM, and FC equations are given in figure 5. The oven-dried o.b. stem weights to a 0-inch stem top diameter in figure 3 are presented in pounds. To convert these figures from pounds to metric tons of CO_2 equivalents (C credits), the result of equation 1 is multiplied by 0.00045359. Oven-dried weights for small-diameter trees or branches and foliage can also be used to explore the opportunities for participating in the biofuel market that utilizes forest residuals and small diameter trees (Grebner and others 2009; Perez-Verdin and others, 2009). Stem profiles, volumes, and weights for individual trees can be obtained by selecting the "Single Tree Volume and Profile" operation on the problem definition screen. Figure 6 shows the individual tree results for a southwide loblolly pine plantation profile function reconciled with a Texas cutover loblolly pine biomass system.

Growth-and-Yield Simulators

The modified cutover site-prepared loblolly pine and red oak-sweetgum bottomland hardwood stand growth-and-yield simulators were used to calculate accumulated elemental C and CO_2 equivalents of o.b. bole and branch components (table 1) (Schultz and others 2008). For purposes of comparison, high-, medium-, and low-quality site indices were chosen appropriate to each growth-and-yield model. Stands were evaluated for sequestered C at ages that represent the point of maximum mean annual increment and not financial maturity. Only bole C can currently be used in C credit calculations. Branch and foliage biomass are considered sources for biofuels. Loblolly pine stands reached maximum mean annual increment 19 to 16 years sooner than the red oak-sweetgum stands but accumulated almost one-half less C across high-, medium-, and low-quality site indices. These results are reflective of the higher carrying capacities attained by the hardwood stands. Yields from the cutover site-prepared loblolly pine growth-and-yield simulator are approximately 15 percent lower than those from old field growth-and-yield simulators. Since C credit programs register afforestation practices, the yields from the cutover site-prepared example should be increased by 15 percent for C market applications. An old field loblolly pine growth-and-yield simulator is available from the authors upon request.

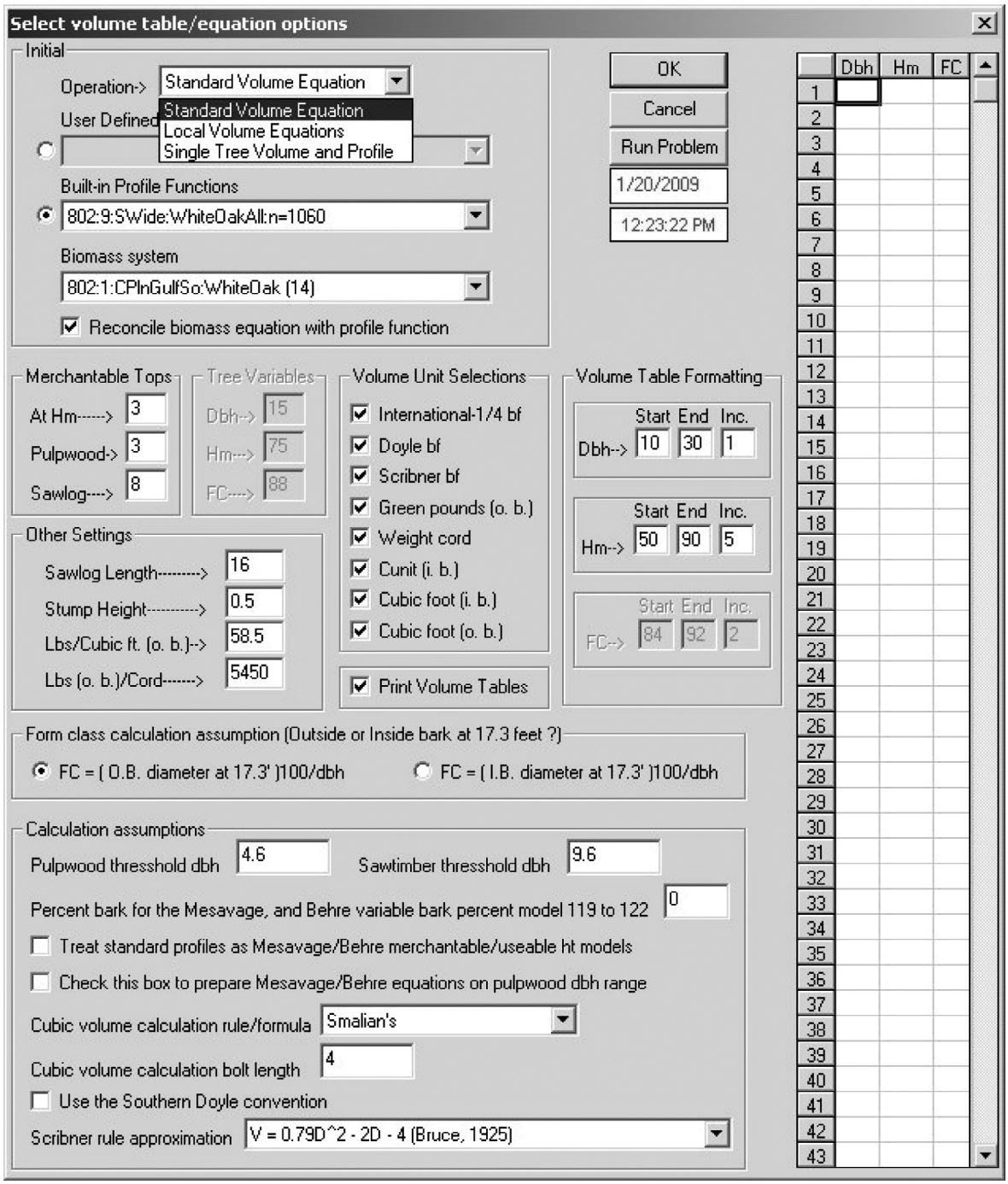

Figure 1—Tree Volume and Weight Calculator (TVolWt) program interface used for collecting user preferences for generating standard and local volume and weight tables, equations, and single-tree component volume and weights.

Annual estimates of sequestered CO_2 equivalents in mt/acre are needed to estimate expected C market revenues. The four modified growth-and-yield simulators provided the basis for comparing revenues from different management strategies, e.g. thinnings, trees per acre, site index, and markets, e.g. C, biomass, pulpwood, sawtimber. Table 2 (Schultz and others 2008) summarizes calculations made from the results of the cutover site-prepared loblolly pine simulator (Matney and Farrar 1992) for o.b. oven-dried weight of bole wood where site index is 60 (base age 25) and the number of trees per acre is 650 at age 5. Bole wood CO_2 equivalent per acre was estimated at 5-year intervals, and differences between intervals were divided by five to obtain the periodic annual bole CO_2 equivalent per acre. Expected average per-acre

Table 1. International-1/4 volumes (board feet) to a 8.0 inch sawlog top diameter (ob) limit for profile: 802:9:SWide:WhiteOakAll.

Dbh	Merchantable height -- 50--	-- 55--	-- 60--	-- 65--	-- 70--	-- 75--	-- 80--	-- 85--	-- 90--
10	40.7	44.7	48.9	53.5	58.7	63.2	68.9	74.9	81.0
11	59.0	64.8	71.1	78.8	86.2	93.1	101.2	109.7	118.5
12	77.4	86.8	94.6	104.3	114.1	123.3	133.8	144.8	156.3
13	98.2	107.7	118.8	131.0	142.3	156.5	168.7	182.4	196.6
14	120.0	131.8	145.3	159.2	173.7	189.9	205.7	221.2	241.0
15	142.5	156.2	172.3	189.0	207.3	225.1	245.3	264.8	284.1
16	166.6	184.8	201.4	222.1	241.9	264.3	285.0	310.7	333.2
17	190.9	211.9	234.8	255.1	279.6	305.4	329.3	357.6	384.5
18	216.4	240.2	266.0	291.4	318.7	348.0	376.1	408.5	437.9
19	247.6	274.7	303.0	329.6	361.1	394.2	424.9	461.2	494.3
20	275.0	306.0	337.6	369.6	404.0	440.9	475.1	515.4	558.9
21	309.1	343.0	378.4	414.3	449.5	490.3	533.3	573.0	620.3
22	344.1	381.9	414.6	457.5	500.7	541.2	588.7	638.6	684.1
23	375.2	416.3	459.2	502.4	550.1	600.0	646.2	700.6	757.8
24	413.8	459.2	506.5	554.3	606.6	655.4	711.6	771.5	826.0
25	453.3	504.1	556.1	603.1	659.1	718.6	773.5	838.1	906.4
26	487.6	541.9	597.6	658.6	720.4	777.9	844.5	915.1	979.6
27	531.2	589.1	649.6	710.8	776.7	846.1	918.5	985.7	1065.8
28	576.3	639.2	704.8	770.5	841.9	917.4	987.3	1068.9	1155.5
29	622.5	690.4	761.9	832.6	902.1	982.3	1067.2	1155.4	1236.1
30	671.0	744.2	807.1	889.4	972.4	1058.8	1150.1	1232.8	1332.0

Figure 2—Representative volume (International 1/4-inch) table output from the Tree Volume and Weight Calculator (TVolWt) program for a white oak Southwide profile function that has been reconciled with a white oak Coastal Plain Gulf South biomass system.

Table 23. Stem DWOB (lbs) to a 0.0 inch stem top diameter(ob) limit for profile: 802:9:SWide:WhiteOakAll.

Dbh	Merchantable height -- 50--	-- 55--	-- 60--	-- 65--	-- 70--	-- 75--	-- 80--	-- 85--	-- 90--
10	590.45	651.56	705.99	769.13	825.63	889.41	955.11	1022.73	1092.33
11	697.57	758.81	832.03	905.43	978.05	1052.72	1129.51	1208.42	1289.53
12	801.69	882.71	953.98	1044.49	1127.24	1212.26	1299.56	1389.19	1481.14
13	922.35	1001.31	1096.18	1190.90	1284.15	1390.68	1489.95	1591.80	1696.18
14	1049.03	1138.15	1245.20	1352.05	1457.10	1577.20	1688.84	1803.23	1937.24
15	1181.56	1280.98	1400.71	1519.98	1649.70	1771.06	1911.06	2039.88	2171.75
16	1319.25	1449.15	1562.07	1706.78	1837.76	1987.60	2126.42	2287.98	2435.23
17	1461.83	1604.92	1754.02	1888.08	2047.53	2213.93	2367.91	2547.12	2710.22
18	1609.09	1765.43	1928.38	2090.31	2266.25	2449.72	2619.30	2816.80	2996.35
19	1784.20	1957.62	2138.24	2300.30	2493.38	2694.62	2880.28	3096.61	3293.01
20	1941.83	2129.30	2324.36	2518.00	2728.61	2948.11	3150.33	3386.17	3632.19
21	2131.50	2337.29	2551.58	2764.12	2971.74	3209.93	3458.21	3684.93	3951.65
22	2329.79	2554.63	2748.29	2998.29	3248.11	3479.83	3747.93	4027.44	4280.34
23	2502.52	2742.33	2991.94	3239.40	3508.28	3788.36	4045.83	4346.40	4659.81
24	2714.62	2974.68	3245.35	3513.69	3805.44	4075.85	4388.66	4714.71	5009.27
25	2934.87	3216.05	3508.63	3769.80	4081.74	4406.49	4704.70	5052.91	5415.79
26	3120.73	3417.76	3726.47	4063.20	4399.37	4710.56	5070.89	5446.21	5784.79
27	3354.08	3673.36	4005.16	4333.76	4690.98	5062.88	5450.11	5802.40	6217.40
28	3595.28	3937.78	4293.52	4645.74	5028.65	5427.27	5792.80	6219.88	6664.77
29	3844.64	4210.90	4591.31	4968.10	5334.91	5756.10	6194.55	6651.21	7062.75
30	4101.93	4492.50	4826.32	5260.05	5691.99	6141.35	6609.16	7034.23	7535.42

Figure 3—Representative biomass component (stem oven-dried weight) table output from the Tree Volume and Weight Calculator (TVolWt) program for a white oak Southwide profile function that has been reconciled with a white oak Coastal Plain Gulf South biomass system.

Table 46. Standard volume/weight equations for profile: 802:9:SWide:WhiteOakAll.

			Parameter Estimates					Measures of Fit	
Model	Unit	TopD	a	b	c	d	e	Std.Error	FitIndex
1	cvobsm	0.0	2.265	0.0024665773				1.6061	0.9991
2	cvobsm	0.0	2.382	0.0028870821	1.96977	0.98257		0.4850	0.9999
1	cvibsm	0.0	2.103	0.0021684303				1.2158	0.9994
2	cvibsm	0.0	2.302	0.0023753107	1.97383	0.99586		0.4676	0.9999
1	gwobtsm	0.0	183.762	0.1561228391				196.5500	0.9966
2	gwobtsm	0.0	162.715	0.2431463321	1.90819	0.96123		30.2772	0.9999
1	gwibtsm	0.0	178.737	0.1381863348				180.5145	0.9964
2	gwibtsm	0.0	160.917	0.2148503692	1.89700	0.97075		29.4092	0.9999
1	dwobtsm	0.0	103.965	0.0912736342				123.9496	0.9961
2	dwobtsm	0.0	94.677	0.1385117586	1.88163	0.98916		17.8525	0.9999
1	dwibtsm	0.0	105.810	0.0797097231				123.0889	0.9948
2	dwibtsm	0.0	94.171	0.1299942146	1.85977	0.98892		16.9925	0.9999
1	cvobbran	-1.0	-0.519	0.0013729256				7.0903	0.9632
2	cvobbran	-1.0	0.321	0.0003032810	2.63155	0.86408		0.3273	0.9999
1	cvibbran	-1.0	-0.484	0.0010644915				5.9172	0.9584
2	cvibbran	-1.0	0.223	0.0002037444	2.67461	0.86478		0.2600	0.9999
1	gwobbran	-1.0	-38.018	0.0913986562				458.2929	0.9650
2	gwobbran	-1.0	11.862	0.0223332821	2.61377	0.85378		22.3967	0.9999
1	gwibbran	-1.0	-36.466	0.0716282796				387.4986	0.9602
2	gwibbran	-1.0	5.314	0.0152342704	2.65118	0.85829		18.1955	0.9999
1	dwobbran	-1.0	-26.081	0.0561738072				279.0114	0.9656
2	dwobbran	-1.0	6.866	0.0127870624	2.59283	0.88866		13.6434	0.9999
1	dwibbran	-1.0	-22.660	0.0439591260				228.2248	0.9630
2	dwibbran	-1.0	3.248	0.0094570068	2.61497	0.88504		11.0245	0.9999
1	gwfoli	-1.0	16.133	0.0092243541				20.2885	0.9903
2	gwfoli	-1.0	8.110	0.0284082380	2.19803	0.55751		1.3708	1.0000
1	dwfoli	-1.0	6.735	0.0049252883				8.2217	0.9946
2	dwfoli	-1.0	4.464	0.0096604025	2.20113	0.66668		0.8108	0.9999

Figure 4—A representative section of the standard volume/weight equation parameters and fit statistics table produced by the Tree Volume and Weight Calculator (TVolWt) program for a white oak Southwide profile function that has been reconciled with a white oak Coastal Plain Gulf South biomass system.

revenues from selling C offsets can be obtained by multiplying the periodic annual CO_2 equivalent per acre (table 2) by the CCX price per mt and summing over the contract period. Note in the loblolly pine example that maximum periodic CO_2 sequestered per acre occurs between years 20 and 25.

DISCUSSION AND SUMMARY

A comprehensive computational model was created as a dll to estimate biomass and C for the major commercial species in the Eastern and Southern United States. Limited minor species and species from other areas are also included. The utility of the dll is extensive since it can be incorporated into other programs that allow dll linkages. The number of biomass/C systems incorporated into the dll was limited by the number of published systems, but projections made from reconciling a profile function with a biomass system of a species with similar form and wood density should produce good estimates for species whose biomass estimates are not available.

A tree volume/weight table and equation generator (TVolWt) and four growth-and-yield systems were modified to call the dll and can be used to simulate scenarios and evaluate the costs and benefits of participation in the biofuel markets or C offset program. For a complete C offset financial analysis, fees and costs associated with sales, registration, aggregators, verifiers, and inventories (Rousseau 2008) must also be estimated. Example applications showed that bottomland red oak-sweetgum hardwood stands produced higher bole and branch biomass than cutover site-prepared loblolly pine and

Standard Volume Equation Model Forms

Model	Model Form
1	$v = a + b(dbh^2)(Hm)$
2	$v = a + b(dbh)^c(Hm)^d$
3	$v = a + b(dbh^2)(Hm)(FC)$
4	$v = a + b(dbh)^c(Hm)^d(FC)^e$

Local Volume Equation Model Forms

Model	Model Form
1	$v = a + b(dbh^2)$
2	$v = a + b(dbh)^{2d}$
3	$v = a + b(dbh)^2 + c(dbh)^4$
4	$v = a + b(dbh)^{2d} + c(dbh)^{4d}$

Merchantable Height Equation Models

Model	Model Form
1	$Hm = ae^{\frac{b}{dbh}}$
2	$Hm = ae^{\frac{b}{dbh^c}}$
3	$Hm = a + b\,dbh$
4	$Hm = a + b(dbh)^c$

Form Class Equation Model Forms

Model	Model Form
1	$FC = a + b\,dbh$
2	$FC = a + b(dbh)^c$

Figure 5—Model forms used in the Tree Volume and Weight Calculator (TVolWt) program for standard and local volumes, merchantable heights, and form class equations.

Table 1—Total bole and branch outside bark elemental carbon sequestered per acre and CO_2 equivalent for low-, medium-, and high-quality site index cutover site-prepared loblolly pine plantations and natural red oak-sweetgum bottomland hardwood stands [elemental carbon was estimated at ages representing the point where mean annual increment is maximized (Schultz and others 2008)]

			Elemental carbon	
Condition	Site index[a]	Age at maximum MAI	Bole (o.b.)	Branch (o.b.)
		years	---------- *mt/acre* ----------	
Cutover site-prepared loblolly pine plantation	Low (45)	30	17.65 (64.78)[b]	2.74 (10.06)[b]
	Medium (60)	27	26.57 (97.51)	1.43 (5.25)
	High (75)	25	35.17 (129.07)	1.10 (4.04)
RO-SG bottomland hardwood stand	Low (80)	49	34.68 (127.28)	7.95 (29.18)
	Medium (100)	45	48.10 (176.53)	14.41 (52.88)
	High (120)	41	59.60 (218.73)	21.42 (78.61)

MAI = mean annual increment; o.b. = outside bark; RO-SG = natural red oak-sweetgum.

[a] Site index is base age 25 for the pine plantations and base age 50 for the hardwood natural stands.

[b] CO_2 equivalent weight equals elemental carbon weight times 3.67.

Table 1. Stem diameters and volumes of a 15.0 dbh tree with a merchantable height of 75 to a 3.0 top diameter for profile: 131:9:SWide:LobPinePln.

Profile			Volume units and top diameter(ob)		
h	Dia(ob)	Dia(ib)	Unit	Volume	TopDia
0.5	16.1	15.3	International-1/4	225.1	8.0
2.5	15.4	14.6	Doyle	132.0	8.0
4.5	15.0	14.2	Scribner	190.9	8.0
8.5	14.4	13.6	Cubic feet(o.b.)	40.94	8.0
12.5	13.8	13.0	Cubic feet(i.b.)	36.39	8.0
16.5	13.3	12.5	Green pounds(o.b.)	2395.0	8.0
17.3	13.2	12.4	Weight cords	0.439	8.0
20.5	12.7	12.0	Cunits(i.b.)	0.364	8.0
24.5	12.2	11.5	Cubic feet(o.b.)	45.24	3.0
28.5	11.7	11.0	Cubic feet(i.b.)	40.20	3.0
32.5	11.1	10.4	Green pounds(o.b.)	2646.4	3.0
36.5	10.5	9.9	Weight cords	0.486	3.0
40.5	9.9	9.3	Cunits(i.b.)	0.402	3.0
44.5	9.2	8.6	Cubic feet(o.b.)	45.45	0.0
48.5	8.5	8.0	Cubic feet(i.b.)	40.39	0.0
52.5	7.8	7.3	Green pounds(o.b.)	2658.8	0.0
56.5	7.0	6.6	Weight cords	0.488	0.0
60.5	6.2	5.9	Cunits(i.b.)	0.404	0.0
64.5	5.4	5.1			
68.5	4.5	4.3			
72.5	3.6	3.4			
75.0	3.0	2.9			

Total stem outside bark cubic volume.:	45.45
Total stem inside bark cubic volume..:	40.39
Total stem outside bark green weight.:	2808.29
Total stem inside bark green weight..:	2624.83
Total stem outside bark dry weight...:	1306.24
Total stem inside bark dry weight....:	1206.28
Branch outside bark cubic volume.....:	6.30
Branch inside bark cubic volume......:	5.68
Branch outside bark green weight.....:	350.91
Branch inside bark green weight......:	340.38
Branch outside bark dry weight.......:	147.85
Branch inside bark dry weight........:	142.07
Green weight of foliage.....:	1.33
Dry weight of foliage.....:	0.66
Outside bark cubic volume to PW top..:	45.24
Inside bark cubic volume to PW top...:	40.20
Outside bark green weight to PW top..:	2785.41
Inside bark green weight to PW top...:	2609.50
Outside bark dry weight to PW top....:	1296.91
Inside bark dry weight to PW top.....:	1200.57
Outside bark cubic volume to ST top..:	40.94
Inside bark cubic volume to ST top...:	36.39
Outside bark green weight to ST top..:	2463.61
Inside bark green weight to ST top...:	2346.21
Outside bark dry weight to ST top....:	1155.60
Inside bark dry weight to ST top.....:	1088.43

Figure 6—Tree Volume and Weight Calculator (TVolWt) example stem profile, volume, and weight outputs for an individual tree using a Southwide loblolly pine plantation profile function reconciled with a Texas cutover loblolly pine biomass system.

Table 2—Total outside bark bole cubic-foot volume, elemental carbon sequestered, carbon dioxide equivalent per acre, and periodic annual sequestered carbon dioxide equivalent per acre for a site index 60 (base age 25) loblolly pine plantation with 650 trees per acre at age 5 (Schultz and others 2008)

Year	Bole o.b.	Bole elemental carbon	Bole CO_2 equivalent[a]	Periodic annual increment (5-year periods) of bole sequestered CO_2 equivalent
	cubic-foot volume per acre	----------	*mt/acre*	----------
10	518	5.14	18.86	
15	1299	9.84	36.11	3.45
20	2104	14.95	54.87	3.75
25	2844	19.99	73.36	3.70
30	3515	24.36	89.40	3.21
35	3999	27.72	101.73	2.47
40	4309	30.08	110.39	1.73

o.b. = outside bark; CO_2 = carbon dioxide.

[a] CO_2 equivalent weight equals elemental carbon weight times 3.67.

arrived at a maximum mean annual increment at an older age, indicative of the hardwoods' larger crowns and greater carrying capacity. Over 5-year intervals from age 10 to 40, the periodic annual increment for loblolly pine occurred between ages 20 and 25 depending on site index and planting density. Sequestered CO_2 per acre per year fell off sharply after age 30. Forest landowners can use the growth-and-yield simulators to determine maximum CO_2 sequestration by varying trees per acre, site index, and thinning regimes within the bounds of their overall management objectives and CCX requirements.

TVolWt, the four growth-and-yield systems, and the core dll program are available for downloading from www.timbercruise.com. Future development of the dll will allow its use in standard timber cruise software for estimating tree component volume and weights.

LITERATURE CITED

Birdsey, R.A. 1992. Carbon storage and accumulation in United States forest ecosystems. Gen. Tech. Rep. WO-59. Radnor, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 51 p.

Farrar, R.M.; Matney, T.G. 1994. A dual simulator for natural even-aged stands of longleaf pine in the South's east gulf region. Southern Journal of Applied Forestry. 17(4): 147–156.

Grebner, D.L.; Londo, A.J.; Sun, C. [and others]. 2007. Potential carbon sequestration opportunities and issues for bottomland hardwood afforestation in the Lower Mississippi Alluvial Valley. Forum of Public Policy: A Journal of the Oxford Round Table. 3(4): 305–312.

Grebner, D.L.; Perez-Verdin, G.; Sun, C. [and others]. 2009. An empirical framework to estimate woody biomass from forestry residue, fuel treatments, and urban waste. In: Solomon, B.D.; Luzadis, V.A., eds. Renewable energy from forest resources in the United States. Routledge, London: [Publisher unknown]. 330 p. Chapter 12.

Iles, J.C. 2008. A stand level growth and yield model for red oak/sweetgum forests in southern bottomlands. [Number of pages unknown]. Mississippi State, MS: Mississippi State University. M.S. thesis.

Matney, T.G. 1992. A thinned/unthinned loblolly pine growth and yield simulator for planted cutover site-prepared land in the mid-gulf south. Southern Journal of Applied Forestry. 16(2): 70–75.

Perez-Verdin, G.; Grebner, D.L.; Sun, C. [and others]. 2009. Woody biomass availability for bioethanol conversion in Mississippi. Biomass and Bioenergy. 33: 492–503.

Rousseau, R.W. 2008. Carbon credits: a non-traditional source of revenue for Mississippi forest landowners. Mississippi State University Extension Service Publ. 2498. http://msucares.com/pubs/publications/p2498.pdf. [Date accessed: July 18, 2011].

Schultz, E.B.; Matney, T.G.; Grado, S.C. [and others]. 2008. A carbon estimation system: rural and urban forest case studies [CD–ROM]. Proceedings of the 2008 Society of American Foresters national convention. 10 p.

Zarnoch, S.J.; Feduccia, D.P.; Baldwin, V.C.; Dell, T.R. 1991. Growth and yield model predictions for thinned and unthinned slash pine plantations on cutover sites in the west gulf region. Res. Pap. SO-264. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 32 p.

RESULTS OF A LONG-TERM THINNING STUDY IN SOME NATURAL, EVEN-AGED PINE STANDS OF THE MIDSOUTH

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Abstract—This paper reports on a long-term thinning study established in stands of naturally seeded loblolly (*Pinus taeda* L.) and shortleaf (*P. echinata* Mill.) pine in southern Arkansas and northern Louisiana. Plots were established in 1949–50 and 1954 in previously unmanaged stands, thinned about once every 5 years from age 20 to 60 years (40 years of active cutting, to 1990). The study was discontinued in 1995 when the stands were about 65 years old. Low-density stands on good sites produced bigger individual pines more quickly than denser stands on medium sites. Long-term sawtimber yields did not follow this pattern, however. While medium-quality sites produced somewhat lower gross yields, denser stands ultimately resulted in significantly higher total yields, primarily because of their better stocking.

INTRODUCTION

Thinning is crucial to managing many southern pine stands, as they often have such high stocking that stagnation, diminished sawtimber production, and mortality are of concern. For instance, Brender (1965) reported that thinned loblolly pine (*Pinus taeda* L.) stands could experience a 10-percent or more volume increase and that rotation age can be shortened by at least 10 years when thinnings are "judiciously" applied. Similarly, Williston (1978) recommended that dense shortleaf pine (*P. echinata* Mill.) plantations be aggressively thinned to avoid stagnation and elevated mortality in crop trees. Changes in technology, coupled with financial pressures to shorten rotation lengths, make it imperative that stands are appropriately treated in order to maximize return.

While many publications on the thinning of pine plantations are available (e.g., Brender 1965, Goebel and others 1974, Williston 1978), informative guidelines for long-term management of even-aged loblolly pine-dominated stands of natural origin are more limited. Mann and Lohrey (1974) provided advice on the precommercial thinning of natural southern pine stands, and Andrulot and others (1972) evaluated a thinning study initiated in 17-year-old loblolly pine-dominated stands that seeded in following the cutting of virgin forest. Both of these studies supported thinning to improve growth and yield and reduce mortality over unthinned controls. More recently, Zeide and Sharer (2000) published a management guide for parts of the Midsouth, based on results from research forests and the experiences of forest managers. For natural stands on medium-quality sites, they recommended thinnings, stocking targets, vegetation control, and a rotation age of 45 years.

This paper reports on a long-term (40+ years) thinning study in loblolly pine-dominated stands of natural origin in southern Arkansas and northern Louisiana. Originally designed to explore the possibilities of thinning mixed pine stands on different quality sites, the longevity of this research now allows for the evaluation of harvest treatments on key attributes such as tree size and stand growth and yield. These attributes, in turn, can be used to guide management recommendations for landowners interested in a specific goal for their properties.

METHODS

Plot Establishment and Description

The following description has been primarily taken from the original project establishment report[2] and a later unpublished summary.[3] The original set of study plots was established in the winter of 1949–50 on what were then Crossett Lumber Company lands in Morehouse Parish, LA, and Ashley County, AR. The Morehouse Parish sites were originally cut about 1918 to a 14-inch diameter at breast height (d.b.h.) limit, and then were repeatedly burned and grazed (but not logged), delaying the establishment of the next pine stand on this site until the late 1920s. The original forest on the Ashley County sites was cut to a 12-inch d.b.h. limit between 1925 and 1930. The initial overstory vegetation on all plots was a mixture of loblolly and shortleaf pine, but the actual proportion of loblolly vs. shortleaf pine was not recorded (Burton 1980). Plots were generally placed to avoid the remnant old pine and scattered large hardwoods (Burton 1980).

The establishment plan included 5 silvicultural treatments (70, 85, and 100 square feet per acre basal area targets; thin to increasing basal area; and best judgment thinning) using 2 thinning directions (thin from below and thin from above) for all but the best judgment thinning (which used both) on 2 levels of site quality (medium and good), each replicated 3 times [(4 treatments × 2 thin directions + 1 treatment) × 2 site qualities × 3 replicates = 54 plots]. Additional plots were included in 1954 for the good sites in Ashley County only,

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[2] Mann, W.F.; Williston, H.L. 1950. Management of young pine stands in the shortleaf-loblolly pine type west of the Mississippi River. 31 p. Unpublished project establishment report, study FS-SO-1107 1.7. On file with: Crossett Experimental Forest, U.S. Department of Agriculture, Forest Service, Southern Research Station, P.O. Box 3516, UAM Station, Monticello, AR 71656.

[3] Leduc, D. 1987. Background, history, and variables of the big thinning study. Unpublished report. On file with: Crossett Experimental Forest. U.S. Department of Agriculture, Forest Service, Southern Research Station, P.O. Box 3516, UAM Station, Monticello, AR 71656.

adding three new basal area targets (55, 115, and 130 square feet per acre) using thinning from below only, once again replicated 3 times (3 treatments × 1 thin direction × 1 site quality × 3 replicates = 9 plots). However, this paper reports (table 1) results from only 60 of the 63 plots established, as 1 plot was severely damaged by bark beetles in 1954 and then destroyed by a tornado in 1973, and 2 other plots were unintentionally logged in 1989. Each plot consisted of a core 0.1-acre circular measurement plot surrounded by a 0.255-acre isolation strip in which every tree at least 2 inches d.b.h. was inventoried (see footnote 2). Periodic losses from logging, wind, ice, and insects have been reported on some of the

Table 1—Number of replicates, measurements, and average replicate site index by thinning level and method

Thinning level[a]	Thinning method	Site quality	Plots	Measurements	Year first measured	Year 65		
						Site index[b]	Stand density	Basal area
			---------- *number* ----------			*feet*	*trees per acre*	*ft² per acre*
55	Below	Good[c]	3	9	1954	94.7	16.7	53.2
70	Below	Medium	3	10	1949–50	76.7	50.0	79.3
		Good	2	10	1949–50	96.5	16.7	49.8
70	Above	Medium	3	10	1949–50	77.0	46.7	80.4
		Good	3	10	1949–50	95.3	30.0	81.5
85	Below	Medium	3	10	1949–50	83.0	56.7	98.1
		Good	2	10	1949–50	96.5	45.0	92.8
85	Above	Medium	3	10	1949–50	82.3	56.7	97.1
		Good	2	10	1949–50	94.0	35.0	78.6
100	Below	Medium	3	10	1949–50	81.0	86.7	112.5
		Good	3	10	1949–50	93.3	50.0	102.0
100	Above	Medium	3	10	1949–50	83.3	70.0	106.2
		Good	3	10	1949–50	99.0	66.7	108.2
115	Below	Good[c]	3	9	1954	99.0	50.0	106.6
130	Below	Good[c]	3	9	1954	98.0	80.0	138.4
Increasing[d]	Below	Medium	3	10	1949–50	80.3	43.3	113.7
		Good	3	10	1949–50	97.3	26.7	108.8
Increasing[d]	Above	Medium	3	10	1949–50	78.7	96.7	112.6
		Good	3	10	1949–50	91.0	63.3	109.9
Judgment[e]	Both	Medium	3	10	1949–50	79.7	80.0	83.2
		Good	3	10	1949–50	97.3	53.3	68.2

[a] Levels represent postharvest basal area (in square feet per acre) thinning targets. After the stands reached their target basal areas, they were maintained at those levels using thinning from below (Burton 1980).

[b] Site index, base age = 50 years.

[c] Later treatment installed with no medium site locations.

[d] Thin from an initial basal area of 70 square feet per acre at age 20 to 105 square feet per acre at age 60.

[e] Best judgment of the staff at the Crossett Experimental Forest with no specific target basal area or fixed thinning direction (some were thinned partly from above, but most were thinned from below (Burton 1980)).

plots, but other than the three plots previously mentioned, no major catastrophic disturbances have strongly affected the plots. Effective fire control in the area began about 1935, with little to no damage from fire over the next 60 years.

When established, the study plots were segregated into two blocks (good vs. medium) based on relative site quality. Assignment to one block or the other was based on early estimates of 50-year site index (SI_{50}), and later tested directly when the stands actually reached 50 years old (Murphy and Farrar 1985). Overall, the early estimates of site quality appeared to have been reasonably effective in achieving the desired separation. However, direct determination of SI_{50} identified some discrepancies in the assignment of site quality. For example, the best quality "medium" site has a SI_{50} = 92, while the lowest quality "good" site has a SI_{50} = 90. While it would be easy to reassign plots to increase the distinctiveness of the extremes of each study, this would not be statistically appropriate because the treatments and replicates were assigned with the preliminary site index values. A comparison of mean site index values by "medium" and "good" site quality classes found that as a group, both classes were highly significantly different [medium SI_{50} = 80.2 feet, good SI_{50} = 96.0 feet, $P < 0.001$ using Welch approximate t-test for unequal sample variances (Zar 1984)], suggesting that broad references to site quality should prove robust.

Thinning Treatments

Implementation—Initially, this study considered the impacts of thinning regimes on high-value pole production (Burton 1977); however, it eventually developed into a general examination of thinning impacts on even-aged pine growth and yield. Some early implementations of thinning were designed to improve pole production, e.g., bias against trees with sweep, changing thinning direction during study, although it is not expected that these have notably influenced the final results. Every 5 years, stands were thinned to different prescriptions after each plot was inventoried. The final thinning occurred in 1989–90, although plots were also remeasured in 1994–95. The first block of plots representing medium sites (SI_{50} from 70 to 92 feet at 50 years) received half of the treatments, while the good sites (SI_{50} from 90 to 101 feet) received the other half, with three plots allocated by site quality and thinning treatment. The plots established in 1954 were only on good-quality sites (SI_{50} from 92 to 101 feet) and received different thinning treatments.

Prior to thinning, an inventory was made and basal area was calculated in the field, after which the prescriptions were applied. Thinning strategies focused on two principle approaches: thinning from below and thinning from above. The original plots were designed to consider the effects of the different direction on the developing stands, with treatments replicated on both medium- and good-quality sites. Thinning from above to target stand densities (70, 85, and 100 square feet per acre) involved cutting from above when the stands were roughly 20 to 25 years old to these targets, then maintaining them at these densities for all of the succeeding treatment periods using thinning from below. Stands assigned to be thinned from below were always thinned from below. The thinning to increasing basal areas started for both thinning directions at 70 square feet per acre, increasing to 75 square feet per acre by age 25 years, 80 square feet per acre at age 30, 85 square feet per acre at age 35, 90 square feet per acre at age 40, 95 square feet per acre at age 50, 100 square feet per acre at age 55, and finally reaching 105 square feet per acre at age 60, after which it was maintained at this level (Burton 1980, Murphy and Farrar 1985). The thinning from above approach actually only removed dominant trees during the cuts in year 20 and 25, afterwards all thinning was from below. The switch in thinning direction was designed to help improve pole production (Burton 1977). The thin from below treatment under increasing basal area used low thinning for all treatment periods. The best judgment thinnings were based on a consensus of the participants, with no restrictions on method, intensity, or residual basal area target. This resulted in mostly thinning from below to basal areas of around 75 square feet per acre (ranging from 67 to 81 square feet per acre), regardless of site quality (Burton 1980). The supplementary plots added in 1954 were thinned from below to basal areas of 55 square feet per acre, 115 square feet per acre, and 130 square feet per acre, and were maintained by low thinning at these levels for the duration of the study.

While the differentiation between loblolly and shortleaf pine was not consistently performed on the plots, the application of the treatments biased the stand in favor of loblolly pine. Burton (1980, p. 3) reported that "[f]or all thinning treatments, if the trees were of equal quality, field workers cut shortleaf and kept loblolly pines." The net effect on stand composition was the increasing dominance of loblolly pine.

Hardwood Competition Control—Since the objective of this study was to consider pine productivity under different thinning treatments, all merchantable hardwoods were cut in the first thinning and any remaining hardwoods were eliminated (Burton 1980). Periodic removal of hardwoods with herbicides was then performed as needed—(see footnote 3) reported treatments in 1949 (using Ammate), 1954 (1 percent emulsion of 2,4,5-T in water), and 1959 (10 percent 2,4,5-T in diesel oil). Mechanical removal of all live hardwoods with root-collar diameters >1.0 inch was also implemented before every inventory through at least age 40 (Burton 1980, Murphy and Farrar 1985).

Inventory Design—Detailed information was recorded over the years on the pines in the study plots, but some measures were inconsistently applied and others were missing from the data. Additionally, the long tenure of this study resulted in different techniques and measurement standards being used, thus potentially confounding possible treatment effects with observer bias. For example, before 1960, many small trees were tracked simply as tallies in diameter classes, so their individual fates were largely unknown. Additionally, no records were kept of mortality during the first couple of inventory periods (Burton 1980). Because of these inconsistencies, this paper will only consider the differences in the diameter and gross volumes of the stand at age 65 years.

Trees as small as 2.0 inches d.b.h. were tallied in the original inventories; however, only trees >3.5 inches d.b.h. (the minimum merchantable standard in the study region) have been used in the calculations of trees per acre, basal area, volume, and mortality in this paper. Merchantability standards for this analysis follow those applied by Burton (1980) and Murphy and Farrar (1985). Board-foot volume (International 1/4-inch rule) is reported in this paper, calculated with local volume equations developed by Farrar and others (1984).

Statistical Analysis and Silvicultural Interpretation—A number of factors contribute to the difficulties in interpreting field data, especially from long-term studies established decades ago. In particular, this thinning study experienced a number of challenges that could not be controlled in this analysis. First, the small sample size per treatment, coupled with the small size of study plots, resulted in a particularly high sensitivity to disturbance. Second, the relative novelty of statistically based comparative studies in the silvicultural research programs of the late 1940s led to plots being established across a broad range of initial conditions and, hence, limited replication. Finally, the duration of the study (~45 years) resulted in multiple investigators with varying goals, who thereby implemented the studies and measurements using somewhat different standards and practices. The primary result of these design and implementation weaknesses is the reduction of power in discerning treatment effects, especially as the study progressed. However, broad conclusions can still be made.

Prior researchers published different approaches to the analysis of data from this study: Burton (1980) considered each plot type (medium, good, and supplemental) as three separate entities and performed analysis on each, while Murphy and Farrar (1985) combined all the data as a completely randomized design using analysis of covariance, with site index as the covariate. This paper differs from previous efforts by concentrating on the long-term implications of the different thinning treatments. Comparisons are limited to the factors most relevant to making long-term decisions for the management regime to implement using the units that best reflect the nature of the stand. Thus, only preharvest d.b.h. and potentially usable (gross) sawtimber yield (both total and annualized) were considered. Results are presented for all treatments as means at the end of the study when the stands had reached approximately 65 years of age. All treatments were compared using Tukey's honestly significant difference test for multiple comparison with unequal numbers. To avoid problems of homogeneity of variance and nonnormality within treatments, a logarithmic transformation [$X' = \log(X + 1)$] was used on the data (Zar 1984).

RESULTS AND DISCUSSION

Because basal area was tightly controlled for this study, it is not surprising that the basal area at 65 years approached the harvest goals (table 1). Towards the end of the study, the heavy thinning treatments dropped to unacceptably low stand densities as the small plots gradually ran low on trees. Thus, for low-density treatments, this study had reached the end of its usefulness. It was clear, however, that sustained harvesting effectively regulated stocking. By the time these stands approached 50 years old, Murphy and Farrar (1985) found average annual density increases of 2.5 to 3.5 square feet per acre, most of which was occurring in sawtimber-sized individuals. This rate was slightly higher than that noted by Nelson (1963) for natural-origin loblolly pine in Georgia, South Carolina, and Virginia.

Mean Preharvest Stand Diameter

Table 2 encapsulates the influence of thinning treatments on average preharvest mean d.b.h. at 65 years. Initially, all study areas had similar d.b.h. values, ranging from 5.0 to 6.3 inches. However, scheduled treatments soon altered this parity, rapidly producing differentiation by harvest strategy. For example, thinning to 55 square feet per acre from below quickly opened the stand by removing the smallest individuals, thus producing a rapid increase in d.b.h. from 7.3 to 9.5 inches in the first 5 years of treatment. Thinning the stands to much higher densities, e.g., 115 or 130 square feet per acre, from below only produced an increase from 7.3 to 8.4 inches in the same time period. Under these conditions, the greater retention of small-diameter stems and higher stocking reduced the diameter response, especially when the canopy had closed and competition for resources was pronounced (Mann and Lohrey 1974). All of these results are intuitive and have been shown in many other thinning studies (e.g., Andrulot and others 1972, Chaiken 1941, Wiley and Zeide 1992).

Maintenance of prescribed thinning levels reinforced these differences. As expected, retaining lower stand densities produced larger individuals on average during the study (fig. 1). In most cases, thinning direction (above vs. below) slightly influenced d.b.h. during the treatment period, largely because of the preferential removal of small trees when thinning from below. For older even-aged stands that have been thinned for many years, the apparent difference between the thinning directions will probably have little practical significance because, if properly done, any size differentiation will have been ameliorated by the repeated removals of small pines.

While the limited number of sampled stands and trees contributed to the lessening of significance between treatments and site quality, differences were still apparent and meaningful. The impact of site quality on d.b.h. over time was likely more influential than thinning design. For treatments paired on both site-quality levels, d.b.h. was consistently higher on the good-quality sites throughout the treatment period (fig. 1). Better sites translate into faster stem growth, allowing the thinned stands to develop more rapidly. Under unthinned natural conditions, the accelerated growth experienced on good sites resulted in earlier canopy closure and heightened competition, thus triggering density-dependent mortality at an earlier age (see also Turnblom and Burk 2000). However, properly thinned stands are precluded from reaching canopy closure, thus ensuring better exploitation of site resources by the residual trees (Wiley and Zeide 1992) and allowing d.b.h. increases to be maintained.

Table 2—Comparison of preharvest average diameter by harvest treatment and site quality

Thin level[a]	Thin method	Site quality	Average age = 25 years	Average age = 65 years[b]	Year 65 Average minimum	Year 65 Average maximum	Year 65 Standard deviation
			d.b.h. inches				
55	Below	Good	7.3	23.9 f	22.5	25.6	1.58
70	Above	Medium	5.7	17.1 a,b,c,d,e	15.6	18.8	1.61
		Good	6.8	23.1 f	21.2	25.5	2.18
70	Below	Medium	5.7	18.0 a,b,c,d,e	15.3	19.8	2.38
		Good	6.6	22.3 d,e,f	21.5	23.1	1.13
85	Above	Medium	5.7	17.7 a,b,c,d,e	17.1	18.7	0.87
		Good	6.0	19.4 a,b,c,d,e,f	18.6	20.2	1.13
85	Below	Medium	5.9	18.0 a,b,c,d,e	15.7	21.2	2.84
		Good	7.0	20.2 b,c,d,e,f	20.2	20.3	0.07
100	Above	Medium	5.8	15.3 a,b	15.0	16.0	0.58
		Good	6.5	19.4 b,c,d,e,f	18.5	21.2	1.53
100	Below	Medium	5.9	16.7 a,b,c,e	16.3	17.0	0.35
		Good	6.3	17.1 a,b,c,d,e	16.5	18.1	0.87
115	Below	Good	7.3	19.8 c,d,e,f	17.7	21.6	1.97
130	Below	Good	7.3	17.9 a,b,c,d,e	16.4	20.5	2.29
INC	Above	Medium	5.2	14.7 a	13.7	15.9	1.12
		Good	5.8	17.7 a,b,c,d,e	16.8	18.3	0.79
INC	Below	Medium	6.1	16.0 a,b,c	15.9	16.0	0.06
		Good	6.8	19.4 b,c,d,e,f	18.5	19.9	0.81
JUD	Both	Medium	5.6	18.8 b,c,d,e,f	17.9	19.2	0.75
		Good	6.4	21.7 d,f	20.3	23.0	1.36

[a] Number represents postharvest basal area (square feet per acre) targets; INC = thin to increasing basal area; JUD = best judgment thinning. See text for detailed descriptions of the thinnings.

[b] Diameters with the same letters are not significantly different at $\alpha = 0.05$ (tests were conducted on transformed data but reported for untransformed). Significance of treatment differences determined with Tukey's honestly significant difference (HD) test for unequal *n* using an $\alpha = 0.05$ (StatSoft 2000).

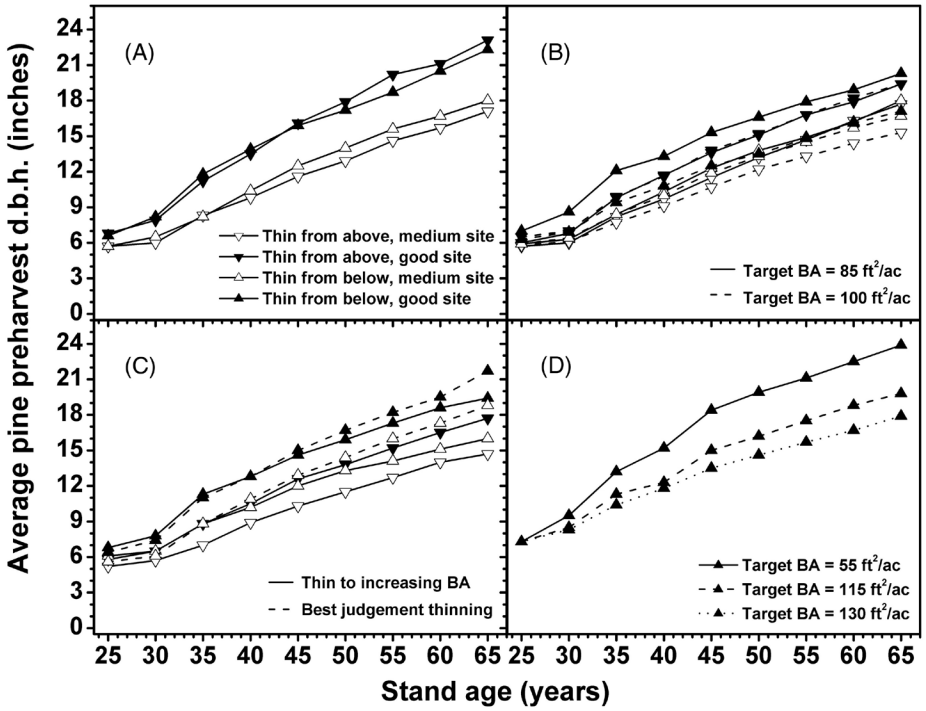

Figure 1—Average pine preharvest d.b.h. over the duration of this thinning study. Target residual basal areas (BA) were (A) 70 square feet per acre, (B) 85 or 100 square feet per acre, (C) thin to increasing BA or best judgment thinning, or (D) thinning from below to varying BA on good sites.

Growth and Yield

According to Miscellaneous Publication 50 (U.S. Department of Agriculture Forest Service 1929), unthinned, natural loblolly pine-dominated sawtimber stands of comparable age and site indices (averaging between 80 and 100 feet at 50 years) have an average periodic annual increment of between 750 and 900 board feet (International 1/8-inch rule) per acre (U.S. Department of Agriculture Forest Service 1929). This study (table 3) showed a 65-year gross annual increment between about 350 and 600 board feet (International 1/4-inch rule) per acre, which is somewhat greater than the production of most uneven-aged naturally regenerated pine stands in this region (Baker and Murphy 1982) but only one-half to two-thirds of that reported in Miscellaneous Publication 50 (U.S. Department of Agriculture Forest Service 1929). However, note that this was for the first 65 years of the stands' lives—not a periodic increment from a single, fast-growing decade. In the decades prior to this, both Burton (1980) and Murphy and Farrar (1985) reported 10 to 20 percent higher volume growth from this study than that reported in U.S. Department of Agriculture Forest Service (1929) over comparable intervals.

Though few differences proved significant, a general trend of increasing sawtimber increments with increasing residual stocking (at least through the range used) was evident (table 3). This long-term trend has been reported by others (e.g., Curtis and others 1997) and explained conceptually (Zeide 2001). It would appear that greater site occupancy by larger numbers of trees eventually produces a larger quantity of wood, even though a portion of it may be lost to mortality. None of the treatments in this study approached the level of stocking that would have resulted in stand-level growth stagnation, nor were any uncut control plots established to approximate this threshold.

Growth started to decline in these sample plots as the study progressed due to a combination of lower stocking levels and the reduced increment of larger, older pines. Burton (1980) reported the highest periodic annual growth rates for these treatments when they were 35 to 40 years old, averaging about 875 board feet per acre (Doyle rule), with few differences between the densest and most open stands. A decade later, Murphy and Farrar (1985) reported that growth decreased to an average of just under 730 board feet per acre per year (Doyle rule), with the three stands with the lowest residual basal areas (55 and 70 square feet per acre) distinctly lower [averaging 611 board feet (Doyle) per acre per year] than the rest of the treatments. This difference was even more pronounced by the time the stands reached 65 years (about 500 board feet per acre per year, International 1/4-inch rule, data not shown).

Impacts of Thinning on Sawtimber Yield

Unless a stand is grossly overstocked, thinning is generally thought not to significantly affect long-term gross yields (Baldwin and others 2000, Smith 1986, Zeide 2001). Short-term increases in sawtimber production following thinning

Table 3—Potentially usable sawtimber yield (averaged by replicates) for each harvest treatment by year 65

Thin level[a]	Thin method	Site quality	Volume at age 65[b]	Total harvest[c]	Mortality loss[c]	Total gross yield[d]	Gross annual increment[e]
			thousand board feet per acre, International 1/4-inch rule				*board feet per acre*
55	Below	Good	12.0	17.0	1.9	31.0 a,b,c,d,e	476
70	Above	Medium	14.3	8.3	0.0	22.6 e	348
		Good	11.0	16.5	4.4	31.9 a,b,c,d,e	491
70	Below	Medium	15.0	9.0	0.0	23.9 d,e	368
		Good	17.5	14.1	0.0	31.6 a,b,c,d,e	486
85	Above	Medium	18.3	10.7	0.0	29.0 a,b,c,d,e	446
		Good	18.2	13.0	1.3	32.6 a,b,c,d	501
85	Below	Medium	18.2	10.4	0.0	28.6 a,b,c,d,e	440
		Good	15.8	16.5	2.5	34.7 a,b,c	534
100	Above	Medium	19.0	6.8	0.0	25.8 c,d,e	396
		Good	20.0	10.3	3.8	34.1 a,b,c	524
100	Below	Medium	18.8	9.4	2.5	30.7 a,b,c,d,e	472
		Good	19.8	8.8	1.6	30.2 a,b,c,d,e	465
115	Below	Good	21.3	12.3	5.4	39.0 a	600
130	Below	Good	26.0	6.4	5.4	37.8 a,b	581
INC	Above	Medium	18.7	5.9	0.2	24.8 c,d,e	381
		Good	20.2	7.5	1.3	28.9 a,b,c,d,e	445
INC	Below	Medium	19.4	6.8	0.2	26.5 b,c,d,e	407
		Good	21.5	12.4	1.1	35.0 a,b,c	539
JUD	Both	Medium	15.9	11.3	0.0	27.2 a,b,c,d,e	418
		Good	14.4	14.6	3.4	32.4 a,b,c,d	499

[a] Numbers represent postharvest basal area (square feet per acre) targets; INC = thin to increasing basal area; JUD = best judgment thinning. See text for detailed descriptions of the thinnings.

[b] Volume at age 65 = standing (live) board-foot volume at stand age = 65 years, in thousands of board feet per acre, International 1/4-inch rule.

[c] Total harvest = cumulative amount of harvested sawtimber from year 25 to year 65; mortality lost = cumulative volume of sawtimber lost to natural causes from year 25 to year 65.

[d] Total gross yield = volume at age 65 + total harvest + mortality lost. Total gross yield values with the same letters are not significantly different at $\alpha = 0.05$ (tests were conducted on transformed data but reported for untransformed, Tukey's HSD test).

[e] Gross annual increment = total gross yield/65 years, in board feet per acre (International 1/4-inch rule).

are possible, especially in younger stands when thinning accelerates individual tree growth, thereby allowing pines to reach the minimum size threshold for sawtimber faster. The results of this study generally support this interpretation within site-quality limitations (table 3). Medium-quality sites generated between 22 and 31 thousand board feet (mbf) per acre over the duration of this study. Good sites were noticeably more productive, yielding a gross sawtimber range of 29 to 39 mbf per acre during the study. Variability in harvest implementation and mortality introduced considerable noise into the treatments, obscuring some of the differences between treatments. Broadly, the most heavily thinned stands (regardless of site quality) had somewhat lower gross yields (table 3), largely due to suboptimal stocking of the sites as the stands aged.

Thinning Removals

Over the years, the stands maintained at the lowest residual basal areas (those ≤85 square feet per acre) typically produced the greatest harvest of pine sawtimber, with the 55-square-feet-per-acre residual treatment producing just over 17 mbf per acre (table 3). Conversely, those treatments retaining the most basal area yielded the least harvested sawtimber during this period. Differences were significant between only a few of the treatments because of mortality-related losses and some variability in initial conditions between treatments. These differences would have become more pronounced if the study had been continued even longer, as the heaviest thinning treatments were too understocked to reach minimum basal area harvest thresholds. For example, by year 65 the 55-square-feet-per-acre treatment, with a mere 17 trees per acre, had failed to reach the 50-square-feet-per-acre target in consecutive treatment cycles.

Site quality also dramatically influenced sawtimber production, with good sites averaging 25 to 50 percent more harvested yield than medium sites. An exception to this pattern was for the 100-square-feet-per-acre treatment thinned from below, in which the medium site had 9.4 mbf per acre cut during the study compared to 8.8 mbf per acre from the good sites. Since both stocking and mortality patterns between these treatments were similar, the most logical explanation for the difference was inconsistency in harvesting, resulting in a greater proportion of unharvested volume on the good site (table 3).

Thinning and Mortality

A combination of small plots, coupled with the lack of an unthinned control, makes it difficult to glean much from the mortality records of this study. Due to a high level of variability, none of the treatments produced statistically significant differences in cumulative mortality, even though no mortality was reported for some treatments, while others generated cumulative losses of up to 5.4 mbf per acre (table 3). In general, the more heavily thinned stands experienced lower natural mortality than those sustaining higher basal areas. Comparable research in other southern pine stands has also shown dramatic declines in mortality following thinning (Andrulot and others 1972, Guttenberg 1954). In addition, the residual pines in thinned stands also tend to be less vulnerable to insect and disease problems, so long as they are not extensively wounded during harvest (Brown and others 1987, Chaiken 1941) or are too spindly to respond quickly to release, thus making them vulnerable to ice damage (Guttenberg 1954).

MANAGEMENT IMPLICATIONS

As in all cases, deciding on a management regime for even-aged pine stands of natural origin in the Midsouth depends on the objectives of the landowner, the nature of the site, and initial forest conditions. Long-term research studies that consider different silvicultural treatments as well as site and species potential are invaluable tools for guiding the forest manager towards the best decision.

Timber Production

Fundamentally, the rapid growth expressed in these treatments has significant implications for regional timber production. As expected, heavily thinned stands had more sawtimber cut and less of a residual stocking than treatments geared towards a higher target basal area. From a management perspective, this range represents a gradient of opportunity based on economics and nontimber attributes. For instance, landowners primarily interested in monetary returns from timber sales would be best served by the heaviest thinning regimes, which produce more harvested timber earlier in the history of the stand. Not only are more trees removed, but the residual experiences more release, thus permitting individual pines to grow more rapidly. Given the past strength of the sawtimber market in southern Arkansas and northern Louisiana, the diameter threshold between sawtimber and lower value products becomes critical. The sooner a pine reaches sawtimber merchantability, the quicker higher returns can be realized.

Another consideration is the timing of the initial thinnings. When this study was established in the middle of the 20th century, it was not uncommon to let naturally regenerated pine stands grow unthinned until they reached merchantable size. In this case, thinning was not started until they were 20 to 25 years old (even on good sites). However, considerable evidence has since accumulated on the value of early precommercial thinning to reduce the extremely high stocking found in most naturally seeded pine stands (e.g., Brender 1965, Burton 1982, Mann and Lohrey 1974). When properly timed, precommercial thinning occurs after canopy closure (to allow for self-pruning) but before competition becomes so intense to result in stagnation and mortality. Often implemented when the stand reaches 5 to 15 years old, precommercial thinning accelerates the growth and reduces the number of years it takes to reach sawtimber size.

A radical example of precommercial thinning can be seen in the “Sudden Sawlog” study (Baldwin and others 1998, Burton 1982, Zahner and Whitmore 1960). In this study, an old field near the Crossett Experimental Forest was planted

with unimproved 1-0 loblolly pine seedlings to a density of about 1,100 trees per acre. Four thinning strategies were implemented, with thinnings starting between 9 and 12 years of age and some treatments reduced to as low as 100 crop trees per acre (Burton 1982). The three most dramatic treatments (the fourth being a traditionally thinned control, starting at age 12) produced stands of average diameters of 17 to 18 inches at age 33 years. The use of precommercial thinnings dramatically reduced the time it took the treatments to reach merchantable size, with sawtimber-sized average stand diameters appearing at 15 to 18 years, or approximately half of the time it took the stands in this study. However, limbiness, juvenile wood production, bole taper, and ice or wind damage are major challenges for radically thinned pine stands (Baldwin and others 2000, Bragg and others 2003, Burton 1982).

Quantity vs. Quality

Foresters have long recognized that individual trees grow most rapidly in diameter when they have few neighbors, but that wide spacing does not always optimize the potential of a piece of land. In other words, maximizing tree growth does not necessarily maximize stand growth. Understocked stands are a management concern when the primary objective is total fiber production (Baker and Shelton 1998a), although there is good evidence that for rapid sawtimber production, such conditions may be advantageous (Burton 1977, Burton and Shoulders 1974). It is also important to keep in mind that the dollar value of a pine is nonlinearly related to size, as there typically is a dramatic step increase in value when the tree crosses the threshold from pulpwood size-only to poles or sawlogs (Burton 1977, Prestemon and Buongiorno 2000). Thus, determining the value of silvicultural treatments is not as straightforward as simply calculating volumetric bole increment.

In young, even-aged pine stands, rapid stem growth is best sustained by periodic thinnings that release sufficient resources such that self-thinning-based growth reductions and mortality are minimized between their applications. Fast growing trees, however, have different physiological properties that should be considered when thinning regimes are designed. For example, widely spaced and unpruned young stands (natural or planted) result in trees with coarser and more abundant branches, a higher proportion of less desirable juvenile wood, and more bole taper (Baldwin and others 2000, Burton 1982, Clark and others 2004). On average, rapidly grown loblolly pines produce less dense wood with lower bending strength than slower growing individuals (Paul 1932, see also review in Bendtsen 1978). Indeed, logs with a greater core of juvenile wood are decidedly less valuable than those that have mature wood (Clark and others 1994, Guilkey and Nelson 1963, Paul 1932). As a result, Guldin and Fitzpatrick (1991) found that uneven-aged loblolly pine stands produced better quality sawlogs on average than even-aged plantations (and, presumably, thinned even-aged natural stands), although it took longer to produce these logs.

Mortality Patterns and Thinning

Thinning is also known to help reduce density-dependent mortality. For example, Guttenberg (1954) and Andrulot and others (1972) reported significantly greater mortality in unthinned stands than those that were thinned, attributable largely to suppression and related forest health problems. Guttenberg (1954) also reported a shift in the cause of mortality, with ice storms claiming over twice that of competition (59 vs. 28 percent) in the thinned plots, whereas competition took 11 times (89 vs. 8 percent) that lost to glazing in unthinned stands. A recent paper (Bragg and others 2003) echoed this conclusion in a review of many studies that pointed to a greatly pronounced risk of glaze-related losses in recently thinned timber.

In general, thinning improves the health and vigor of the residual pines by permitting individuals with stunted crowns to recover and even thrive (Baker and Shelton 1998b, Guttenberg 1954). High-vigor loblolly pine are less vulnerable to insect pests such as the southern pine beetle (*Dendroctonus frontalis*) (Belanger 1980, Brown and others 1987), and once they have adjusted to the lower stand densities and increased in size, can better tolerate severe winds or ice accumulation (Bragg and others 2003, Zeide and Sharer 2000). However, thinning can lead to additional mortality in residual loblolly pine if they are excessively damaged by the logging, or if poorly formed or suppressed trees are left behind (Belanger 1980, Chaiken 1941).

CONCLUSIONS

One of the greatest difficulties in assessing the success of silvicultural systems is the ability to conduct long-term studies. Rarely do the appropriate combinations of institutional interest and commitment, vegetative conditions, and market forces coincide. Thus, thinning studies that span decades are of particular value, even if their original implementation leaves something to be desired. This study of different thinning techniques in loblolly pine stands of natural origin provided a long-term assessment of stand growth and yield for typical sites in the Upper West Gulf Coastal Plain Province of southern Arkansas and northern Louisiana. Coupled with other similar efforts, a clearer picture of the implications of thinning has been developed, allowing for a range of recommendations to be made based on landowner preferences.

In this portion of the Midsouth, conventional wisdom on even-aged loblolly and shortleaf pine stands suggests a 35- to 45-year rotation, depending on site quality and management objectives (Burton 1980, Zeide and Sharer 2000). Industrially oriented forest landowners typically intensively manage their improved pine plantations on a 25- to 30-year sawtimber rotation, driven by the goal of maximizing return on investment (Arano and Munn 2006, Hotvedt and Straka 1987). Even though these silvicultural prescriptions have a reasonable basis in their application, they should not necessarily be considered the only options available to forest managers. One of the greatest benefits of the 40+ years of growth-and-yield information presented here

is the ability to use this long-term data to compare different thinning treatments for not only their fiber productivity, but to understand the nature of the stands during their development and at the end of the study. This perspective can then be matched to the objectives of the landowner(s).

After all, not every landowner is interested in the same end product, and hence may not maximally realize their management goals under a one-size-fits-all prescription. For instance, Burton (1980) recommended heavy early thinnings and aggressive thinning from below and competition control to promote sawtimber production, while suggesting a higher residual density strategy for those interested in wood fiber production. A landowner interested in high-value pole production would focus on straight, relatively limb-free crop trees, perhaps through pruning or high-residual basal areas. By necessity, this represents a management strategy applied differently with different goals than one focused on optimizing sawtimber or fiber production (Burton 1977, Zeide 2001). Furthermore, though they may appreciate the income generated from managing their timber, many landowners are also interested in attributes such as aesthetics, recreation, wildlife habitat, carbon storage, deferred income, or leaving a legacy for their descendants (Hoover and others 2000, Johnson 1995, Zeide 2001), all of which will affect long-term stand structure and composition.

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LITERATURE CITED

Andrulot, E.R.; Blackwell, L.P.; Burns, P.Y. 1972. Effects of thinning on yield of loblolly pine in central Louisiana. Bull. 6. Ruston, LA: Louisiana Technological University College of Life Sciences. 145 p.

Arano, K.G.; Munn, I.A. 2006. Evaluating forest management intensity: a comparison among major forest landowner types. Forest Policy and Economics. 9: 237–248.

Baker, J.B.; Murphy, P.A. 1982. Growth and yield following four reproduction cutting methods in loblolly-shortleaf pine stands–a case study. Southern Journal of Applied Forestry. 6: 66–74.

Baker, J.B.; Shelton, M.G. 1998a. Rehabilitation of understocked loblolly-shortleaf pine stands—I. Recently cutover natural stands. Southern Journal of Applied Forestry. 22: 35–40.

Baker, J.B.; Shelton, M.G. 1998b. Rehabilitation of understocked loblolly-shortleaf pine stands—II. Development of intermediate and suppressed trees following release in natural stands. Southern Journal of Applied Forestry. 22: 41–46.

Baldwin, V.C., Jr.; Leduc, D.J.; Ferguson, R.B. [and others]. 1998. The not-so-sudden results of the sudden saw log study—growth and yield through age 45. 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: 574–578.

Baldwin, V.C.; Peterson, K.D.; Clark, A. [and others]. 2000. The effects of spacing and thinning on stand and tree characteristics of 38-year-old loblolly pine. Forest Ecology and Management. 137: 91–102.

Belanger, R.P. 1980. Silvicultural guidelines for reducing losses to the southern pine beetle. In: Thatcher, R.C.; Searcy, J.L.; Coster, J.E.; Hertel, G.D., eds. The southern pine beetle. Tech. Bull. 1631. Washington, DC: U.S. Department of Agriculture Forest Service, Science and Education Administration: 165–177.

Bendtsen, B.A. 1978. Properties of wood from improved and intensively managed trees. Forest Products Journal. 28: 61–72.

Bragg, D.C.; Shelton, M.G.; Zeide, B. 2003. Impacts and management implications of ice storms on southern forests. Forest Ecology and Management. 186: 99–123.

Brender, E.V. 1965. Thinning loblolly pine. In: Wahlenberg, W.G., ed. A guide to loblolly and slash pine plantation management in Southeastern USA. Macon, GA: Georgia Forest Research Council: 99–110.

Brown, M.W.; Nebeker, T.E.; Honea, C.R. 1987. Thinning increases loblolly pine vigor and resistance to bark beetles. Southern Journal of Applied Forestry. 11: 28–31.

Burton, J.D. 1977. Managing for high-value poles in the loblolly-shortleaf belt. Southern Journal of Applied Forestry. 1: 11–15.

Burton, J.D. 1980. Growth and yield in managed natural stands of loblolly and shortleaf pine in the west Gulf Coastal Plain. Res. Pap. SO-159. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 23 p.

Burton, J.D. 1982. Sawtimber by prescription—the sudden sawlog story through age 33. Res. Pap. SO-179. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 11 p.

Burton, J.D.; Shoulders, E. 1974. Fast-grown, dense loblolly pine sawlogs: a reality. Journal of Forestry. 72: 637–641.

Chaiken, L.E. 1941. Growth and mortality during 10 years following partial cuttings in loblolly pine. Journal of Forestry. 39: 324–329.

Clark, A.; Saucier, J.R.; Baldwin, V.C.; Bower, D.R. 1994. Effects of initial spacing and thinning on lumber grade, yield, and strength of loblolly pine. Forest Products Journal. 44: 14–20.

Clark, A.; Strub, M.; Anderson, L.R. [and others]. 2004. Impact of early pruning and thinning on lumber grade yield from loblolly pine. 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: 199–204.

Curtis, R.O.; Marshall, D.D.; Bell, J.F. 1997. LOGS—a pioneering example of silvicultural research in coast Douglas-fir. Journal of Forestry. 95: 19–25.

Farrar, R.M.; Murphy, P.A.; Willett, R.L. 1984. Tables for estimating growth and yield of uneven-aged stands of loblolly-shortleaf pine on average sites in the west gulf Area. Bull. 874. Fayetteville, AR: University of Arkansas, Division of Agriculture, Arkansas Agricultural Experiment Station. 21 p.

Goebel, N.B.; Warner, J.R.; Van Lear, D.H. 1974. Periodic thinnings in loblolly pine stands: growth, yield, and economic analyses. Res. Ser. 28. Clemson, SC: Clemson University, Department of Forestry. 11 p.

Guilkey, P.C.; Nelson, T.C. 1963. Silvicultural factors affecting sawlog quality of loblolly pine. Journal of Forestry. 61: 190–192.

Guldin, J.M.; Fitzpatrick, M.W. 1991. Comparison of log quality from even-aged and uneven-aged loblolly pine stands in south Arkansas. Southern Journal of Applied Forestry. 15: 10–17.

Guttenberg, S. 1954. Growth and mortality in an old-field southern pine stand. Journal of Forestry. 52: 166-168.

Hoover, C.M.; Birdsey, R.A.; Heath, L.S.; Stout, S.L. 2000. How to estimate carbon sequestration on small forest tracts. Journal of Forestry. 93: 13–19.

Hotvedt, J.E.; Straka, T.J. 1987. Using residual values to analyze the economics of southern pine thinning. Southern Journal of Applied Forestry. 11: 99–106.

Johnson, R. 1995. Supplemental sources of income for southern timberland owners. Journal of Forestry. 93: 22–24.

Mann, W.F.; Lohrey, R.E. 1974. Precommercial thinning of southern pines. Journal of Forestry. 72: 557–560.

Murphy, P.A.; Farrar, R.M. 1985. Effects of different thinning regimes on growth and inventory of 50-year-old loblolly-shortleaf 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: 241–246.

Nelson, T.C. 1963. Basal area growth of natural loblolly pine stands. Res. Note SE-10. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 4 p.

Paul, B.H. 1932. The relation of certain forest conditions to the quality and value of second-growth loblolly pine lumber. Journal of Forestry. 30: 4–21.

Prestemon, J.P.; Buongiorno, J. 2000. Determinants of tree quality and lumber value in natural uneven-aged southern pine stands. Canadian Journal of Forest Research. 30: 211–219.

Smith, D.M. 1986. The practice of silviculture. 8th ed. New York: John Wiley. 527 p.

StatSoft. 2000. Statistica for Windows I. General conventions and statistics. Tulsa, OK: StatSoft, Inc. 1,878 p.

Turnblom, E.C.; Burk, T.E. 2000. Modeling self-thinning of unthinned Lake States red pine stands using nonlinear simultaneous differential equations. Canadian Journal of Forest Research. 30: 1410–1418.

U.S. Department of Agriculture Forest Service. 1929. Volume, yield, and stand tables for second-growth southern pines. Misc. Publ. 50. Washington, DC. 202 p.

Wiley, S.; Zeide, B. 1992. Development of a loblolly pine plantation thinned to different density levels in southern Arkansas. Rep. 322. Fayetteville, AR: University of Arkansas, Division of Agriculture, Arkansas Agricultural Experiment Station. 44 p.

Williston, H.L. 1978. Thinning shortleaf pine plantations in northern Mississippi. Southern Journal of Applied Forestry. 2: 137–140.

Zahner, R.; Whitmore, F.W. 1960. Early growth of radically thinned loblolly pine. Journal of Forestry. 58: 628–634.

Zar, J.H. 1984. Biostatistical analysis. 2nd ed. Englewood Cliffs, NJ: Prentice Hall. 718 p.

Zeide, B. 2001. Thinning and growth: a full turnaround. Journal of Forestry. 99: 20–25.

Zeide, B.; Sharer, D. 2000. Good forestry at a glance: a guide for managing even-aged loblolly pine stands. Bull. 3. Fayetteville, AR: University of Arkansas, Agricultural Experiment Station, Arkansas Forest Resources Center. 19 p.

GROWTH AND STEM FORM QUALITY OF CLONAL *PINUS TAEDA* FOLLOWING FERTILIZATION IN THE VIRGINIA PIEDMONT

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Abstract—Clonal forestry offers the opportunity to increase yields, enhance uniformity, and improve wood characteristics. Intensive silvicultural practices, including fertilization, will be required to capture the full growth potential of clonal plantations. However, variation in nutrient use efficiency that exists among clones could affect growth responses. Our research objective was to determine the range of growth response and stem form quality due to fertilization in clones of *Pinus taeda*. A split-plot experimental design was used, with the whole plots being 2 levels of fertilization (with or without) and the split-plot factor being 25 clones. Whole-plot treatments were blocked and replicated four times. Trees were planted in May 2003, with fertilizer applied in May 2004 and May 2006. Five years after planting, a repeated measures analysis showed fertilizer by time and clone by time interactions significantly affected volume. Although there were no fertilizer by clone interactions in this trial across all 25 clones, the response to fertilizer varied, with 40 percent of the clones showing a volume improvement at 3.5 years of <3 percent while 20 percent showed improvement >15 percent. Our results suggest that a screening technique for clonal response to silvicultural treatments such as fertilization may be necessary given the wide range of fertilizer responses found among clones in this field trial and the large numbers of clones being developed by forest industry.

INTRODUCTION

Two fundamental properties influence tree growth in plantations: (1) genetics, or the ability of a plant to acquire and use resources and (2) the environment, or the availability of resources. Both properties may be manipulated to alter growth rates. Genetics may be constrained by deploying clones with high-growth rates and favorable stem form, while resource availability may be increased through fertilization. Currently clones of loblolly pine (*Pinus taeda*) propagated through both rooted cuttings and somatic embryogenesis are being deployed in increasing numbers throughout the Southeast (Frampton and others 2000, MacKay and others 2006). Fertilization is already a common practice, with approximately 0.5 million ha fertilized annually as of 2004, primarily with both nitrogen and phosphorus (Fox and others 2007). However, it is conceivable that clone by fertilizer interactions could arise, altering growth rates in response to fertilization differently for different clones. The purpose of this research is to determine (1) if the magnitude of potential clone by fertilizer interactions is substantial enough to be a problem and (2) if these interactions are common enough to be a problem. This could be a pressing issue for both landowners who purchase and plant clonally propagated loblolly pine, and for companies that are producing and testing these clones.

We know from the literature that there are few genetics by environment interactions in open-pollinated loblolly pine (McKeand and others 2006), with the notable exception of some genotype by silviculture interactions (Roth and others 2007). In other words, a higher performing family will always be better than a lower performing family, on any uniform site, or under any uniform silvicultural system. Traditional tree breeding utilizes population means to determine whether interactions occur. However, clonal screening is a process that seeks and chooses only a small handful of spectacularly high-performing outliers. Most clonal screening programs are currently planting a large number of clones across a number of sites but are only selecting the best few after no more than 6 years. Clonal screening results in the intentional selection and planting of unusual individuals from the tail end of the population distribution across large acreages. There is a great deal of stochasticity in finding these few elite genotypes, because outliers are by definition infrequent and unusual. Evidence pertaining to clone by environment interactions is currently based only on a small number of field trials which have thus far yielded inconsistent results (McKeand and others 2006).

As of this year several tens of thousands of acres have already been planted with clonal loblolly pine, primarily as single clone blocks. This trend is likely to increase in the next decade, as clonal forestry offers the opportunity to dramatically increase financial returns by increasing timber and pulp yields and plantation uniformity (Dougherty 2007). Clonal screening seeks not only to maximize yields but also to find ideotypes with other favorable characteristics such as straight stems, infrequent forking, flat branch angles, and low incidence of diseases, such as fusiform rust (Nelson and Johnsen 2008). Clones cost about 10 times more than open-pollinated trees, and thus, landowners will demand high performance for their high initial investment (Dougherty 2007). Clonal screening trials are already very large and expensive due to the number of clones being tested and as a result are typically grown under a uniform silvicultural regime. These trials are not testing for clone by fertilizer interactions. Thus, if clone by fertilizer interactions do occur frequently in loblolly pine, it will be necessary to incorporate a screening process for this into clonal screening programs.

Our alternate hypotheses are thus—that we will find significant clone by fertilizer interactions for (1) stem-volume growth, (2) growth efficiency, and (3) stem form. We are

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interested in economic traits for obvious reasons. However, we are also interested in some physiological traits so that if clone by fertilizer interactions for volume growth occur, we can determine the processes causing this interaction to better determine appropriate screening methods.

METHODS

Study Site

Our study was located at the Reynolds Homestead Forest Resources Research Center (36°40' N, 80°10' W) in the upper Piedmont of Patrick County, VA. Annual precipitation is 1308 mm, while mean annual maximum and minimum temperatures are 18.5 °C and 7.0 °C, respectively. Average July high temperature is 29.3 °C, and average January low temperature is –4.0 °C (World Climate 2008). Topography consists of gently sloping hills ranging in elevation from 320 to 340 m. Past intensive agricultural land use has resulted in a truncated Ap horizon, with clayey B horizons mixed with the A. Mapped soil series include Lloyd clay loam (fine, kaolinitic, thermic Rhodic Kanhapludults), Louisa loam (loamy, micaceous, thermic, shallow Ruptic-Ultic Dystrudepts), and Hiwassee loam (very-fine, kaolinitic, thermic Rhodic Kanhapludults). Site and study design descriptions can also be found in Tyree and others (2008) and King and others (2008).

Study Design

A split-plot experimental design was installed with the whole plots being two levels of fertilization (with or without) and the split-plot factor being 25 clones. Whole-plot treatments were blocked and replicated four times. One ramet (experimental unit) of each clone was planted in each plot. Ramets were rooted cuttings planted on May 19, 2003, at a 3.0- by 2.5-m spacing. Clonal material donated by the Forest Biology Research Cooperative (University of Florida, Gainesville) was from the Loblolly Pine Lower Gulf Elite Breeding Population, which includes both Atlantic coastal and Florida provenances. A border row of open-pollinated seedlings was planted around each plot. Trenches were dug and lined with plastic between plots to contain the fertilizer treatment. Site preparation prior to planting included application of glyphosphate (Round-up®) for weed control. The site was subsequently ripped and the planting rows were shallowly cultivated. After planting, complete weed control was maintained for the first 2 years. Fertilizer was applied by hand-banded application on May 4, 2004, and May 4, 2006. Each application consisted of 224 kg/ha of diammonium phosphate and 184 kg/ha of ammonium nitrate, yielding 112 kg/ha of elemental nitrogen and 53 kg/ha of elemental phosphorous.

Measurements

The trial has been measured annually since planting for height, diameter, crown width, and a variety of stem-quality metrics, such as sinuosity. Stem volume was calculated based on Burkhart (1977). Crown volume was calculated as a cone ($0.33\ \pi\ r^2\ h$) where radius was crown radius for a representative whorl at breast height and height was crown height. Stem sinuosity was scored in classes as follows: 1 for a straight stem, 2 for evident sinuosity that was not severe, 3 for severe sinuosity that could affect stem quality, and 4 for extremely severe sinuosity that would affect stem quality. Trees were categorized as either forked or not forked. Branch angle for the average of all branches on a representative whorl near breast height was scored in classes as follows: 1 for zero to 15 degrees from horizontal, 2 for 15 to 30 degrees, 3 for 30 to 45 degrees, and 4 for >45 degrees.

Statistics

Data presented in this paper are all from winter 2007 to 2008, and thus are for the first five growing seasons. Analyses were done in SAS 9.2 (SAS Institute Inc., Cary, NC) using PROC MIXED and PROC GLIMMIX with appropriate split-plot error structures (block and block * fertilizer were random effects). Continuous data were modeled with a normal distribution, score data with a Poisson distribution, and presence-absence data with a binary distribution. Regression of stem volume vs. crown volume was performed with PROC REG.

RESULTS AND DISCUSSION

Stem Volume

Both fertilizer and clone significantly affected stem-volume growth after five growing seasons in this trial. However, the clone by fertilizer interaction was not significant, as is summarized in table 1. Despite this lack of significance, it is clear from figure 1A that stem-volume response to fertilization does vary by clone. Figure 1B depicts only the four best performing clones from this trial based on fertilized stem volume. Hypothetically, were this to have been a fertilized clonal screening trial, these would be the clones selected and placed into production. It is evident from figure 1B though, that a landowner planting these clones on a marginal site, but not fertilizing them, would see markedly reduced stem-volume growth for clones B3 (63 percent of fertilized stem volume) and C2 (68 percent). On the other hand clones D2

Table 1—*P*-values from mixed models of a loblolly pine clone by fertilizer split-plot design on the Virginia Piedmont

Effect	Stem volume	Growth efficiency	Sinuosity	Forking	Branch angle
Fertilizer	0.053	0.072	0.092	0.423	0.991
Clone	<0.001	0.460	0.015	0.340	0.722
Clone * fertilizer	0.179	0.371	0.989	0.882	0.999

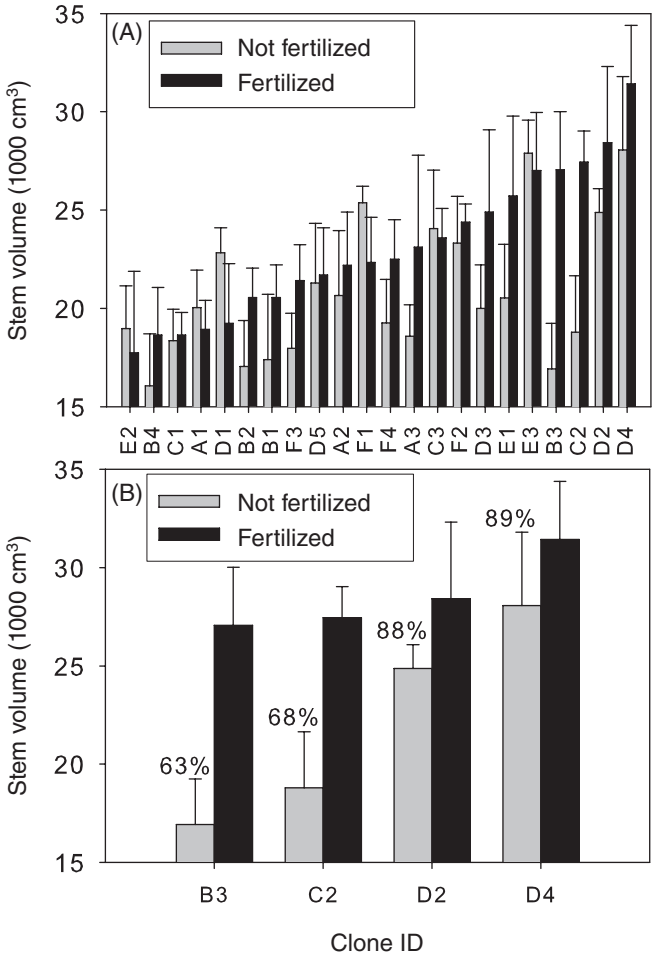

Figure 1—Volume response for the first five growing seasons of clonal loblolly pine to fertilization in the Virginia Piedmont. Panel A shows all 22 clones from the full trial, while panel B shows only the 4 best growers under fertilization. Clones identified with the same letter are full sibling to one another. Percentage values in panel B are for the volume of the unfertilized treatment relative to the fertilized treatment for each clone.

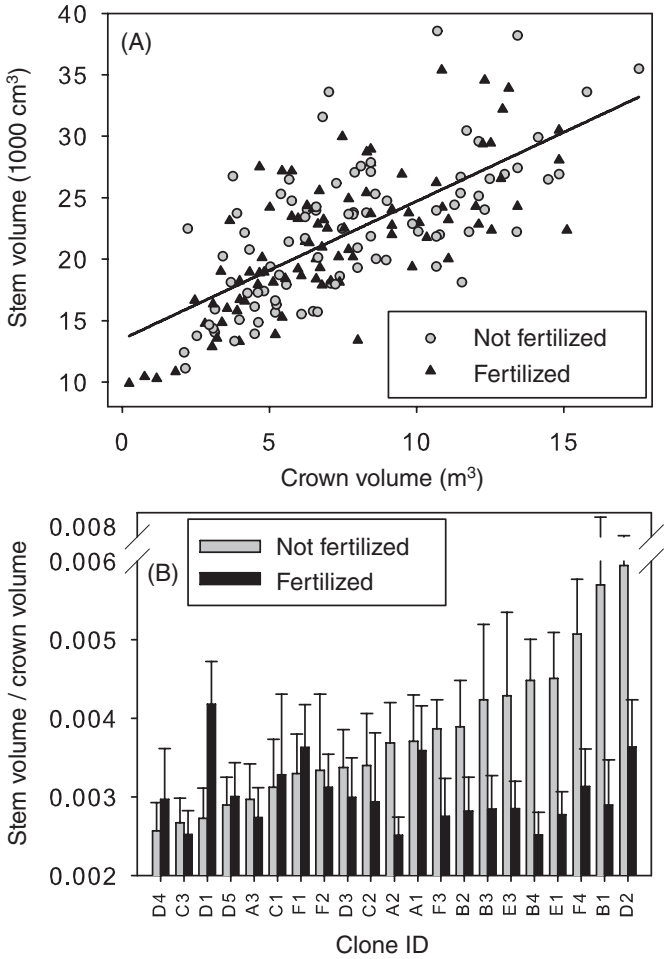

Figure 2—Growth efficiency response for the first five growing seasons of clonal loblolly pine to fertilization in the Virginia Piedmont. The regression in panel A is statistically significant ($P < 0.001$, $R^2 = 0.487$), indicating that greater leaf area is consistent with greater stem growth. Panel B depicts units of stem volume per units of crown volume, which can be interpreted as a metric of growth efficiency. Greater growth efficiency corresponds to a larger bar.

and D4 show only a moderate reduction in stem-volume growth when unfertilized (88 to 89 percent of fertilized stem-volume growth). While there is no statistically significant clone by fertilizer interaction among even these four clones ($P = 0.330$), it is evident that there would be a practically significant interaction to landowners planting, but not fertilizing, these specific genotypes.

Growth Efficiency

For the purposes of this paper we are defining growth efficiency as units of stem volume produced per units of crown volume. This metric may also be considered as a crown ideotype, indicating how large a crown is necessary to produce a given unit of wood. Crown volume is being used as an approximate surrogate for leaf area. Figure 2A shows that trees with greater crown volume (and thus leaf area) tend to have larger stem volumes as well. Greater leaf area corresponds to greater photosynthetic capacity. More carbon can be fixed, which can then be allocated to the stem. The wide variance in stem volumes produced at any given crown volume, as seen in figure 2A, may be due to differences between clones in carbon allocation patterns, although this dataset is not currently sufficient to support this hypothesis. While neither clone nor clone by fertilizer effects were significant (table 1) fertilizer did significantly reduce growth efficiency, as is seen in figure 2B. This most likely corresponds to greater production of needles as a result of fertilization leading to increased self-shading. As with stem volume, while the clone by fertilizer interaction is not statistically significant, there is a great deal of variability in growth efficiency response to fertilization between the different clones. Again, while there is not a statistically significant interaction, a range of crown ideotypes exist and depend on the specific clone selected, and whether or not fertilizer is applied as part of the silvicultural regime.

Stem Form

Sinuosity—Severe (class 3 or 4) sinuosity was present in 19.7 percent of trees after five growing seasons. Both fertilizer and clone significantly affected stem sinuosity (table 1). As expected, fertilization increases growth rates, which generally leads to an increased incidence and severity of stem sinuosity. However, while the clone by fertilizer interaction was not statistically significant, figure 3A shows clear variability in sinuosity severity in response to fertilization between clones. Some clones are never sinuous whether fertilized or unfertilized, others are highly sinuous regardless of fertilization, and yet others are sinuous only when fertilized. It should be noted that only class 4 sinuosity is likely to be evident in sawtimber-sized logs, and that lesser classes will not likely affect marketability at rotation age.

Forking—Forking affected 18.1 percent of trees by the fifth-growing season. Even relatively minor forks that may have only a minimal effect on wood quality at rotation age were included in this data. Neither the two main effects nor the interaction significantly affected the frequency of stem forking. However, as with sinuosity there do appear to be differences between clone-specific fertilizer responses with respect to forking (fig. 3B). Some clones do not appear to fork, while others fork frequently. Still others do not fork until fertilized and then show an increase in the frequency of forking.

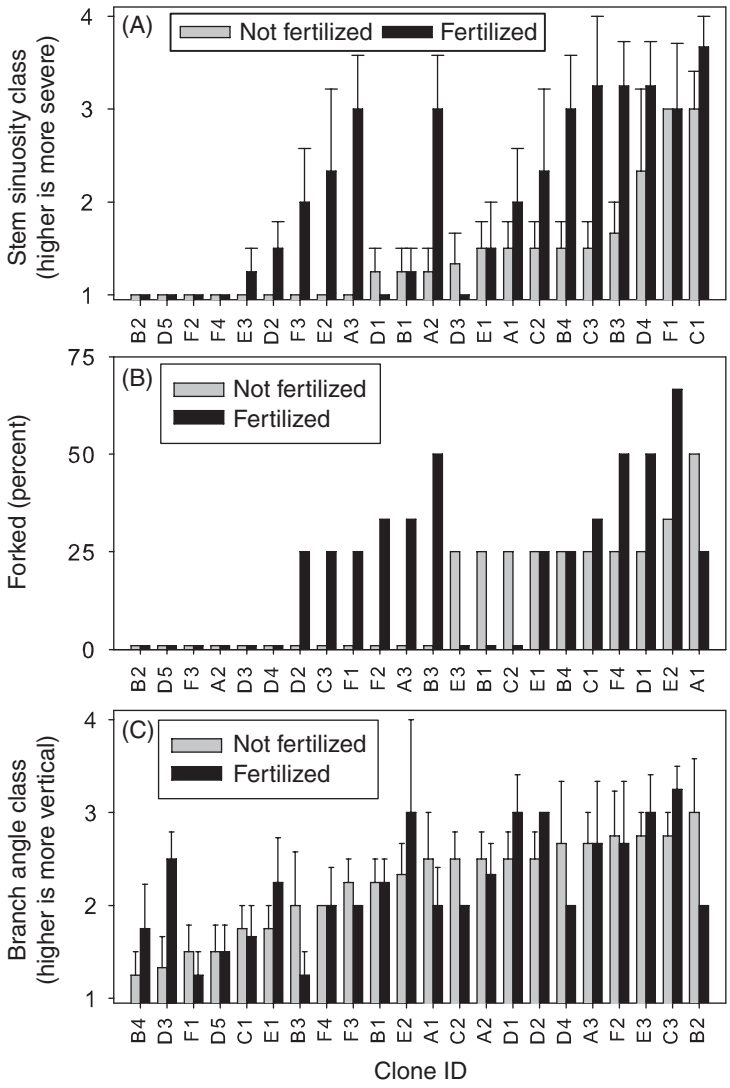

Figure 3—Stem quality response for the first five growing seasons of clonal loblolly pine to fertilization in the Virginia Piedmont. Clones identified with the same letter are full sibling to one another. Percentage values in panel B represent the proportion of ramets that forked for each clone by fertilizer treatment.

Practically speaking, a clone that is discarded from a fertilized screening program for forking may show a low incidence of forking on marginal, unfertilized sites, and may be an appropriate selection for these sites under this silvicultural regime.

Branch Angle—Steeper branch angles cause larger knots in the main stem, reducing wood quality. Branch angle was not significantly affected by fertilizer or clone effects, or their interaction. While there is some variability between clones that is apparent in figure 3C, the magnitude of these effects is less than those previously presented, and may not be of practical concern.

Nonsignificant *P*-Values and Clonal Screening

While the *P*-values for clone by fertilizer interactions depicted in table 1 were not statistically significant for any variables measured, there may still be a practical concern for clonal screening programs. Figure 4 shows that, even within this small field trial, statistically significant clone by fertilizer interactions for stem volume do exist for some randomly selected groups of clones. Figure 4 was generated by randomly selecting different subsets of clones from the full trial and calculating the *P*-value for the clone by fertilizer interaction. It reveals that about 33 percent of the groups randomly selected showed significant clone by fertilizer interactions ($P < 0.100$). Because clonal screening programs choose a very small number of clones to put into production from extremely large clonal trials, an overall lack of significant clone by fertilizer interaction does not necessarily indicate that there will be no significant interaction among the small group selected.

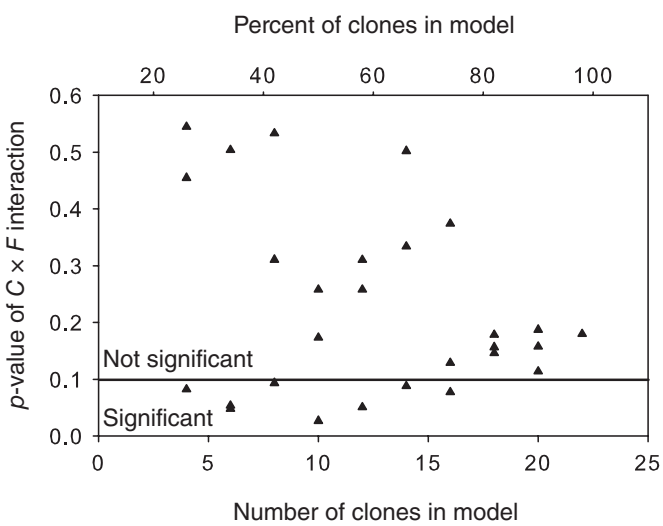

Figure 4—*P*-values of the clone by fertilizer interaction for stem volume for different groups of clones from the full trial. The *P*-value of all 22 clones is the furthest point to the right, while all points to the left were randomly selected groups of clones from the full trial. Approximately 33 percent of randomly selected groups of clones showed a statistically significant ($P < 0.100$) interaction, as denoted by the horizontal line.

CONCLUSION

Clonal forestry offers the opportunity to select specific individuals that possess a number of different favorable characteristics and plant them across large acreages. However, the results presented in this paper indicate that many important traits may vary between clones, particularly with respect to fertilizer response. Clones screened under a uniform silvicultural system may show different stem volume, growth efficiency, and stem form responses when grown under a different silvicultural system. While clone by fertilizer interactions were not statistically significant for any of these traits based on this trial, graphical evidence and the arguments presented above support that they may nonetheless be of practical concern to both companies producing and testing clones, and landowners who are buying and planting clones.

ACKNOWLEDGMENTS

We'd like to thank the Forest Biology Research Cooperative for donating the clonal material used in this trial. Debbie Bird and Clay Sawyers both helped with measurements. Some of the funding for this project was provided through the NSF Center for Advanced Silvicultural Systems at Virginia Tech.

LITERATURE CITED

Burkhart, H.E. 1977. Cubic-foot volume of loblolly pine to any merchantable top limit. Southern Journal of Applied Forestry. 1: 7–9.

Dougherty, D. 2007. Improved returns on forestlands: a financial analysis of MCP and varietal seedlings on private land in the Southeastern United States. Forest Landowner. 66: 15–18.

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.

Frampton, J.; Li, B.; Goldfarb, B. 2000. Early field growth of loblolly pine rooted cuttings and seedlings. Southern Journal of Applied Forestry. 24: 98–105.

King, N.T.; Seiler, J.R.; Fox, T.R.; Johnsen, K.H. 2008. Post-fertilization loblolly pine clone physiology and growth performance. Tree Physiology. 28: 703–711.

MacKay, J.J.; Becwar, M.R.; Park, Y.S. [and others]. 2006. Genetic control of somatic embryogenesis initiation in loblolly pine and implications for breeding. Tree Genetics & Genomes. 2(1): 1–9.

McKeand, S.E.; Jokela, E.J. Huber, D.A. [and others]. 2006. Performance of improved genotypes of loblolly pine across different soils, climates, and silvicultural inputs. Forest Ecology and Management. 227(1-2): 178–184.

Nelson, C.D.; Johnsen, K.H. 2008. Genomic and physiological approaches to advancing forest tree improvement. Tree Physiology. 28(7): 1135–1143.

Roth, B.E.; Jokela, E.J.; Martin, T.A. [and others]. 2007. Genotype x environment interactions in selected loblolly and slash pine plantations in the Southeastern United States. Forest Ecology and Management. 238(1–3): 175–188.

Tyree, M.C.; Seiler, J.R.; Fox, T.R. 2008. The effects of fertilization on soil respiration in 2-year-old *Pinus taeda* L. clones. Forest Science. 54(1): 21–30.

World_Climate. 2008. Climate data for 36°N 80°W. http://www.worldclimate.com/cgi-bin/grid.pl?gr=N36W080. [Date accessed July 8].

SITE-SPECIFIC FOREST MANAGEMENT: MATCHING GENOTYPES AND SILVICULTURE TO OPTIMIZE CARBON SEQUESTRATION

Michael Tyree, John Seiler, and Chris Maier[1]

Abstract—The use of improved genotypes as well an increased understanding of the role of intensive silviculture have made southeastern pine forests some of the most productive forests in the world. The objectives of this research were to determine how two superior loblolly pine (*Pinus taeda*) genotypes, representing two distinct ideotypes, respond to manipulations of nutrient availability. Second, based on estimates of carbon (C) capture and loss, predict how treatment responses may influence net ecosystem productivity (NEP). A combination of nitrogen (N) and phosphorus fertilization and the incorporation of high C to N logging residue (LR) provided a range of nutrient availability. We found both clones increased aboveground growth in response to fertilization, but to different degrees. Additionally, there were large differences in aboveground biomass partitioning in response to LR incorporation between clones. Finally, there were significant clonal differences in soil CO_2 efflux indicating that there may be strong differences in NEP between genotype and nutrient availability.

INTRODUCTION

Southern pine plantations in the Southeastern United States are some of the most intensively managed and highly productive forested systems in the world (Allen and others 2005). Planted pine forests may create great opportunity for sequestering large amounts of carbon (C) (Johnsen and others 2001). Currently, southern pine plantations occupy more than 13 million ha and are forecast to increase 67 percent to 22 million ha by the year 2040 (Wear and Greis 2002). Intensive management of pine forests has been shown to affect net ecosystem C exchange by decreasing the time necessary for a stand to shift from a C source to a C sink (Lai and others 2002; Maier and others 2004, 2002; Sampson and others 2006). Net primary productivity (NPP) and heterotrophic respiration (R_h) are two opposing processes that contribute to net ecosystem productivity (NEP). Forest management such as nutrient additions, competition control, and site preparation most dramatically impact NEP by increasing NPP (Albaugh and others 2004, Maier and others 2004) but has also been shown to suppress the rate of soil organic matter decomposition (Fog 1988) either through reduced microbial activity (Blazier and others 2005, Gough and Seiler 2004, Homann and others 2001, Olsson and others 2005, Tyree and others 2008), decreased microbial biomass (Bååth and others 1981, Lee and Jose 2003, Thirukkumaran and Parkinson 2000, Tyree 2008), changes in microbial population composition (Bittman and others 2005, Lilleskov and others 2002), or some combination of these factors.

A typical harvesting operation in a southern pine stand can generate up to 50 t of logging debris/ha (Allen and others 2006), which historically has been gathered into large piles and burned or abandoned. These represent huge stores of organic C, which left exposed to the air will largely oxidize being released back into the atmosphere as CO_2. Forest managers routinely spread this logging debris back onto the site in an attempt to spread nutrients across a site as well as minimize disturbance from trafficking. Further, the idea has been proposed to incorporate this logging residue (LR) into the soil (see review by Johnson and Curtis 2001). Not only would this provide nutrients to successive stands, but it may additionally increase soil C sequestration as it is likely some fraction will remain as recalcitrant soil C. Agricultural studies have long shown that adding large amounts of C into the mineral soil will also impact nutrient cycling and decomposition rates of the material (Holland and Coleman 1987). Depending on the availability of organic C to decomposition and the ratio of C to N, there is both the potential to increase longer term nutrient cycling, particularly N, phosphorus (P), and sulfur. However, incorporating logging residue into the soil following harvesting may result in the acceleration of microbial decomposition of the labile C pool leading to an increase in microbial populations (Aggangan and others 1999, Ouro and others 2001) and the short-term immobilization of essential nutrients such as N (Aggangan and others 1999, Perez-Batallon and others 2001), which may result in decreased tree growth.

An exciting area of research that shows huge potential for increasing productivity of intensively managed forests is the use of superior planting stock. In fact, it has been estimated that increases in volume gains of 10 to 30 percent have been made possible as a result of selective breeding, and gains of 50 percent or more may be possible by combining the use of clones and intensive silviculture (Allen and others 2005, Martin and others 2005). With increased emphasis being placed on site-specific management there is a need to determine how specific genotypes will vary across environments (Fox 2000) as well as match specific genotypes to site conditions, e.g., resource availability.

The overall objective of this research is to monitor the short-term (1 to 3 years) effects of intensive silviculture on C pools

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and fluxes, involved in C capture, biomass partitioning, and C evolution back to the atmosphere, in young loblolly pine (*Pinus taeda*). Specifically, we wanted to determine how two superior loblolly pine genotypes, which represent two distinct ideotypes, respond to manipulations of nutrient availability. Secondly, based on estimates of C capture and loss predict how treatment responses may influence NEP. We hypothesize both genotypes will respond to increased N and P by increasing NPP, but to different extents. Additionally, the incorporation of LR would decrease growth in both clones, but effect "narrow crown" ideotype to a lesser extent. Finally, differences in growth and C allocation between clones will significantly impact the rate of C evolution from these soils as well as the short-term C balance.

MATERIAL AND METHODS

Study Site

The study site was located in Berkeley County, SC, at an elevation of 24 m above mean sea level. Average annual temperature was 14.6 °C and 17.4 °C with an average daily maximum of 17.3 °C and 25.2 °C and an average daily minimum of 11.7 °C and 11.2 °C for the 2006 and 2007 year, respectively. Highest daily average temperature was 26.8 °C and 32.5 °C occurring in August 2006 and August 2007, respectively, and a low of –0.9 °C and 0.4 °C occurring in December 2006 and February 2007, respectively. Total precipitation was 90.2 cm in 2006 and 74.9 cm in 2007 spread evenly throughout the year, which was well below the average of 120 cm recorded between 1949 and 1973 (Long 1980). The dominant soil series is Ocilla (loamy, siliceous, semiactive, thermic Aquic Arenic Paleudults). Harvest of the previous 21-year-old loblolly pine stand took place in May 2004, and the site was sheared of residual material in July 2004. LR treatments were applied in October 2004, and site preparation (bedding) took place in early November 2004. Loblolly pine clones were planted in January of 2005, and data for this study was collected between January 2006 and January 2008.

Experimental Design

The study design was a split-plot, randomized complete block design replicated three times with the whole-plot treatments arranged as a full 2 by 2 factorial, which was measured repeatedly. Each 0.18-ha plot (48 x 38 m) was planted with approximately 243 container-grown, clonal loblolly pine seedlings in 9 rows at a 1.8-m spacing within rows and a 4.3-m spacing between row centers. Two levels of LR and two clones (CL32 and CL93) served as the whole-plot treatments. The two levels of LR were no LR incorporated and LR incorporated into the mineral soil (LR) at a rate of 25 Mg oven-dry weight/ha, which was concentrated onto the beds (approximately 75 Mg oven-dry weight/ha; C to N = 700). Both LR treatments also incorporated the residual forest floor of approximately 25 Mg oven-dry weight/ha. The two loblolly pine clones chosen both exhibit superior height growth but represent two distinct ideotypes. Clone 93 (CL93, "narrow crown" ideotype) has been shown to allocate more of its resources to stem growth while clone 32 (CL32, "broad crown" ideotype) allocates more resources to leaf area. Each plot was split into two 0.0013-ha measurement plots, located at opposite ends of the whole plot; each consisted of six seedlings (four measurement trees + two buffer trees) and served as the experimental unit. Each split plot received one of two fertilizer applications: no nutrient additions or N and P fertilization (F). During the 2006 growing season fertilizer was applied twice and totaled 209 kg N and 116 kg P/ha in the form of diammonium phosphate and ammonium nitrate (AN). Roughly one-third was applied on April 6 and the remaining two-thirds applied on May 8, 2006. F for the 2007 growing season was applied on March 9, 2007, at a rate of 200 kg N/ha in the form of AN.

Measurements

In January 2008, a representative tree, which most closely fit the mean tree height from each plot, was harvested to estimate biomass partitioning. The aboveground portion of each seedling was cut 10 cm above groundline. Belowground plant tissues were sampled by excavating a 1- by 1- by 0.5-m volume around the main stem. At the lab each tree was dissected into foliage, branches, main stem, tap root, and lateral roots. All samples were oven dried (>2 weeks) at a temperature of 65±5 °C then weighed gravimetrically to the nearest gram.

Total soil CO_2 efflux (F_S) from the soil surface was estimated using a LI-6200 portable infrared gas analyzer (LI-COR Biosciences, Lincoln, NE) with a dynamic closed soil chamber giving a total system volume of 6300 cm^3 (Selig and others 2008, Tyree and others 2008). Soil respiration measurements were taken approximately every 1 to 1.5 months starting January 2006 through December 2007 (16 sampling dates). A broken chamber hose on February 2006 and machine leaks on May and June 2006 forced us to remove all 3 sampling dates leaving 13 separate dates for the experiment. Measurements were taken at approximately the same location on each date and in the same sequential blocking order between 0800 and 1600 hours and taking between 3 to 4 hours to complete. Two subsamples were taken per measurement plot. One measurement was taken at the base of the tree and the other taken between trees to account for spatial variation on the planting bed. No measurements were taken between planting rows. Soil CO_2 evolution was measured over a 30-second period and F_S rates calculated as µmols CO_2 m^2/second.

Carbon balance was estimated by calculating NPP in Mg C/ha over two growing seasons by converting total biomass to total C content by assuming a 0.5-conversion factor. R_h was roughly estimated from our measurements of F_S. Average F_S was determined for each plot (13 sampling dates by 2 locations) and converted to metric tons of carbon per hectare over the entire study. Based on the literature, R_h was assumed to be 50 percent of F_S for this experiment (Andrews and others 1999, Hanson and others 2000, Maier and Kress 2000).

Data Analyses

Treatment differences in F_S were determined using analysis of variance with repeated measures using a MIXED model.

Covariance structures were selected using AIC, AAIC, and BIC fit statistics included in the SAS output. Treatment differences in biomass partitioning and relative C budgets were analyzed using a general linear model (GLM). Residuals and the normality curves were plotted for all analyses to confirm that the data met assumptions of equal variance and normality for all parameters measured. When data were transformed by their natural log to meet assumptions, all values were expressed as untransformed least square means and standard errors. All analyses were performed using the MIXED and GLM procedures in SAS (SAS 2006).

RESULTS AND DISCUSSION

We observed significant differences for both above- and belowground plant biomass ($P = 0.002$ and $P = 0.08$, respectively) with our nutrient manipulation treatments (fig. 1). Consistent with other researchers' findings (Albaugh and others 1998, Gough and Seiler 2004), the addition of N and P fertilizer resulted in a 101- and 65-percent increase in above- and belowground biomass, respectively, at the end of the third-growing season relative to control treatments. LR additions increased aboveground biomass by 25 percent but had no effect on belowground biomass. Interestingly, when both fertilizer and LR were added there was only a modest increase in above- and belowground biomass (29 and 18 percent, respectively) indicating that the addition of LR resulted in increased competition for nutrients by soil microbes. Data from an accompanying field study showed that microbial biomass C and microbial activity (R_h) increased with the addition of LR for these same plots (Tisdale 2008, Tyree 2008).

In contrast to our hypothesis we found no significant ($P > 0.1$) difference between genotypes in above- and belowground biomass. However, upon closer investigation we observed differences in aboveground biomass partitioning with soil amendment treatments (fig. 2). For example, plots receiving LR resulted in a 47-percent increase in foliar to stem ratio in CL32 and no change in CL93 ($P = 0.005$, fig. 2A). Plots receiving fertilizer showed the opposite trend with CL32 showing a decrease in foliar to stem ratio by 22 percent and CL93 showing a 19-percent increase ($P = 0.06$, fig. 2B).

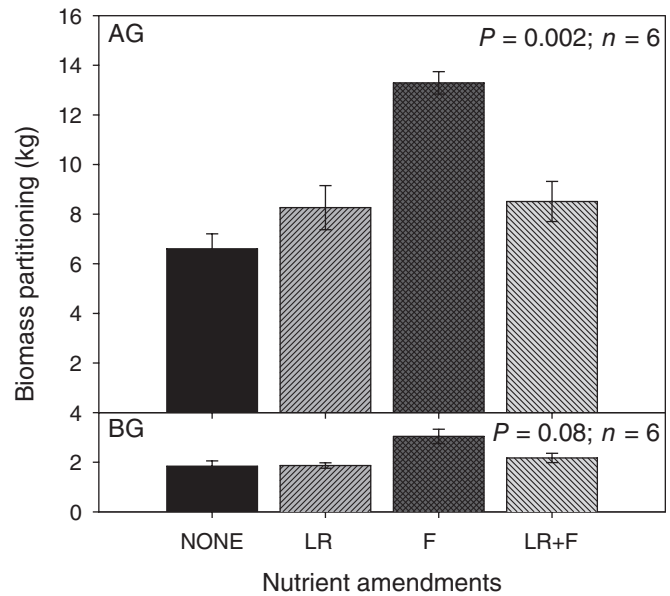

Figure 1—Mean above- (AG) and belowground (BG) biomass partitioning for logging residue (LR) and fertilization (F) nutrient manipulation treatments. Clonal loblolly pine seedlings were planted January 2005 in Berkley County, SC, and destructively harvested on January 2008.

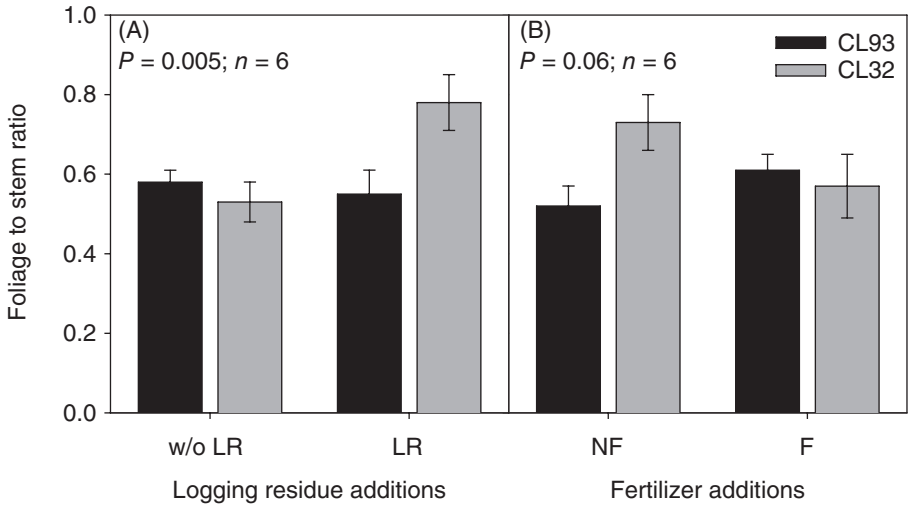

Figure 2—Mean foliage to stem ratio for (A) logging residue (LR) and (B) fertilization (F) treatments. Clonal loblolly pine seedlings were planted January 2005 in Berkley County, SC, and destructively harvested on January 2008. (w/o LR = no LR incorporated; NF = no nutrient additions.)

These data indicate that although there is no difference between genotypes in total aboveground biomass across a fertility gradient there are differences in how that aboveground tissue is distributed. Always keeping in mind that our primary objective is to produce forest products and that C sequestration will be secondary, and by planting genotypes that preferentially allocate to stem biomass we are able to optimize both these objectives.

In support of our hypothesis we observed a significant (P = 0.03) clone by LR by fertilizer three-way interaction when data from all 13 sampling dates were analyzed. In control treatments (none) F_S was substantially greater in CL32 than CL93 (fig. 3). This was supported by minirhizotron data from a project collaborator which showed that CL32 maintained greater fine-root (<2 mm) length then CL93.[2] Both fertilizer and LR additions resulted in a decreased F_S in CL32 (fig. 3). We found that the addition of LR in CL32 reduced stem volume and belowground biomass by 25 and 30 percent, respectively, in CL32 and not at all in CL93 (Tyree 2008). In contrast, fertilizer had only a slight increase on F_S in CL93 while LR additions resulted in a much more extreme increase in CL32. Finally, the addition of both LR and fertilizer showed no detectable difference in F_S in either genotype.

In most nutrient manipulation treatments we found that the C balance between clones did not differ except in the control and LR+F treatments (table 1). When no amendments were added (none) CL32 maintained a slightly greater C balance then CL32, but when both LR and fertilizer were applied CL93 maintained a greater C balance. We found that the C balance over the first two full growing seasons was negative

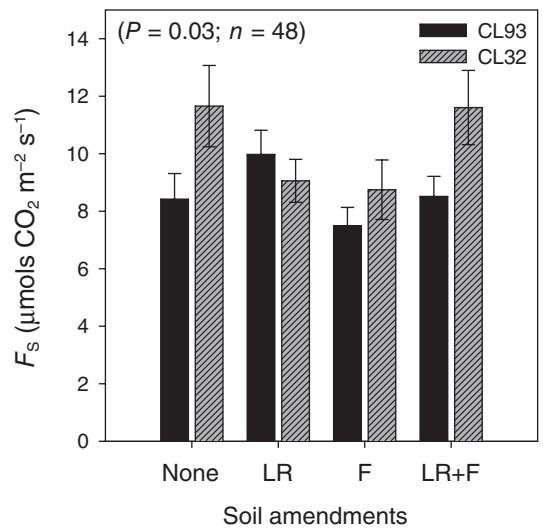

Figure 3—Total soil CO_2 efflux from the soil surface (F_S) averaged over 13 sampling dates from January 2006 through December 2007. Measurements were taken using a closed system portable infrared gas analyzer (LI-6200). (LR = logging residue; F = fertilization.)

Table 1—Average (standard error) carbon budgets for soil amendment treatment by clone. Values are estimated from January 2006 through January 2008. Positive numbers indicate a carbon sink and negative numbers indicate carbon source.

	Soil amendment added			
Response	None	LR added	F added	LR+F
	t carbon/ha			
	Clone 32 ("broad crown" ideotype)			
Net primary productivity	6.24 (0.33) b	6.47 (0.96) b	11.11 (0.57) a	6.17 (0.54) b
Heterotrophic respiration	–35.72 (1.17) a	–33.25 (6.59) a	–28.82 (2.31) a	–38.05 (5.43) a
LR incorporation	0	38	0	38
Carbon balance	–29.49 (1.00) a	11.21 (7.40) b	–17.72 (2.84) a	6.12 (5.05) b
	Clone 93 ("narrow crown" ideotype)			
Net primary productivity	5.16 (1.05) c	7.21 (1.05) bc	10.94 (0.51) a	8.25 (0.85) ab
Heterotrophic respiration	–29.54 (5.23) a	–35.80 (6.33) a	–25.22 (2.68) a	–30.16 (2.65) a
LR incorporation	0	38	0	38
Carbon balance	–24.38 (4.75) a	9.41 (6.37) b	–14.27 (3.12) a	16.09 (1.82) b

LR = logging residue; F = fertilization.

Different letters in a row indicate significant differences between soil amendment treatments using Fisher's LSD (alpha = 0.05, n = 3).

[2] Personal communication. Seth Pritchard, College of Charleston, Department of Biology, Charleston, SC.

when LR was not applied. This is consistent with findings that early on these stands act as a C source due to the relatively low NPP and higher soil respiration following site disturbance and as these stands age increased NPP and decreased soil respiration increase NEP (Gough and others 2005, Sampson and others 2006). Under both clones the addition of fertilizer increases C balance relative to control treatments which is consistent with the findings of other studies (Lai and others 2002; Maier and others 2002, 2004) . Finally, the addition of LR resulted in a positive C balance early in the rotation indicating this may be a useful treatment in increasing C sequestration, but more long-term observation is needed.

CONCLUSION

In support of our findings we found that our nutrient manipulation treatments had a significant effect on plant biomass. Although there was no treatment by genotype interaction with plant biomass there were strong differences in the way aboveground biomass was partitioned between genotypes. The addition of LR resulted in a substantial increase in the foliage to stem ratio in CL32 but not CL93. Implication of this finding may be that clones can be chosen based on site nutrient availability to optimize stem production. Second, we found genetic by nutrient availability interactions in F_S, which suggests that future C models may need to account for genotypes as clonal forestry becomes more popular. Finally, NPP was the dominant factor controlling C balance. F increased NPP, therefore, increasing the C balance perhaps decreasing the time necessary for this stand to function as a C sink. Additionally, we found that incorporation of LR resulted in a positive C balance early in stand development and may be a potential treatment in increasing C sequestration. However when selecting genotypes to plant, biomass partitioning patterns need to be accounted for in addition to NPP to satisfy both timber production and C sequestration objectives.

ACKNOWLEDGMENTS

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LITERATURE CITED

Aggangan, R.T.; O'Connell, A.M.; McGrath, J.F.; Dell, B. 1999. The effects of *Eucalyptus globulus* Labill. leaf letter on C and N mineralization in soils from pasture and native forest. Soil Biology & Biochemistry. 31(11): 1481–1487.

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(1): 3–19.

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(2): 317–328.

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.

Allen, H.L.; Fox, T.R.; Maier, C.A.; McKeand, S. 2006. Site carbon sequestration and intensive pine plantation management. Critical processes and properties regulating carbon cycling in southern forests. Asheville, NC: Southern Forest Research Partnersship, Inc.; U.S. Department of Agriculture Forest Service, Southern Research Station. [Number of pages unknown].

Andrews, J.A.; Harrison, K.G.; Matamala, R.; Schlesinger, W.H. 1999. Separation of root respiration from total soil respiration using carbon-13 labeling during free-air carbon dioxide enrichment (FACE). Soil Science Society of America Journal. 63(5): 1429–1435.

Bååth, E.; Lundgren, B.; Soderstrom, B. 1981. Effects of nitrogen-fertilization on the activity and biomass of fungi and bacteria in a podzolic soil. Zentralblatt Fur Bakteriologie Mikrobiologie Und Hygiene I Abteilung Originale C-Allgemeine Angewandte Und Okologische Mikrobiologie. 2(1): 90–98.

Bittman, S.; Forge, T.A.; Kowalenko, C.G. 2005. Responses of the bacterial and fungal biomass in a grassland soil to multi-year applications of dairy manure slurry and fertilizer. Soil Biology & Biochemistry. 37(4): 613–623.

Blazier, M.A.; Hennessey, T.C.; Deng, S.P. 2005. Effects of fertilization and vegetation control on microbial biomass carbon and dehydrogenase activity in a juvenile loblolly pine plantation. Forest Science. 51(5): 449–459.

Fog, K. 1988. The effect of added nitrogen on the rate of decomposition of organic-matter. Biological Reviews of the Cambridge Philosophical Society. 63(3): 433–462.

Fox, T.R. 2000. Sustained productivity in intensively managed forest plantations. Forest Ecology and Management. 138(1-3): 187–202.

Gough, C.M.; Seiler, J.R. 2004. Belowground carbon dynamics in loblolly pine (*Pinus taeda*) immediately following diammonium phosphate fertilization. Tree Physiology. 24(7): 845–851.

Gough, C.M.; Seiler, J.R.; Wiseman, P.E.; Maier, C.A. 2005. Soil CO_2 efflux in loblolly pine (*Pinus taeda* L.) plantations on the Virginia Piedmont and South Carolina Coastal Plain over a rotation-length chronosequence. Biogeochemistry. 73(1): 127–147.

Hanson, P.J.; Edwards, N.T.; Garten, C.T.; Andrews, J.A. 2000. Separating root and soil microbial contributions to soil respiration: a review of methods and observations. Biogeochemistry. 48(1): 115–146.

Holland, E.A.; Coleman, D.C. 1987. Litter placement effects on microbial and organic-matter dynamics in an agroecosystem. Ecology. 68(2): 425–433.

Homann, P.S.; Caldwell, B.A.; Chappell, H.N. [and others]. 2001. Douglas-fir soil C and N properties a decade after termination of urea fertilization. Canadian Journal of Forest Research. 31(12): 2225–2236.

Johnsen, K.H.; Wear, D.; Oren, R. [and others]. 2001. Carbon sequestration and southern pine forests. Journal of Forestry. 99: 14–21.

Johnson, D.W.; Curtis, P.S. 2001. Effects of forest management on soil C and N storage: meta analysis. Forest Ecology and Management. 140(2-3): 227–238.

Lai, C.T.; Katul, G.; Butnor, J. [and others]. 2002. Modeling the limits on the response of net carbon exchange to fertilization in a south-eastern pine forest. Plant Cell and Environment. 25(9): 1095–1119.

Lee, K.H.; Jose, S. 2003. Soil respiration, fine root production, and microbial biomass in cottonwood and loblolly pine plantations along a nitrogen fertilization gradient. Forest Ecology and Management. 185(3): 263–273.

Lilleskov, E.A.; Fahey, T.J.; Horton, T.R.; Lovett, G.M. 2002. Belowground ectomycorrhizal fungal community change over a nitrogen deposition gradient in Alaska. Ecology. 83(1): 104–115.

Long, B.M. 1980. Soil survey of Berkeley County, South Carolina. [Place of publication unknown]: U.S. Department of Agriculture, Soil Conservation Service and Forest Service. 94 p. In cooperation with: South Carolina Land Resources Conservation Commission and South Carolina Agricultural Experiment Station.

Maier, C.A.; Albaugh, T.J.; Allen, H.L.; Dougherty, P.M. 2004. Respiratory carbon use and carbon storage in mid-rotation loblolly pine (*Pinus taeda* L.) plantations: the effect of site resources on the stand carbon balance. Global Change Biology. 10(8): 1335–1350.

Maier, C.A.; Johnsen, K.H.; Butnor, J. [and others]. 2002. Branch growth and gas exchange in 13-year-old loblolly pine (*Pinus taeda*) trees in response to elevated carbon dioxide concentration and fertilization. Tree Physiology. 22(15-16): 1093–1106.

Maier, C.A.; Kress, L.W. 2000. Soil CO_2 evolution and root respiration in 11 year-old loblolly pine (*Pinus taeda*) plantations as affected by moisture and nutrient availability. Canadian Journal of Forest Research. 30(3): 347–359.

Martin, T.A.; Dougherty, P.M.; McKeand, S.E. 2005. Strategies and case studies for incorporating ecophysiology into southern pine tree improvement programs. Southern Journal of Applied Forestry. 29(2): 70–79.

Olsson, P.; Linder, S.; Giesler, R.; Hogberg, P. 2005. Fertilization of boreal forest reduces both autotrophic and heterotrophic soil respiration. Global Change Biology. 11(10): 1745–1753.

Ouro, G.; Perez-Batallon, P.; Merino, A. 2001. Effects of silvicultural practices on nutrient status in a *Pinus radiata* plantation: nutrient export by tree removal and nutrient dynamics in decomposing logging residues. Annals of Forest Science. 58(4): 411–422.

Perez-Batallon, P.; Ouro, G.; Macias, F.; Merino, A. 2001. Initial mineralization of organic matter in a forest plantation soil following different logging residue management techniques. Annals of Forest Science. 58(8): 807–818.

Sampson, D.A.; Waring, R.H.; Maier, C.A. [and others]. 2006. Fertilization effects on forest carbon storage and exchange and net primary production; a new hybrid process model for stand management. Forest Ecology and Management. 221: 91–109.

SAS. 2006. SAS online. Version 9. Cary, NC: SAS Institute Inc.

Selig, M.F.; Seiler, J.R.; Tyree, M.C. 2008. Soil carbon and CO_2 efflux as influenced by the thinning of loblolly pine (*Pinus taeda* L.) plantations on the Piedmont of Virginia. Forest Science. 54(1): 58–66.

Thirukkumaran, C.M.; Parkinson, D. 2000. Microbial respiration, biomass, metabolic quotient and litter decomposition in a lodgepole pine forest floor amended with nitrogen and phosphorous fertilizers. Soil Biology & Biochemistry. 32(1): 59–66.

Tisdale, J.L. 2008. Quantifying the effects of organic residues on soil nitrogen and phosphorus availability. Raleigh, NC: North Carolina State University, Department of Forestry. [Number of pages unknown]. M.S. thesis.

Tyree, M.C. 2008. Genetics by nutrient availability interactions on short-term carbon pools and fluxes in young *Pinus taeda* plantations. Blacksburg, VA: Virginia Tech. [Number of pages unknown]. Ph.D. dissertation.

Tyree, M.C.; Seiler, J.R.; Fox, T.R. 2008. The effects of fertilization on soil respiration in two-year-old *Pinus taeda* L. clones. Forest Science. 54(1): 21–30.

Wear, D.N.; Greis, J.G. 2002. Southern forest resource assessment: summary report. Gen. Tech. Rep. SRS-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 103 p.

FIRE AND FUELS MANAGEMENT

Prescribed fire in Compartment 1 on the Crossett Experimental Forest, Ashley County, Arkansas, in April 2010. (Photo by James M. Guldin)

UNDERSTORY FUEL VARIATION AT THE CAROLINA SANDHILLS NATIONAL WILDLIFE REFUGE: A DESCRIPTION OF CHEMICAL AND PHYSICAL PROPERTIES

Evelyn S. Wenk, G. Geoff Wang, and Joan L. Walker[1]

Abstract—Upland forest in the Carolina Sandhills National Wildlife Refuge is characterized by a longleaf pine (*Pinus palustris*) canopy with a variable understory and ground-layer species composition. The system was historically maintained by fire and has been managed with prescribed fire in recent decades. A management goal is to reduce turkey oak (*Quercus laevis*) stem density and maintain the natural biodiversity in the understory. The patchy understory of this forest type creates several fuel complexes on a small within-stand scale. We measured chemical content (energy and ash) of five common species, identified three fuel complexes based on dominant vegetation and fuels [longleaf pine litter, turkey oak, and wiregrass (*Aristida stricta*)], and described the fuels present in each fuel complex. Longleaf pine litter contained the highest energy (21,716 J/g) and little bluestem (*Schizachyrium scoparium*) the lowest (19,202 J/g). Among the fuel complexes, turkey oak-dominated sites had the highest potential fuel weight (12.4 tons/ha) and wiregrass-dominated sites the lowest (6.9 tons/ha). Both turkey oak- and wiregrass-dominated sites had a more aerated fuel bed than longleaf pine litter-dominated sites. We concluded that the plant community structure creates different fuel conditions, suggesting that fires will burn heterogeneously, creating spatial diversity in postfire conditions.

INTRODUCTION

Fire is an integral part of the longleaf pine (*Pinus palustris*) ecosystem in the Southeastern United States, with 3- to 5-year fire intervals found in areas where pine litter accumulation is sufficient (Christensen 1981, Platt 1999). Fires at such intervals encourage the regeneration of longleaf pine and many herbaceous species (Brockway and Lewis 1997, Christensen 1981). Consequently, land managers throughout the Southeast commonly use fire as a management tool. At the Carolina Sandhills National Wildlife Refuge (CSNWR), controlled burning is used to reduce fuel loads, maintain an open understory, and encourage longleaf pine regeneration.

The primary ecosystem at the CSNWR is the xeric sandhills ecosystem, an upland longleaf pine forest characterized by a longleaf pine overstory, turkey oak (*Quercus laevis*) in the shrub and midstory, and a wiregrass (*Aristida stricta*) and mixed forb ground layer. However, plant cover has changed in the past two centuries due to land and resource use, resulting in decreased pine and grass cover and increased hardwood cover (Christensen 1981), and accordingly, new fuel complexes that may change the way fire behaves within the ecosystem.

Understanding how fuels affect fire behavior and desired fire effects is a necessary component of using controlled burns (Johnson and Miyanishi 1995). Fire behavior is affected by both the chemical (intrinsic) and physical (extrinsic) properties of a fuel (Pyne 1984). The chemical property of a fuel is best described by its energy and mineral (ash) contents, where the energy content affects the amount of heat released and the mineral content affects the ignitability of the fuel. Physical properties include fuel load, morphology, and arrangement. Several studies have found differences in chemical properties among fuel species within an ecosystem (Dickinson and Kirkpatrick 1985, Dimitrakopoulos and Panov 2001) and among ecosystems (Dickinson and Kirkpatrick 1985, Golley 1961), as well as between native and nonnative invasive species (Dibble and others 2007, Lippincott 2000). Several of these studies couple chemical analysis with laboratory or field studies on fire behavior to suggest that intrinsic properties are related to other measures of flammability, such as rate of flame movement. Moreover, there is evidence that chemical properties of species in fire-prone environments are more likely to encourage fire (Mutch 1970). The dependence of the longleaf pine ecosystem on fire provides an ideal setting for further studies on fuel chemistry.

The patchy distribution of vegetation in the longleaf pine ecosystem creates numerous fuel complexes on a small within-stand scale. Previous studies on fuel variation in the sandhills concentrated on the heterogeneity in pine canopy cover and pine fuel loading. Small-scale variation in pine fuels was shown to affect fire intensity and shrub abundance, with areas of increased fuel loading causing higher temperature burns and increased shrub mortality (Thaxton and Platt 2006). Several studies looked at the effect of distance to nearest pine, pine density, or canopy species on fire temperature and turkey oak mortality (Platt and others 1991, Rebertus and others 1989, Williamson and Black 1981). Fuel arrangement and architecture, such as the positions of oak leaves and longleaf pine needles in the fuel bed, may also affect fire behavior (Rebertus and others 1989, Williamson and Black 1981), but differences in fuel arrangement have not been measured in sandhills ecosystems. Even though the presence of varying fuel complexes on the small scale has been used for fire behavior and fire effects studies in

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the sandhills, a comprehensive description of those fuel complexes is still missing.

The importance of fire in maintaining the sandhills longleaf pine ecosystem, and the role of fuels in determining fire behavior, necessitate a better understanding of the chemical and physical properties of dominant fuels in the ecosystem. The objectives of this study were to: (1) quantify chemical properties (energy and ash content) for five common species (four native, one nonnative) found in the sandhills and (2) compare physical fuel properties (potential fuel weight, litter depth, and fuel arrangement) of three fuel complexes found within longleaf pine stands in the sandhills. A stronger understanding of fuel properties will improve our use of prescribed fire as a management tool, especially where altered understory species composition has led to new fuel complexes.

METHODS

Study Area

The study site was located in CSNWR (34.58° N, 80.23° W), which is situated on the fall line of the Upper Atlantic Coastal Plain in Chesterfield County, SC. Elevations for this area range from 70 m along Black Creek to 180 m on the highest ridges. Soils are well-drained sands of the Alpin-Candor series. Mean annual precipitation is 110 cm and mean annual temperature is 15.6 °C. Although several plant communities are found within CSNWR, all study sites were located in the upland longleaf pine-wiregrass community.

We established two experiments (one on fuel chemistry and one on the physical properties of fuels) to meet our two objectives and described the experimental design of each separately.

Fuel Chemistry

We used a complete block design with six blocks, and to account for variability among individuals, we sampled at least 10 individuals of each species per block, using composite samples for the analyses. In November 2007 we collected plant matter from five species [needle litter from longleaf pine, dead leaves of turkey oak, and leaves of wiregrass, little bluestem (*Schizachyrium scoparium*), and weeping lovegrass (*Eragrostis curvula*)] for chemical analysis. The first four species are common native species and are the dominant fuel in the system; the fifth is an exotic species, recently increasing in cover at CSNWR. Prior to lab analysis, we separated live and dead tissue for the three grass species (wiregrass, little bluestem, and weeping lovegrass) and analyzed each separately.

To prepare samples for analytical tests, we oven dried the plant matter at 65 °C for 48 hours and ground it to 60-mesh using a Thomas Model 4 Wiley® Mill (Thomas Scientific, Swedesboro, NJ). We measured energy content with an IKA® C 200 oxygen bomb calorimeter (IKA®, Wilmington, NC) in isoperibol mode, running three subsamples of each species per block. Subsample weights ranged from 0.8 to 1.3 g, depending on the species. We calibrated the calorimeter with certified benzoic acid to determine the heat capacity of the system. Energy content was measured in J/g.

Mineral ash analysis was performed by the Agricultural Service Laboratory at Clemson University. Ash content was determined by heating samples for 2 hours at 600 °C in a Thermo Scientific muffle furnace (Thermo Scientific, Barrington, IL). Two subsamples (each approximately 1 g) were analyzed for each sample.

Physical Descriptions of Fuels

In our second experiment we identified three fuel complexes according to the natural fuel distribution to describe physical differences in the variation of understory fuels. Fuel complexes were identified visually based on the dominant species. The first was dominated by longleaf pine needle litter, with live fuels nearly absent, the second by turkey oak stems and litter, and the third by wiregrass. Longleaf pine needle litter was present in all fuel complexes. The frequency of each fuel complex across the landscape was not determined.

We used a complete block design with seven blocks; four blocks contained one plot of each fuel complex, and three blocks had two plots of each. We selected sites with all three fuel complexes present and in close proximity to each other within each block. We installed 4- by 4-m plots for each fuel complex, because at this scale, fuels were relatively uniform; on a larger scale there would have been significant variation within a fuel treatment plot. All plots were located in areas with a mature longleaf pine canopy over Alpin soils. Finally, all sites were selected to have a similar burn history, with the last prescribed burn conducted in the spring of 2003 or 2004.

In February and March 2008 we sampled all plots to estimate potential fuel weight by measuring all standing vegetation (live and dead) <2 m in height and all ground litter. The fuel components measured for all fuel complexes included turkey oak stems, wiregrass plants, and litter (separated into needle litter, turkey oak litter, and unidentified fractions).

We destructively sampled turkey oak stems outside the study plots to construct height vs. weight regressions, which we used to estimate stem weights of all stems within plots. Separate regression analyses were used for live stems zero to 70 cm, live stems >70 cm, dead stems zero to 70 cm, and dead stems >70 cm. We sampled wiregrass plants outside the study plots to develop relationships between wiregrass plant basal area and weight. Wiregrass basal area was estimated by taking two perpendicular measurements of plant crown diameter. Basal area and plant weight were estimated for all plants within each plot. Litter weight was estimated from samples collected adjacent to study plots. Litter samples from a 1- by 1-m area were used to estimate the litter weight of the longleaf pine litter- and wiregrass-dominated plots, and samples from a 1- by 2-m area were used to estimate the litter weight of the turkey oak-dominated plots. We sorted the litter samples into three components: longleaf pine litter, turkey oak litter, and unidentified. Oven-dry weights were obtained for each fraction. We took litter depth measurements from

five points within each plot (plot center and 1 m in from each corner along diagonals). Pine and oak litter lodged in turkey oak stems were not considered in litter height measurements, though litter lodged in wiregrass plants was, due to its contact with the litter bed.

Dwarf huckleberry (*Gaylussacia dumosa*), little bluestem, splitbeard bluestem (*Andropogon ternarius*), and longleaf pine cones made up a minimal proportion of the fuels. Their presence was noted, but their weights were not estimated for the study.

Statistical Analysis

We used two-way analyses of variance using PROC GLM (SAS Institute Inc., Cary, NC) to test for significant differences among species for the chemical analysis, and among fuel complexes for the physical descriptions of fuels. For each model, a block term was included to account for location effects. For the analyses of physical fuel descriptions, weighting was used to account for the different number of plots in each block. We used least squares means tests in PROC GLM to make specific comparisons among species and fuel complexes, and linear contrasts to compare woody vs. grass species for energy and ash content.

RESULTS

Fuel Chemistry

Significant differences in energy content were found among the common fuel species ($P < 0.0001$), and mean energy values ranged from 19,202 J/g (little bluestem, live) to 21,716 J/g (longleaf pine needles) (table 1). Needle and leaf litter from the woody species contained more energy than the leaves from grass species ($P < 0.0001$). Of the native grass species, wiregrass had higher energy content than little bluestem, whereas the exotic grass species, weeping lovegrass, had a similar energy content as the dominant native grass species, wiregrass.

Mean ash content ranged from 1.36 percent (weeping lovegrass, dead) to 2.58 percent (little bluestem, live) (table 2). We found significant differences among species ($P < 0.0001$), although ash content of leaves and needles from woody species was not significantly different from that of leaves from grass species ($P = 0.4529$).

Physical Descriptions of Fuels

Mean potential fuel weight for each fuel complex was divided into five categories: turkey oak stems, wiregrass, longleaf pine litter, turkey oak litter, and unidentified litter (fig. 1). We found significant differences in total potential fuel weight among fuel complexes ($P = 0.0014$), with turkey oak-dominated plots containing greater fuel loads than longleaf pine litter- or wiregrass-dominated plots. For the individual fuel components, differences among fuel complexes were significant for turkey oak stems ($P < 0.0001$), wiregrass ($P = 0.0006$), and turkey oak litter ($P = 0.0002$). These differences in fuel components verified our fuel complex designations. For example, the turkey oak-dominated plots had the most turkey oak stems and litter, and the wiregrass-dominated plots had the highest wiregrass weights.

Litter depth for each fuel complex is shown in figure 2. Differences among fuel complexes were significant ($P = 0.0214$), with longleaf pine litter-dominated plots having a lower litter depth than either wiregrass- or turkey oak-dominated plots. A density-related measure of litter arrangement is shown in figure 3. The ratio of litter weight to

Table 1—Mean energy content of some sandhills species found on the Carolina Sandhills National Wildlife Refuge

	Energy content (J/g)	
Species	Mean	Standard error
Longleaf pine, needle litter	21,716 A	50.0
Turkey oak, leaf litter	20,389 B	64.6
Wiregrass, live	19,972 C	39.6
Weeping lovegrass, dead	19,658 D	92.8
Wiregrass, dead	19,587 D	103.4
Weeping lovegrass, live	19,578 D	55.4
Little bluestem, dead	19,277 E	97.1
Little bluestem, live	19,202 E	53.9

Test of significance is at the 95-percent confidence level; means with the same letter are not significantly different.

Table 2—Mean ash content of some sandhills species found on the Carolina Sandhills National Wildlife Refuge

	Ash content (percent)	
Species	Mean	Standard error
Weeping lovegrass, dead	1.36 A	0.199
Wiregrass, dead	1.48 AB	0.048
Longleaf pine, needle litter	1.72 BC	0.090
Wiregrass, live	1.97 CD	0.104
Weeping lovegrass, live	2.13 D	0.137
Turkey oak, leaf litter	2.14 D	0.062
Little bluestem, dead	2.53 E	0.195
Little bluestem, live	2.58 E	0.184

Test of significance is at the 95-percent confidence level; means with the same letter are not significantly different.

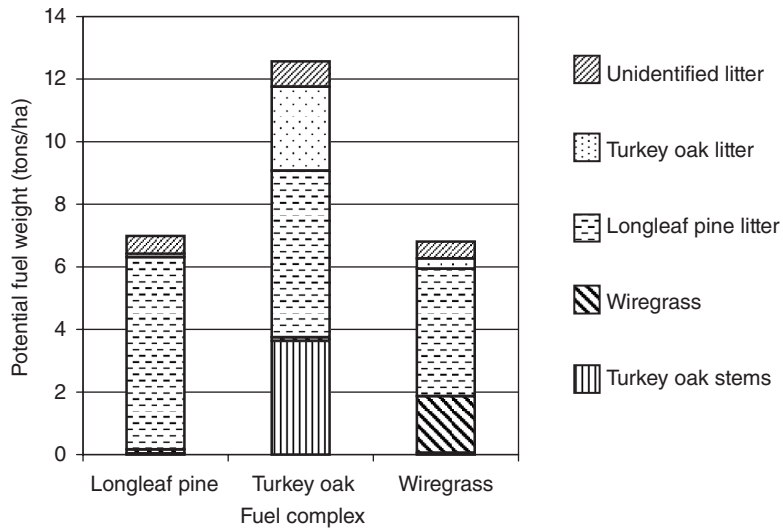

Figure 1—Effect of fuel complex on understory potential fuel weight (tons/ha), potential fuel weights are based on standing turkey oak stems <2 m tall, wiregrass plants, and all litter.

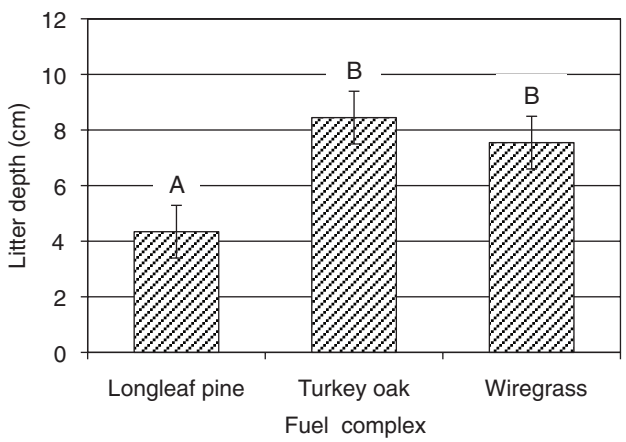

Figure 2—A summary of mean litter depth (cm) for three fuel complexes at the Carolina Sandhills National Wildlife Refuge. Different letters above bars show significant differences at a 95-percent confidence level.

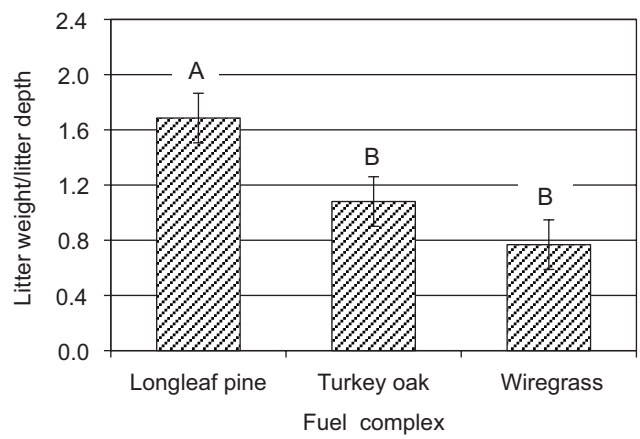

Figure 3—A measure of fuel bed aeration (the ratio of litter weight to litter depth) for three fuel complexes at the Carolina Sandhills National Wildlife Refuge. Different letters above bars show significant differences at a 95-percent confidence level.

litter depth was used to illustrate the aeration of the fuel bed. Longleaf pine litter-dominated plots had a higher weight to depth ratio than either turkey oak- ($P < 0.0001$) or wiregrass-dominated plots ($P < 0.0001$). No significant difference was found between turkey oak- and wiregrass-dominated plots ($P = 0.2331$).

DISCUSSION

Our analysis of energy content produced a range of values similar to those found in other studies from longleaf pine ecosystems (Golley 1961, Hough 1969) and within the range reported for other ecosystems (Dibble and others 2007, Dickinson and Kirkpatrick 1985, Dimitrakopoulos and Panov 2001). However, energy content for the same fuel type can vary by site (Hough 1969) and season (Golley 1961), and our values represent energy contents of sandhills species during the dormant season. It is possible that live tissue sampled midsummer could vary from that sampled in winter. Additionally, previous research has reported that energy content decreases as litter decomposes (Hough 1969). Because we sampled fresh litter for the woody species (longleaf pine and turkey oak), it is likely that litter deposited in the lower fuel bed may contain less energy than our reported values.

We expect that the energy content of fuels reported in this study play an important role in fire behavior when these fuels burn. The high energy content of longleaf pine needles supports previous field studies that report increased fire temperatures in areas with higher pine densities (Platt and

others 1991, Williamson and Black 1981). The difference in energy and ash content between wiregrass and little bluestem is notable because of potential implications on fire behavior. Wiregrass, with higher energy content and lower ash content, would be more readily flammable and would generate a higher heat output than little bluestem. These differences could result in an increased rate of spread and higher fire temperatures in areas dominated by wiregrass as opposed to areas dominated by little bluestem.

Our study sites for quantifying the physical properties of fuels differed in many ways from sites used in earlier studies on fuels and fire in the sandhills. Williamson and Black (1981) considered only oak-dominated areas without a pine canopy; at CSNWR, turkey oaks are found throughout areas under a longleaf pine canopy. Excluding pine canopy cover would not be representative of the system we studied. In fact, we found longleaf pine litter was present in similar quantities across the landscape, with other fuel components differentiating the three fuel classes.

Potential fuel weight varied from 4.5 to 18.5 tons/ha in our plots, falling on the lower end of 2.1 to 59.0 tons/ha reported by Thaxton and Platt (2006). The smaller range of our values is likely due to the exclusion of sites containing 100-hour or larger pine fuels. The decision to exclude large fuels was made to minimize the influence of confounding factors in later studies on fire intensity and behavior.

Plots dominated by turkey oak stems had the highest litter weights. In addition, measures of fuel arrangement indicated a well-aerated fuel bed of litter with relatively high energy content. These results suggested the potential for increased fire intensity and temperature in turkey oak-dominated sites compared to longleaf pine litter-dominated sites, contrasting with studies noting decreased flammability of oak litter and lower temperature burns near oaks as compared to pines (Rebertus and others 1989, Williamson and Black 1981). However, comparisons with those studies are difficult to make because oaks were isolated from pines, which is not the case at CSNWR.

The significant differences found in our description of litter arrangement confirmed previous speculations about variation in litter placement (Rebertus and others 1989, Williamson and Black 1981). We showed that wiregrass- and turkey oak-dominated sites have an aerated fuel bed compared to longleaf pine litter-dominated sites. Pine needle litter lodges in the vegetation, and turkey oak leaves curl and pack loosely, also catching pine needle litter in a more elevated position. Due to the lack of vegetation in the longleaf pine litter-dominated plots, a denser fuel bed was recognized where needles are packed horizontally. It is likely that, if given similar weights of fuel, a looser arrangement will increase air flow prior to and during burns, reducing the moisture content of the litter and increasing the rate of spread of the fire.

Heterogeneity in vegetation and litter on a small scale has been shown to affect species composition and abundance (Platt and others 2006) and fire effects such as hardwood mortality (Rebertus and others 1989, Thaxton and Platt 2006). Fire is used at CSNWR to reduce fuel loads, suppress hardwoods in the understory (especially turkey oak), and maintain the natural biodiversity of the system. The observed presence of different fuel complexes (confirmed by differences in fuel loading, composition, and arrangement) and chemical differences of the fuels suggests that prescribed fires will burn heterogeneously within longleaf pine stands. However, monitoring fire behavior and fire effects would be necessary to determine the specific responses of prescribed fire to fuel variation.

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: 167–183.

Christensen, N.L. 1981. Fire regimes in southeastern ecosystems. In: Mooney, H.A.; Bonnickson, T.M.; Christensen, N.L. [and others], eds. Fire regimes and ecosystem properties: Proceedings. Gen. Tech. Rep. WO-26. Washington, DC: U.S. Department of Agriculture Forest Service: 112–136.

Dibble, A.C.; White, R.H.; Lebow, P.K. 2007. Combustion characteristics of North-Eastern USA vegetation tested in the cone calorimeter: invasive versus non-invasive plants. International Journal of Wildland Fire. 16: 426–443.

Dickinson, K.J.M.; Kirkpatrick, J.B. 1985. The flammability and energy content of some important plant species and fuel components in the forests of southeastern Tasmania. Journal of Biogeography. 12: 121–134.

Dimitrakopoulos, A.P.; Panov, P.I. 2001. Pyric properties of some dominant Mediterranean vegetation species. International Journal of Wildland Fire. 10: 23–27.

Golley, F.B. 1961. Energy values of ecological materials. Ecology. 42(3): 581–584.

Hough, W.A. 1969. Caloric value of some forest fuels of the Southern United Sates. Res. Note SE-120. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experimental Station. 6 p.

Johnson, E.A.; Miyanishi, K. 1995. The need for consideration of fire behavior and effects in prescribed burning. Restoration Ecology. 3(4): 271–278.

Lippincott, C.L. 2000. Effects of *Imperata cylindrica* (L.) Beauv. (Cogongrass) invasion on fire regime in Florida sandhill (USA). Natural Areas Journal. 20: 140–149.

Mutch, R.W. 1970. Wildland fires and ecosystems—a hypothesis. Ecology. 51(6): 1046–1051.

Platt, W.J. 1999. Southeastern pine savannas. In: Anderson, R.C.; Fralish, J.S.; Baskin, J.M., eds. Savannas, barrens, and rock outcrop plant communities of North America. Cambridge, UK: Cambridge University Press: 23–51.

Platt, W.J.; Carr, S.M.; Reilly, M.; Fahr, J. 2006. Pine savanna overstory influences on ground-cover biodiversity. Applied Vegetation Science. 9: 37–50.

Platt, W.J.; Glitzenstein, J.S.; Streng, D.R. 1991. Evaluating pyrogenicity and its effects on vegetation in longleaf pine savannas. In: Proceedings, 15th annual Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station. 17: 143–161.

Pyne, S.J. 1984. Introduction to wildland fire: fire management in the United States. New York: John Wiley. 455 p.

Rebertus, A.J.; Williamson, G.B.; Moser, E.B. 1989. Longleaf pine pyrogenicity and turkey oak mortality in Florida xeric sandhills. Ecology. 70(1): 60–70.

Thaxton, J.M.; Platt, W.J. 2006. Small-scale fuel variation alters fire intensity and shrub abundance in a pine savanna. Ecology. 87(5): 1331–1337.

Williamson, G.B.; Black, E.M. 1981. High temperature of forest fires under pines as a selective advantage over oaks. Nature. 293: 643–644.

SHORT-TERM EFFECTS OF FUEL REDUCTION TREATMENTS ON SOIL MYCORRHIZAL INOCULUM POTENTIAL IN BEETLE-KILLED STANDS

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Abstract—Heavy fuel loads were created by southern pine beetle (*Dendroctonus frontalis* Ehrh.) outbreak throughout the southeastern Piedmont during the early 2000s. Prescribed burning and mechanical mulching (mastication) were used to reduce fuel loading, but many ecological impacts are unknown. Successful forest regeneration depends on ectomycorrhizal (ECM) or vesicular-arbuscular mycorrhizal (VAM) fungi which form important symbiotic relationships with most forest plants. Fuel reduction treatments may impact mycorrhizal propagule abundance and/or vigor through propagule consumption, changes in soil chemistry, and/or effects on host vegetation. The objective of this study was to compare soil VAM and ECM inoculum potential after prescribed burning and mulching treatments to no treatment (control) using greenhouse bioassays. Neither VAM nor ECM inoculum potential were significantly different among treatments, but were highly variable within treated stands.

INTRODUCTION

Background

There was epidemic southern pine beetle (*Dendroctonus frontalis* Ehrh.) activity in the Southeastern United States during the early 2000s. Beetle-killed pine trees fall in 1 to 2 years, and stands are quickly colonized by herbaceous and early successional woody vegetation. Resulting conditions create a fuel hazard and greatly impede forest management activities.

Natural resource managers in the southeastern Piedmont region requested information about consequences to various ecosystem properties associated with using prescribed fire and mechanical mulching as site preparation treatments.

Mycorrhizas

Mycorrhizas are symbiotic relationships between soil fungi and plant roots and confer drought and disease tolerance to the plant by increasing their absorptive root surface area (Sylvia and others 2005). Most forest plants are dependent on mycorrhizal colonization for their establishment and productivity (Janos 1980). Soil fungi that form associations with the majority of plants in the southeastern Piedmont are glomalean and basidiomycetous fungi and form vesicular-arbuscular (VAM) and ectomycorrhizas (ECM), respectively. Major VAM tree genera in this region are *Acer* L., *Fraxinus* L., *Liriodendron* L., *Prunus* L., and *Liquidambar* L. In addition, many shrub and most herbaceous plants in the region are VAM. Major ECM tree genera are *Carya* Nutt., *Fagus* L., *Pinus* L., and *Quercus* L.

Sources of mycorrhizal propagules in forests are old roots, mycelia, sclerotia, and spores (Brundrett and Kendrick 1988). Therefore, existing vegetation likely plays an important role as refugia for mycorrhizal fungi that colonize forest regeneration. Spores are thought to play a minor role in initiating mycorrhizal colonization in forested ecosystems (Janos 1980) but interestingly were the focus of several studies that concluded that forest disturbance changed the mycorrhizal dynamics in soil.

Changes in mycorrhizal dynamics may be caused by disturbance-related changes in the abundance and/or activity of propagules (Klopatek and others 1988) which has been termed “soil inoculum potential” (Smith and Read 2000). Such changes may arise from direct damage to propagules (Klopatek and others 1988), damage to host vegetation, i.e., indirect damage to mycorrhizal propagules (Buchholz and Gallagher 1982), or changes in soil chemistry (Herr and others 1994).

Total soil inoculum potential is the cumulative potential for all sources of mycorrhizal propagules to initiate colonization with the roots of host plants. It is unclear if prescribed fire and mechanical fuel reduction result in changes in total soil VAM and ECM inoculum potential. Therefore, the objective of this study was to compare short-term soil VAM and ECM inoculum potential among treatments.

METHODS

Study Area

The study was conducted in 12 beetle-killed pine stands each approximately 1 ha in size in the Clemson University Experimental Forest. The stands were artificially planted or naturally regenerated and approximately 18 to 33 years in age when killed. Mean diameter of *Pinus* spp. stems (live or dead) in the year 0 vegetation community was 21.9 cm. Metal stakes were placed on a 25- by 25-m spacing to create a grid

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system throughout each stand and permanent references for conducting fuel, vegetation, and soil sampling.

Fuel Reduction Treatments

The 12 stands were randomly assigned to 1 of 3 fuel reduction treatments in an unbalanced design to create 3 replications each of control and mulching and 6 replications of prescribed burning. The mulching treatment was accomplished using a tracked machine equipped with a hydraulic-driven masticating head. The mulching treatment commenced in late May 2005 and was completed in late June 2005.

The original study plan involved burning in two different seasons to achieve two different levels of fire intensity. However, prescribed burning was delayed in 2005 due to weather. Therefore, all burning was conducted in a 3-day period between March 30 and May 3, 2006, using manual strip-head firing.

Mycorrhizal Bioassays

Plots used to sample vegetation in another component of the current study were used to collect soil samples for mycorrhizal bioassays. Two 10- by 50-m plots were established in each beetle-killed stand and contained five 10- by 10-m subplots. Soil sampling was performed between May 22 and June 8, 2006, for VAM bioassays and between July 10 and July 20, 2006, for ECM bioassays. For each sampling period, 4 intact 211-mL soil cores were obtained from the centers of each subplot providing a total of 20 observational units per vegetation sampling plot. A total of 40 soil cores were obtained from each vegetation sampling plot after collection for VAM and ECM bioassays.

Soil samples were returned to the Clemson University Greenhouse Complex at the end of each sampling day. Soil cores for ECM bioassays were immediately placed in a HEPA-filtered chamber constructed in a greenhouse and previously shown to reduce contamination by airborne ECM fungi (Stottlemyer and others 2008). Soil cores collected for VAM bioassays were planted with corn (*Zea mays* L. 'Viking') seed and were allowed to grow for 4 weeks. Soil cores collected for ECM bioassays were planted with loblolly pine (*P. taeda* L.) seed and were allowed to grow for 6 weeks. All seedlings grew under natural light for the duration of the growing periods and no fertilizers were applied.

At the end of their respective growing periods, corn and pine seedlings were destructively harvested and rinsed free of soil. A subsample of 50 1-cm corn root segments (<1 mm in diameter) were mounted on glass slides after clearing with 10 percent potassium hydroxide (KOH), staining with trypan blue, and destained in 50-percent glycerol. Slides were assessed with a compound microscope equipped with a crosshair eyepiece under 110× magnification. The presence/absence of VAM hyphae was noted at each intersection of the crosshair and a root segment. VAM colonization values were calculated using the equation: VAM colonization = number intersections at which hyphae were present ÷ 50. Root systems of more than 450 corn seedlings were assessed for VAM colonization and root and shoot growth after accounting for seedling mortality and nongerminants.

Pine root systems are heterorhizic with distinct short roots and long (lateral) roots from which short roots subtend (Brundrett and others 1996b). Three lateral roots ≥6 cm in length were randomly selected from each seedling. Each short root was tallied and classified as mycorrhizal or nonmycorrhizal using a dissecting microscope. Nonmycorrhizal short roots were slender and elongated, possessed root hairs and root caps, and lacked fungal mantles. Mycorrhizal short roots were bifurcate or monopodial, possessed fungal mantles, and lacked root hairs and root caps. Colonization values were calculated using the equation: ECM colonization = number of mycorrhizal short roots ÷ total number of short roots. Root systems of more than 380 pine seedlings were assessed for ECM colonization and root and shoot growth after accounting for seedling mortality and nongerminants.

Statistical Analysis

Average percentages of mycorrhizal colonization of corn and pine seedlings were calculated for each beetle-killed stand and compared among fuel reduction treatments using analysis of variance (PROC GLM; SAS Institute Inc., Cary, NC).

RESULTS AND DISCUSSION

Mycorrhizal Colonization of Bioassay Seedlings

Impacts of prescribed fire and mechanical treatments on mycorrhizal dynamics are not fully understood. Past studies that used "most probable number" bioassay methods showed that prescribed burning decreased soil VAM (Rashid and others 1997) and ECM (Torres and Honrubia 1997) inoculum potential in different forest ecosystems. However, this methodology involves soil collection, dilution with sand, and mixing prior to growing bait plants. Mixing soil likely disrupts old root systems and mycelia networks (Horton and others 1998) which are the primary mode of colonization of forest regeneration (Brundrett and others 1996a).

In the current study, there were no significant differences in percentage VAM and ECM colonization of bioassay seedlings among fuel reduction treatments in beetle-killed stands that were subjected to different fuel reduction treatments (fig. 1). Fungal mycelia may have escaped injury or resistant propagules including spores and sclerotia may have initiated mycorrhizal colonization with corn or pine seedlings. Therefore, the possibility for multiple propagules to initiate mycorrhizal colonization after forest disturbance highlights the importance of bioassay methods that assess total soil inoculum potential.

There was a high degree of variability in soil mycorrhizal inoculum potential within beetle-killed stands treated with different fuel reduction treatments (table 1). Variability in soil inoculum potential may have been caused by variation in the intensity of the treatments. For example, a wide range of fire intensities and residence times are likely in beetle-killed stands due to spatial variability in fuel loading. In addition, existing hyphal networks and old roots in soil are the primary

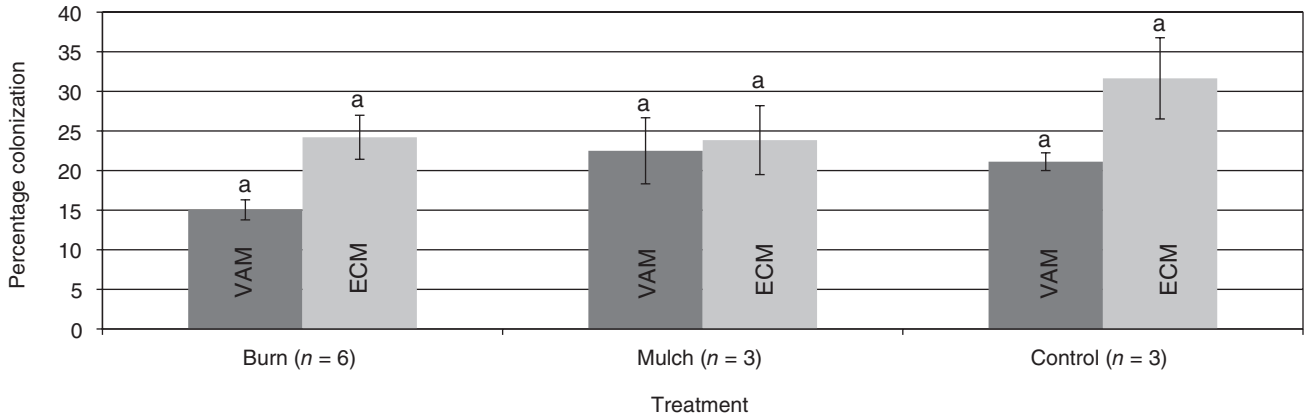

Figure 1—Comparisons of mean percentage vesicular-arbuscular mycorrhizal (VAM) and ectomycorrhizal (ECM) colonization of corn (*Zea mays* L.) and loblolly pine (*Pinus taeda* L.) seedlings. Seedlings were grown in intact soil cores collected after beetle-killed stands were subjected to different fuel reduction treatments. Bars represent standard errors of the means. Similar letters indicate that VAM or ECM inoculum potential was not significantly different among treatments.

Table 1—Ranges in percentage mycorrhizal colonization of corn (*Zea mays* L.) and loblolly pine (*Pinus taeda* L.) seedlings used for greenhouse bioassays of total soil mycorrhizal inoculum potential. Seedlings were grown in intact soil cores collected following different fuel reduction treatments in beetle-killed pine stands.

	Ranges in percentage mycorrhizal colonization		
Mycorrhiza	Control (no treatment) (*n* = 3)	Prescribed burn (*n* = 6)	Mulch (*n* = 3)
Vesicular-arbuscular mycorrhizal	16.67–24.10	8.47–20.50	11.50–37.85
Ectomycorrhizal	14.09–45.55	10.27–37.88	10.99–41.42

source of mycorrhizal inoculum for new plant germinants. Therefore, the composition and structure of pretreatment vegetation has the potential to influence posttreatment soil inoculum potential.

CONCLUSIONS

Prescribed fire and mechanical mulching have been proposed to reduce high fuel loading in beetle-killed pine stands in the Southeastern United States. Forest managers in the region are interested in whether these treatments offer viable options for reducing fuels without jeopardizing site productivity. Until recently, the effectiveness of the treatments at reducing fuels in extremely high fuel loading and their impacts on important ecosystem processes were largely unknown.

Mycorrhizal fungi have the potential to influence the trajectory of vegetation succession after a disturbance. We compared VAM and ECM inoculum potential among treatments using bioassays of intact soil cores in the greenhouse. Neither VAM nor ECM inoculum potential were significantly different among fuel reduction treatments but were highly variable within posttreatment beetle-killed stands. Understanding how pretreatment vegetation, fire behavior, and soil fertility influenced soil inoculum potential is the focus of current research.

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LITERATURE CITED

Brundrett, M.C.; Ashwath, N.; Jasper, D.A. 1996a. Mycorrhizas in the Kakadu region of tropical Australia. Plant and Soil. 184: 173–184.

Brundrett, M.C.; Bougher, N.; Dell, B. [and others]. 1996b. Working with mycorrhizas in forestry and agriculture. ACIAR Monograph 32. 374 p.

Brundrett, M.C.; Kendrick, B. 1988. The mycorrhizal status, root anatomy, and phenology of plants in a sugar maple forest. Canadian Journal of Botany. 66: 1153–1173.

Buchholz, K.; Gallagher, M. 1982. Initial ectomycorrhizal density response to wildfire in the New Jersey pine barren plains. Bulletin of the Torrey Botanical Club. 109: 396–400.

Herr, D.G.; Duchesne, L.C.; Tellier, R. [and others]. 1994. Effect of prescribed burning on the ectomycorrhizal infectivity of a forest soil. International Journal of Wildland Fire. 4: 95–102.

Horton, T.R.; Cazares, E.; Bruns, T.D. 1998. Ectomycorrhizal, vesicular arbuscular and dark septate fungal colonization of bishop pine (*Pinus muricata*) seedlings in the first 5 months of growth after wildfire. Myccorhiza. 8: 11–18.

Janos, D. 1980. Mycorrhizae influence tropical succession. Biotropica. 12: 56–64.

Klopatek, C.C.; Debano, L.F.; Klopatek, J.M. 1988. Effects of simulated fire on vesicular arbuscular mycorrhizae in pinyon-juniper woodland soil. Plant and Soil. 109: 245–249.

Rashid, A.; Ahmed, T.; Ayub, N.; Khan, A.G. 1997. Effect of forest fire on number, viability and post-fire re-establishment of arbuscular mycorrhizae. Myccorhiza. 7: 217–220.

Smith, S.E.; Read, D.J. 2000. Mycorrhizal symbiosis. 2nd ed. New York: Academic Press. 605 p.

Stottlemyer, A.D.; Wang, G.G.; Wells, C.E. [and others]. 2008. Reducing airborne ectomycorrhizal fungi and growing non-mycorrhizal loblolly pine (*Pinus taeda* L.) seedlings in a greenhouse. Mycorrhiza. 18: 269–275.

Sylvia, D.M.; Fuhrmann, J.J.; Hartel, P.G.; Zuberer, D.A. 2005. Principles and applications of soil microbiology. 2nd ed. Upper Saddle River, NJ: Prentice Hall. 638 p.

Torres, P.; Honrubia, M. 1997. Changes and effects of natural fire on ectomycorrhizal inoculum potential of soil in a *Pinus halepensis* forest. Forest Ecology and Management. 96: 189–196.

FUEL DYNAMICS ACROSS SOUTHERN APPALACHIAN LANDSCAPES

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Abstract—This study was conducted in Rabun County, GA, on the Warwoman Wildlife Management Area to measure the annual accumulations and decomposition of leaf litter, fine woody fuel, and total fuel loadings on undisturbed sites across different topographical positions in the Southern Appalachian Mountains. There were five "treatments" used in this study, representing five topographic positions: ridge tops, middle, and lower slopes on northeast (325 to 125 degrees) and southwest (145 to 305 degrees) aspects. Ten plots (replicates) were established at each topographic position for a total of 50 plots. Results suggested that there were few differences in accumulation and decomposition of leaf litter, 1-, 10-, and 100-hour fuels among different topographical positions. The only exception was coarse woody debris, which had significantly more on northeast facing slopes (26.6 tons/ha) compared to all other slope positions (10.8 tons/ha). Ericaceous shrubs were present on 74 percent of plots and could have influenced the results.

INTRODUCTION

For approximately 70 years, fire has been suppressed in Southern Appalachian ecosystems, and the result of fire exclusion has been a buildup of fuels and an expansion of ericaceous shrubs such as mountain laurel (*Kalmia latifolia*) and rhododendron (*Rhododendron maximum*) (Vose 2000). These increases, coupled with an increasing human population, have increased the probability of intense, severe fires. Mountain laurel and rhododendron can burn intensely resulting in mixed severity or even stand replacement fires (Stanturf and others 2002). Harrod and others (2000) suggest that reductions in fire frequency through active fire suppression and changing patterns in land uses have resulted in a decrease in fire frequency, thus increasing stand densities. The result has been a less diverse and productive understory. In addition, an increase in canopy density and decreasing grass cover have combined to shift the disturbance regime from frequent low-intensity surface fires to infrequent but catastrophic crown fires (Harrod and others 2000).

The Southern Appalachian Mountains have diverse topography, which produces a complex mosaic of site types. Each site type is affected by soil and topography (slope, slope position, elevation, and aspect), which influence temperature, light, and moisture (Graham and McCarthy 2006, Waldrop and others 2007). These variables produce drastically different fuel conditions that change both temporally and spatially. The fuel dynamics of this area can be as complex as the mountains themselves. Rubino and McCarthy (2003) stated that stand composition varies drastically with topographic gradient resulting in different edaphic climaxes that can be found within close proximity of one another. This mosaic of vegetative communities can change fuel characteristics over very short distances (<100 meters) with changing microclimate (Graham and McCarthy 2006).

There is a need to understand how hardwood fuels are distributed across Southern Appalachian landscapes to give fire planners the knowledge to apply appropriate silvicultural treatments to obtain desired management objectives. There have been several studies examining fuel loads in the Southern Appalachian Mountains and central hardwood region including fuel loading in the central hardwood region (Kolaks and others 2003), evaluation of coarse woody debris (CWD) and forest vegetation (Rubino and McCarthy 2003), and forest floor fuel dynamics in mixed-oak forest (Graham and McCarthy 2006). These studies yielded useful information about fuel loadings in these ecosystems enabling fire planners to use the data directly for fire planning or in fire behavior modeling, but none of these studies analyzed how accumulations and decomposition of fuels differed across differing topographical gradients (Waldrop and others 2007). Graham and McCarthy (2006) stated that varying microclimates resulting from highly dissected landscapes produce very different fuel conditions dependent on slope position, percent slope, and slope aspect, all of which affect moisture. Moisture in turn influences both the productivity (inputs) and decay (loss) rates of fuel on these sites.

In a recent study, fuels on disturbed and undisturbed sites in the Southern Appalachian Mountains, Waldrop and others (2007) measured fuel loadings on 1,008 plots of which 705 were undisturbed. Total basal area for undisturbed sites averaged 29.1 m^2/ha and was higher on lower slope positions and decreased towards the ridge tops. Litter was heavier on ridge tops and decreased downhill in both the northeast and southwest slopes, suggesting that decomposition exceeded leaf litter inputs on the more mesic sites. This study found there was 12 percent less litter on northeast lower slopes (3.8 tons/ha) than on ridge tops (4.2 tons/ha). There were significantly less 1-hour fuels on ridge top positions (0.6 tons/ha) as compared to the other slope positions (0.7 tons/ha).

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There were also no significant differences in 1,000-hour fuel loadings on undisturbed sites, though there was on average more 1,000-hour fuels on northeast lower slopes (44 tons/ha) as compared to other slope positions (35 tons/ha). There was more mountain laurel on southwest slopes and rhododendron was most common on northeast lower slopes.

Previous studies suggest that differing decomposition rates balance the loading of downed woody fuels across topographic gradients (Abbott and Crossley 1982, Graham and McCarthy 2006, Kolaks and others 2003, Waldrop and others 2007). However, inputs and decay rates of leaf litter and fine and coarse woody fuels have never been measured across differing topographic gradients. The overall objective of this study was to measure inputs and decomposition rates of leaf litter and fine and coarse woody fuels across a topographic gradient in the Southern Appalachian Mountains.

METHODS

Study Site

The study measured a portion of the study sites used by Waldrop and others (2007) in the Warwoman Wildlife Management Area (WMA), occupying approximately 6397 ha, within the Chattooga River Ranger District of the Chattahoochee National Forest, Rabun County, GA. The WMA is characterized by short, steep slopes with elevations ranging from 244 to just over 1036 m. The average temperature and precipitation during the study period for the area was 12.5 °C and received on average 172.8 cm of precipitation (National Oceanic and Atmospheric Administration 2008). The long-term average precipitation (100-year average, 1907 to 2007) was 127.3 cm annually (National Oceanic and Atmospheric Administration 2008).

Waldrop and others (2007) reported red oak species were the most common species in the overstory on the WMA followed by yellow pines, and then all other understory species. Red oak species account for 25 percent of the total basal area while white oak species, including chestnut oak, occupy only 6 percent of the total basal area. This was surprising because chestnut oak is considered a dominant species present on the drier upper slope positions in the Southern Appalachian Mountains.

Experimental Design

This study used a completely randomized design, with a subset of plots established by Waldrop and others (2007). There were 50 plots chosen at random, 10 replicates from each "treatment." This subset of plots was used to measure the input/accumulation and decomposition (loss) of fuels across differing landscape gradients in the current study. Waldrop and others (2007) defined topographic position as a combination of slope position and aspect, and assumed that tree productivity and, thus, fuel loadings would be greater on more productive sites. The five "treatments" consisted of topographic positions including ridge tops, middle slopes, and lower slopes on northeast (325 to 125 degrees) and southwest (145 to 305 degrees) aspects. Ten plots (replicates) were established at each topographic position for a total of 50 plots.

Litter Trap Design and Sampling Procedures

In most litter and woody detritus decay studies, a litter bag of some type is used. However, Binkley (2002) used a method previously described as a litter "sandwich." In the litter sandwich method one piece of screen (with 2- or 3-mm openings) was first fastened to a stable frame (small wooden pieces 5 by 5 cm work well), then, after the major litter fall period, another piece of screen was placed on top of the freshly fallen litter. This pattern of fresh litter fall and new screen application can be continued for as long as the study is designed to last. This design mimics the natural dynamics of the forest floor; in addition it alleviates the problem of excluding soil micro- and macrofauna. Samples can easily be cut from the layers of screen at designated intervals without disturbing other material. Five 1-m^2 litter-trap sandwiches were placed at each of the 50 study plots, for a total of 250 litter traps, prior to the major leaf fall in September 2005.

January 2006 was designated as the end of leaf fall and sampling began in February 2006. The end of year 2 sampling started in January of 2007. Sampling took place every 3 months and consisted of a 10-cm^2 subsample cut from within each litter trap, for a total of 5 samples per plot and 250 samples per sample period. During sampling the litter traps were emptied and the material was sorted into different fuel categories, litter (including acorns and bark); pine cones; 1- (zero to 0.64 cm), 10- (0.64 to 2.5 cm), 100- (2.5 to 7.6 cm), and 1,000-hour fuels (>7.6 cm). All sorting was conducted in the field. The separated materials were weighed and divided equally among the five traps (the total mass of material was divided by five) at each point. The material was divided into the five litter traps because, in most cases, there was too much litter to place into a single litter trap. All 10- and 100-hour fuels were placed into litter trap 1, because the quantities were not sufficient to place them into all five litter traps and be able to collect subsamples in subsequent sampling periods. After the material was redistributed, a screen was placed on top of the material and loosely (care was taken not to compress the material) stapled to the wooden frame forming the sandwich. After screens were stapled on top of the 5 litter traps, a 10-cm^2 subsample was cut from a random location within each trap; the location was the same for all 250 litter traps.

A 4- by 20-m (80 m^2) grid was established in a randomly assigned azimuth to sample CWD (>7.62 cm in diameter). Each piece of CWD was painted so it would not be remeasured in subsequent sampling periods. Each of the CWD grid plots was surveyed during the resampling periods; approximately every 3 months for a total of nine sampling periods.

The process of sorting and weighing fuels was repeated after the litter fall of 2006. The final sampling period started in December of 2007. At that time, there were only 212 traps intact and viable to sample. Others were destroyed by wildlife, mostly just after mast fall.

Data Analysis

Data from this study were analyzed in three major components: (1) accumulation/production (input of leaf litter and fine woody fuels), (2) loss (decay) of leaf litter and fine woody fuels, and (3) total fuel loadings across the topographical gradients. To obtain an accurate assessment of the five slope and aspect positions and differences associated with the material collected, a one-way analysis of variance was conducted (SAS Institute 2002) with differences considered significant at $\alpha = 0.05$, and a linear contrast was performed to compare the mean of northeast slopes (the slope positions that should have been the more mesic) to the mean of all other slope positions. Mean separation was determined by Fisher's protected least significant differences. A matched pair t-test was conducted on the means for leaf litter, 1- and 10-hour fuels collected in years 2005 and 2006, to determine if there were differences in the quantity of material collected between the 2 years.

RESULTS

Accumulation and Production

There were no significant differences in accumulations of leaf litter, 1-, 10-, and 100-hour fuels detected among slope and aspect position for either of the 2 years (2005 or 2006) (fig. 1). There were significant differences in the quantity of litter, 1- and 10-hour fuels produced between the 2 years. More litter was produced in 2006 than 2005 ($P = 0.0172$), but there were more 1- and 10-hour fuels produced in 2005 than 2006 ($P = 0.0172$ and 0.0434). In addition, 2005 produced more 100-hour fuels than 2006; no 100-hour fuels were collected in 2006.

One possible reason that the 2006 collection had more leaf litter than 2005 was the litter traps were not placed in the field until August of 2005. The leaves that had fallen earlier in 2005 were not recovered and placed into the litter traps. There were more 1- and 10-hour fuels collected in 2005 than in 2006; this could have been caused by major wind events or possible damage from a major ice storm in December of 2005. The high variance within each treatment for both 2005 and 2006 was caused by high variability from plot to plot. This variability and lack of significant differences can possibly be attributed to the presence of ericaceous shrubs. In combination, mountain laurel and rhododendron were present on 37 of 50 (74 percent) plots in this study. There could be strong influences on the input of fuel material with either or both of the ericaceous shrub species because both are composed of heavy volatile chemicals that weigh proportionally more than leaves produced by other species (Clinton 2002). These differences in specific leaf weights among species were ignored in previous studies when determining litter fuel loading. In addition there can be a 20-percent reduction in available water to plants and decomposers under rhododendron thickets (Clinton 2002).

Analysis of CWD data showed significant differences in CWD loading between the northeast facing slopes and all other slope positions (fig. 2). There were no significant differences between slope and aspect positions for CWD. Subsequent linear contrast showed there was a significant difference between the mean of northeast facing slopes as compared to the mean of all other slope positions ($P = 0.0042$).

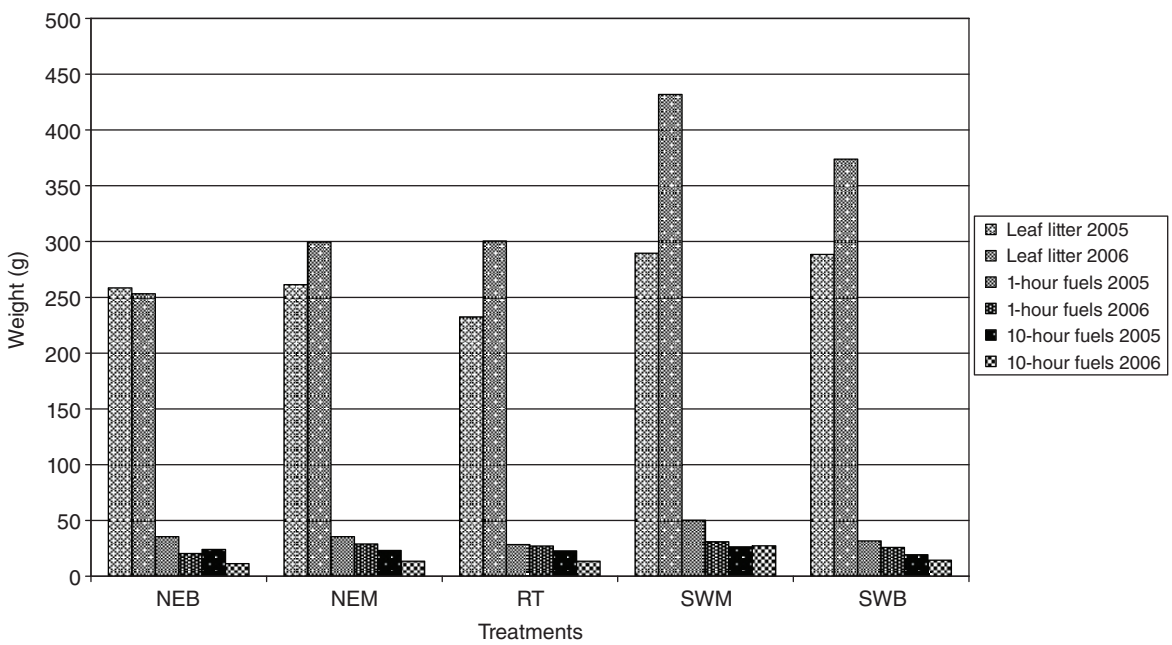

Figure 1—Mean accumulations of leaf litter and fine woody fuels for 2005 and 2006. (NEB = northeast bottoms, NEM = northeast midslopes, RT = ridgetops, SWM = southwest midslopes, and SWB = southwest bottoms)

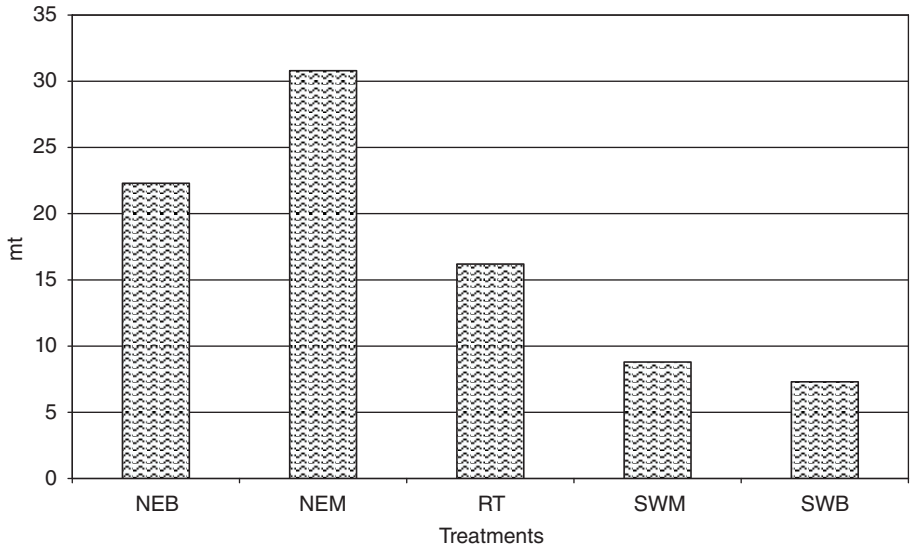

Figure 2—Initial 2005 coarse woody debris survey. (NEB = northeast bottoms, NEM = northeast midslopes, RT = ridgetops, SWM = southwest midslopes, and SWB = southwest bottoms)

It was not expected to find significant differences for total fuel loadings across the topographical gradient. There were no differences detected in the quantity of material collected in the litter traps among slope and aspect positions, and therefore, there should not have been detectable differences in total fuel loadings among the same slope and aspect positions. There was, though not significant, more leaf material collected on southwest slope positions (3.5 tons/ha) as compared to northeast slope positions (2.4 tons/ha), and the 1-, 10-, and 100-hour fuels are virtually identical across all treatments.

Decomposition

The 2005 leaf litter, 1-, 10-, and 100-hour fuels (material that had been in the field for 2 years) were not significantly different among slope positions or aspect. Similarly, the 2006 leaf litter, 1- and 10-hour fuels (material that had been in the field for 1 year) had no significant differences among slope positions or aspect (fig. 3). It was expected that on the more protected sites (northeast slopes and lower southwest slopes) there would be smaller remaining masses, indicating more loss through decay. These sites hold more moisture and therefore should have faster turnover times as compared to drier site types. However, that pattern was not observed. Ericaceous shrubs could have had an effect on decay. These tightly closed canopies act to keep the microsite moist and cool, and it is well accepted that both moisture and temperature are two of the main driving forces in the decay process (Clinton 2002). However, Nilsen and others (2001) reported that although it is moist under a rhododendron canopy, there is 20 percent less water available to plants and decomposers due to evapotranspiration by rhododendron. In this study, there was a greater abundance of rhododendron (40 percent) and mountain laurel (54 percent) than the findings reported by Waldrop and others (2007), at 25 and 42 percent, respectively. This could have biased the study findings to lower decay rates than are actually found on the same slope and aspect positions without ericaceous shrubs. The dense closed canopy and higher evapotranspiration rates associated with rhododendron, combined with the high acid and lignin content found in rhododendron leaves, create conditions that are suboptimal for decomposers, thereby slowing the decomposition process. Another possible influence on decay in this study was infiltration of fine roots into the layers of litter traps. Mass of fine roots present at the time the material was removed from within the litter traps could not be estimated; however, there were many samples from the more mesic sites that had a substantial quantity of fine roots, based on visual observations.

CONCLUSION

Significant differences in accumulation and decomposition were not detected for leaf litter, 1-, 10-, and 100-hour fuels among the five different slope position and aspect combinations for either year sampled. The results support findings from previous studies. The overall mean for leaf litter accumulations was 299.4 g/m^2 which fall within the range of 291 to 785 g/m^2 reported in other studies. Mean accumulation of woody material was 104.3 g/m^2 just below means of 106.9 and 107.6 g/m^2 reported in previous studies.

Northeast facing lower slopes had significantly greater 1,000-hour fuels than other topographic positions (26.6 and 10.8 tons/ha, respectively). This supports findings of Waldrop and others (2007), who reported finding 44 tons/ha on northeast facing slopes and 35 tons/ha for all other slope positions, and Kolaks and others (2003), who reported 8.4 tons/ha on protected slopes and 3.9 tons/ha on unprotected slopes.

Further study is needed, with study plots biased against ericaceous shrub species, to validate input and decay

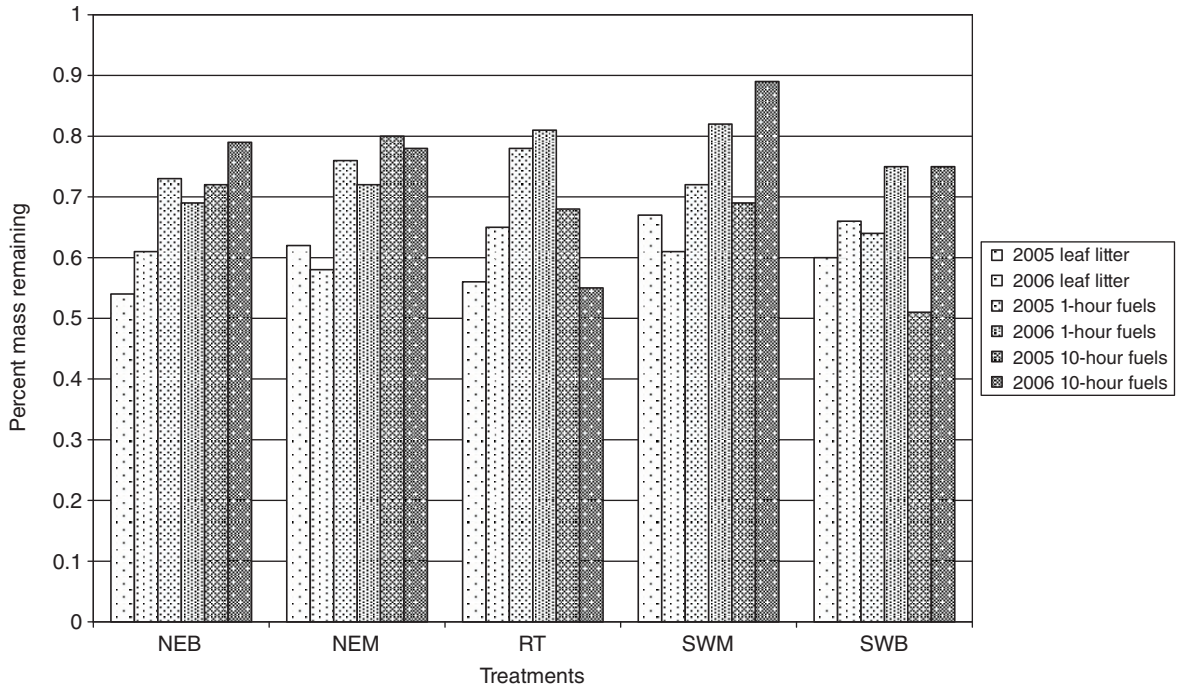

Figure 3—Percent mass remaining for leaf litter, 1- and 10-hour fuels 2005 and 2006. (NEB = northeast bottoms, NEM = northeast midslopes, RT = ridgetops, SWM = southwest midslopes, and SWB = southwest bottoms)

differences over different landscape positions in the Southern Appalachian Mountains. By biasing against these species, a more diversified species composition could be captured in the litter traps and the influence of ericaceous shrub species could be eliminated or greatly reduced.

ACKNOWLEDGMENTS

I would like to thank every member of the U.S. Forest Service work unit at Clemson. Each and every one helped me immeasurably with the daunting task that I undertook. I would especially like to thank Mr. Mitch Smith, who spent just as many hours as I in the field. This is Contribution Number 71 of the National Fire and Fire Surrogate Project, funded by the U.S. Joint Fire Science Program and by the U.S. Forest Service through the National Fire Plan.

LITERATURE CITED

Abbott, D.T.; Crossley, D.A., Jr. 1982. Woody litter decomposition following clear-cutting. Ecology. 63: 35–42.

Binkley, D. 2002. Ten-year decomposition in a loblolly pine forest. Canadian Journal of Forest Research. 32: 2231–2235.

Clinton, B.D. 2002. *Rhododendron maximum* L. In: Francis, J.K., ed. Wildland shrubs of the United States and its territories. Gen. Tech. Rep. IIFT-WB-1. www.fs.fed.us/global/iitf/wildland_shrubs.htm. [Date accessed: July 21, 2011].

Graham, J.B.; McCarthy, B.C. 2006. Forest floor fuel dynamics in mixed-oak forest of south-eastern Ohio. International Journal of Wildland Fire. 15: 479–488.

Harrod, J.C.; Harmon, M.E.; White, P.S. 2000. Post-fire succession and 20th century reduction in fire frequency on xeric Southern Appalachian sites. Journal of Vegetation Science. 11: 465–472.

Kolaks, J.E.; Cutter, B.E.; Loewenstine, E.F. [and others]. 2003. Fuel loading in the central hardwoods (CD–ROM). Pap. 1A-3. In: Proceedings on the 2nd international wildland fire ecology and fire management congress. Orlando, FL: American Meteorology Society. 6 p.

Nilsen, E.T.; Clinton, B.D.; Lei, T.T. [and others]. 2001. Does *rhododendron maximum* L. (ericaceae) reduce the availability of resources above and belowground for canopy tree seedlings? The American Midland Naturalist. 145: 325–343.

National Oceanic and Atmospheric Administration. 2008. Climatological data annual summary, Georgia 2004, 2005, 2006 and 2007. Vol. 108, 109, 110, and 111; Number 13, ISSN 0145-0492.

Rubino, D.L.; McCarthy, B.C. 2003. Evaluation of coarse woody debris and forest vegetation across topographic gradients in a southern Ohio forest. Forest Ecology and Management. 183: 221–238.

SAS Institute Inc. 2002. SAS user's guide: statistics. Version 9.00. Cary, NC: SAS Institute Inc.

Stanturf, J.A.; Wade, D.D.; Waldrop, T.A. [and others]. 2002. Fire in southern forest landscapes. In: Wear, D.M.; Greis, J., eds. Southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 607–630.

Vose, J.M. 2000. Ecosystem perspectives on using prescribed fire to achieve desired ecosystem conditions. In: Moser, W.K.; Moser, C.F., eds. Fire and forest ecology: innovative silviculture and vegetation management. Proceedings of the 21st Tall Timbers fire ecology conference: 12–17.

Waldrop, T.A.; Brudnak, L.; Rideout-Hanzak, S. 2007. Fuels on disturbed and undisturbed sites in the Southern Appalachian Mountains, USA. Canadian Journal of Forest Research. 37: 1134–1141.

FUEL-REDUCTION TREATMENTS FOR RESTORATION IN EASTERN HARDWOODS: IMPACTS ON MULTIPLE ECOSYSTEM COMPONENTS

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Abstract—The Southern Appalachian Mountains and Ohio Hills sites are unique within the National Fire and Fire Surrogate Study because they are in hardwood-dominated forests. The efficacy of four fuel-reduction treatments was evaluated to restore these unmanaged hardwood forests to the structure and function of open woodland habitats. Treatments included control, prescribed burning, mechanical, and a combination of burning and mechanical treatments. Overstory basal area and density of saplings and shrubs were used as measures of ecosystem structure while soil carbon and breeding bird species richness were indicators of ecosystem function. The combination of burning and mechanical treatments provided the most rapid progress toward restoration but not all goals were met. At both study sites, overstory and understory vegetation remained too dense, soils were largely unaffected, and bird species richness showed only ephemeral increases. Repeated treatments are needed to replicate historical structure and function.

INTRODUCTION

Several studies document first-order effects after fuel-reduction treatments. However, none has attempted to establish the interactions between fuel reduction and multiple ecological processes. The National Fire and Fire Surrogate (FFS) Study was established to compare ecological and economic impacts of prescribed fire and mechanical fuel-reduction treatments (Youngblood and others 2005). Thirteen independent study sites across the United States (eight in the West and five in the East) use identical treatment (prescribed fire and mechanical fuel-reduction treatments) and measurement protocols. All western sites are dominated by ponderosa pine (*Pinus ponderosa*). Eastern sites include hardwood-dominated sites in the Ohio Hills sites and Southern Appalachian Mountains of North Carolina, a pine-hardwood site in the Piedmont of South Carolina, a site dominated by longleaf pine (*P. palustris*) in Alabama, and a site dominated by slash pine (*P. elliottii*) in Florida.

The two hardwood-dominated sites of the FFS (Southern Appalachian Mountains and Ohio Hills) are substantially different from other FFS sites because of their history, plant composition, animal composition, and soils. At both sites, the primary management objective was to reduce severity of potential wildfires by reducing live and dead fuels. Secondary objectives were to increase oak regeneration by reducing competition from red maple (*Acer rubrum*) and yellow-poplar (*Liriodendron tulipifera*), to improve wildlife habitat by creating early successional habitat, and to increase cover of grasses and forbs. It may be possible to obtain each of these goals by restoring these communities to the open woodland habitats once common in these regions [described in syntheses by Stanturf and others (2002) and Van Lear and Waldrop (1989)]. Fire and mechanical treatments used at both sites were designed to restore stand structure to an open woodland condition. A number of papers have described how the fuel-reduction treatments have impacted individual components of these ecosystems such as insects (Campbell and others 2007, Greenberg and others 2010), soils (Coates and others 2008), herpetofauna (Kilpatrick and others 2010), vegetation (Schwilk and others 2009), and fuels (Waldrop and others 2010). This paper examines several types of variables at both sites as indicators of restoration success and to evaluate common patterns.

METHODS

Study Sites

Both the Southern Appalachian Mountains and Ohio Hills study sites consist of three replicate blocks, with four fuel-reduction treatments applied to a randomly chosen unit within each block. The Southern Appalachian Mountains FFS site is located in the Green River Game Land in the Blue Ridge Physiographic Province, Polk County, NC. The climate of the region is warm continental, with mean annual precipitation of 1638 mm and mean annual temperature of 17.6 °C (Keenan 1998). The forests of the study area were 80 to 120 years old, and no indication of past agriculture or recent fire was present. The most abundant species in the canopy were northern red oak (*Quercus rubra*), chestnut oak (*Q. prinus*), white oak (*Q. alba*), black oak (*Q. velutina*), pignut hickory (*Carya glabra*), mockernut hickory (*C. tomentosa*), and shortleaf pine (*P. echinata*). A relatively dense evergreen shrub assemblage was present in the understory of a majority of the study site, with mountain laurel (*Kalmia latifolia*) and rhododendron (*Rhododendron maximum*) the most common species.

The Ohio Hills FFS site is located on the unglaciated Allegheny Plateau of southern Ohio. The climate of the region is cool, temperate with mean annual precipitation of 1024 mm and mean annual temperature of 11.3 °C (Sutherland and

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others 2003). The most abundant species in the current canopy were white oak, chestnut oak, hickories (*Carya* spp.), and black oak; however, the midstory and understory are now dominated by species that have only in the last few decades become common in this community, e.g., sugar maple (*A. saccharum*), red maple, and yellow-poplar (Yaussy and others 2003). Analysis of fire scars in stems of trees that were cut as part of the establishment of the FFS experiment indicated that fires were frequent (return intervals of 8 to 15 years) from 1875 to 1930.

Treatments and Experimental Design

Each of the three replicate blocks at each site is composed of four treatment units. In the Ohio Hills site, individual treatment units were 19 to 26 ha whereas in the Southern Appalachian Mountains site they were approximately 14 ha in size. A 50- by 50-m grid was established in each treatment unit, and 10 sample plots of 0.10 ha were established randomly within each treatment unit. Treatments consisted of prescribed fire, a mechanical treatment, the combination of prescribed fire and mechanical treatments, and an untreated control. At the Southern Appalachian Mountains site, the mechanical treatment was designed to create a vertical fuel break. Chainsaw crews removed all stems >1.8 m tall and <10.2 cm diameter at breast height (d.b.h.) as well as all mountain laurel and rhododendron stems, regardless of size. In the Ohio Hills, the mechanical treatment was thinning from below to a basal area comparable to that present prior to Euro-American settlement. This was a commercial thinning operation that reduced basal area from 27.0 to 20.9 m^2/ha. All detritus generated by treatments was left on site in both areas.

Mechanical treatments were accomplished between September 2000 and April 2001 in Ohio and between December 2001 and February 2002 at the Southern Appalachian Mountains site. The prescribed fires were applied during March to April 2001 in the Ohio Hills and March 2003 at the Southern Appalachian Mountains site. These dormant season fires consumed unconsolidated leaf litter and fine woody fuels while leaving the majority of the coarse woody fuels only charred. At the Southern Appalachian Mountains site, the fire prescription was also designed to kill ericaceous shrubs. Details of fire behavior are given by Tomcho (2004) for the Southern Appalachian Mountains site and by Iverson and others (2004) for the Ohio Hills.

Measurements and Analyses

All treatment units were sampled during the pretreatment year 2000 in the Ohio Hills and 2001 in the Southern Appalachian Mountains. Additional measurements were made in Ohio 1, 2, and 4 years after treatment. The Southern Appalachian Mountains site was measured 1, 3, and 5 years after treatment. Numerous variables were measured at grid points and sample plots for many components of the FFS study. Those used for this paper are a sample of variables used to evaluate restoration success, including overstory basal area, midstory saplings and shrubs, soil organic carbon, and breeding bird species richness. Measures of overstory and midstory characteristics will allow an evaluation of how well these treatments met the goal of creating the structure of an open woodland community. Responses of soil carbon and bird diversity provide measures of ecosystem function that will provide a more complete understanding of how well fuel-reduction treatments meet restoration goals. Specific measurements and analyses followed standard protocols developed for the FFS and were described in detail by Boerner and others (2008), Greenberg and others (2007), and Waldrop and others (2008). Individual tests were considered significant at the 0.05 level.

RESULTS

Overstory Basal Area

Basal area on the Southern Appalachian Mountains site varied from 23.8 to 27.3 m^2/ha prior to treatment, but the differences were not significant (fig. 1A). After one growing season, basal area had not changed significantly from pretreatment levels except in plots treated with the mechanical+burn combination. In those plots, basal area declined from 23.8 to 21.0 m^2/ha due to mortality after hot fires. Burn-only and mechanical-only plots continued

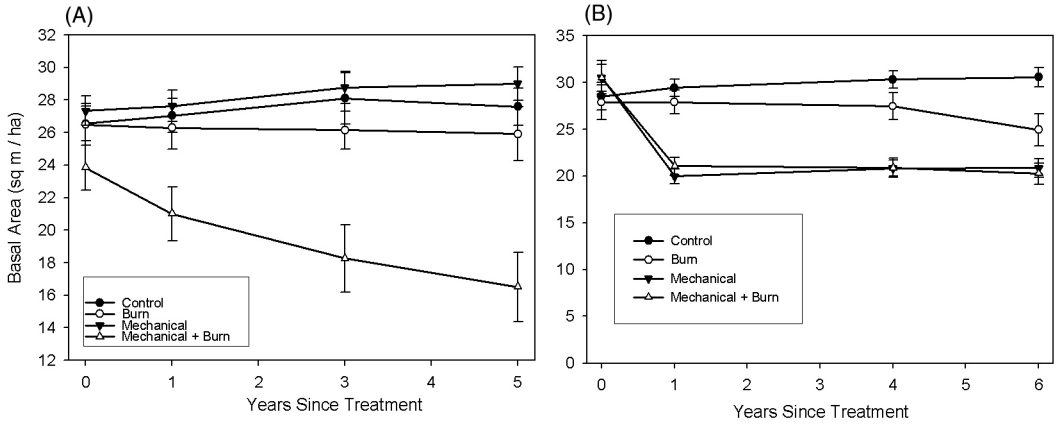

Figure 1—Change in basal area (m^2/ha) by treatment and year for the (A) Southern Appalachian Mountains and (B) Ohio Hills sites of the National Fire and Fire Surrogate Study (from Waldrop and others 2008).

to have no significant differences from control plots at measurements during years 3 and 5. However, basal area continued to decline significantly between sample years in the mechanical+burn plots, leaving only 16.5 m^2/ha of live trees after 5 years. The basal area of trees that died during the first year after treatment was significantly higher in burn-only and mechanical+burn plots than in those plots not treated with fire. Some mortality occurred in all treatment units between each sample period, but there was significantly more mortality in the mechanical+burn plots for 3 years following burning. At the end of 5 years, the only treatment that continued to have significant amounts of mortality was the mechanical+burn treatment. Species composition of the overstory was unaffected by treatment with mortality consistent among all species.

In Ohio Hills, the overstory responded differently than at the Southern Appalachian Mountains, primarily due to the difference in mechanical treatments (commercial thinning instead of understory cutting). Plots randomly selected for mechanical treatment had significantly higher basal area prior to treatment (30.5 m^2/ha) than did plots selected for burn-only (27.9 m^2/ha) or no treatment (28.5 m^2/ha) (fig. 1B). Commercial thinning operations did not achieve the target basal area of 14 m^2/ha, leaving 20.0 and 20.1 m^2/ha in the mechanical-only and mechanical+burn plots, respectively. Basal area in both treatment units remained about the same throughout the remainder of the 6-year measurement period. Basal area in burn-only treatment units was not significantly different than in untreated control plots the first year after burning. However, basal area increased in control plots over time as trees grew but decreased over time in burn-only plots as trees died. The reduction of live basal area in burn-only plots was significant between years 4 and 6. Some mortality occurred in all treatment units the first year after treatment but the amounts were small and not significantly different. Between 2 and 4 years after treatment, mortality increased in the areas treated with fire to levels significantly higher than in the controls or mechanical-only treatments. Mortality remained significantly higher in burn-only plots through year 6. Species composition of the overstory was unaffected by treatment with mortality consistent among all species.

Midstory Saplings and Shrubs

Numbers of sapling-sized trees (>4.5 feet tall and <4 inches d.b.h.) of all species groups tended to be significantly reduced 1 year after burning at both the Southern Appalachian Mountains and Ohio Hills (fig. 2). Sapling numbers increased over time at Ohio Hills, sometimes exceeding pretreatment densities. At the Southern Appalachian Mountains, however, there was a reduction in numbers at year 5 because this was the first-growing season after the second burn. Chainsaw felling at the Southern Appalachian Mountains reduced sapling density immediately after treatment, but there were no significant differences in sapling numbers for red maple and oaks by years 3 and 5, respectively. The mechanical treatment at Ohio Hills had little impact on sapling numbers the first year after treatment. Sapling numbers increased significantly by year 4 as large numbers of small trees grew into the sapling size class (fig. 3H). The mechanical+burn treatment at the Southern Appalachian Mountains showed similar results to the mechanical treatments at Ohio Hills with large increases in sapling density as trees grew into this size category by year 3 (fig. 2G).

Recruitment of oaks is a desirable outcome for both timber and wildlife objectives. Oak sapling density was greatly increased by the mechanical treatment at Ohio Hills and the mechanical+burn treatment at both study sites (figs. 2A and 2B). However, heavy competitors such as yellow-poplar and red maple also increased in number by as many as 1.5 times the number of oaks at the Southern Appalachian Mountains (figs. 2C and 2E) and six times their number at Ohio Hills (figs. 2D and 2F). No treatment was successful at increasing oak sapling density without an equal or greater increase in the density of red maple or yellow-poplar.

Cover of the shrub layer at the Southern Appalachian Mountains was unaffected by the burn-only treatment (fig. 3). The mechanical treatment, both with and without burning, was more effective at removing this vertical fuel layer than was burning alone, primarily because burning did little to remove the rhododendrons (fig. 3B) which grew in moist areas that did not burn. Mountain laurel cover was significantly reduced the year following the mechanical-only and mechanical+burn treatments (fig. 3A). Burning again after 3 years essentially eliminated this species from the shrub layer from burn-only and mechanical+burn plots as opposed to the mechanical-only plots where mountain laurel is growing tall enough to reenter this layer. The predominant shrubs at Ohio Hills are different species from those at the Southern Appalachian Mountains, consisting of *Rubus* and *Smilax* species. *Rubus* was essentially absent from plots before mechanical or burning treatments (fig. 3A) but increased significantly by year 4 as these species responded to canopy opening and grew tall enough to reach the shrub layer. *Smilax* (fig. 3B) was reduced by all active treatments during the first year but was beginning to return to pretreatment levels by year 4 in mechanical and mechanical+burn treatment areas.

Soil Organic C

During the first posttreatment growing season, soil organic carbon (C) content was significantly affected by restoration treatment in both study sites. At the Southern Appalachian Mountains site, all three manipulative treatments resulted in reduced soil organic C content, with an average reduction of 15.6 percent relative to the control (fig. 4). At the Ohio Hills site, the only statistically significant difference was between the burned units and mechanical+burn units, with soil organic content at the latter 25.7 percent greater than the former (fig. 4). The significant effect of the restoration treatments on soil organic C persisted through the fourth posttreatment growing season at the Ohio Hills site, but not at the Southern Appalachian Mountains site. Fourth-year soil organic C content in the mechanical treatment in the Ohio Hills site was significantly greater, by an average of 28.5 percent, than that in the other three treatments (fig. 4).

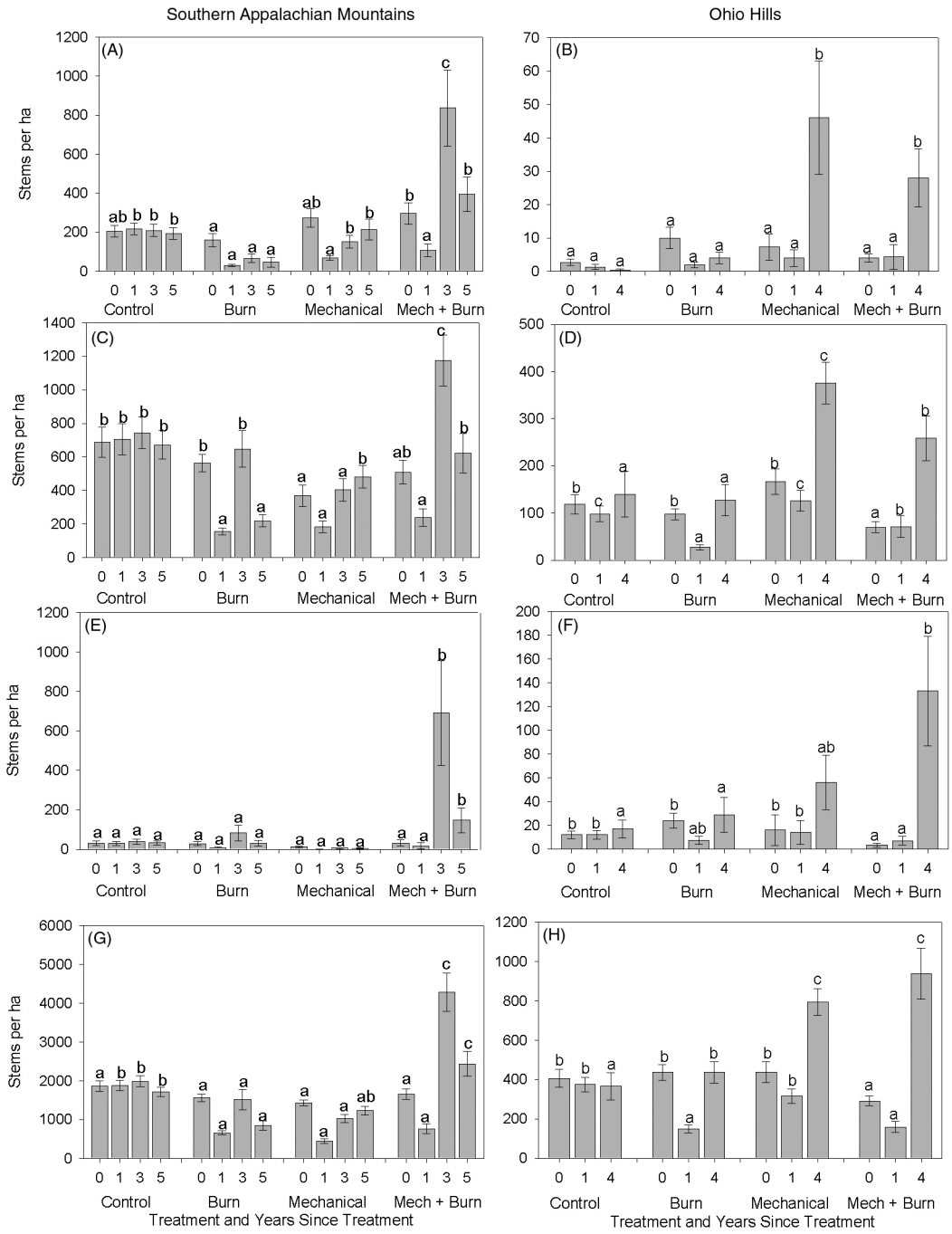

Figure 2—Density of hardwood saplings (stems/ha) by treatment and year for select species and species groups at the Southern Appalachian Mountains (A—oak, C—red maple, E—yellow-poplar, G—all species) and Ohio Hills (B—oak, D—red maple, F—yellow-poplar, H—all species) sites of the National Fire and Fire Surrogate Study. Error bars indicate differences among years within a treatment. Letters indicate differences among treatments within a year (from Waldrop and others 2008).

During the first posttreatment year, soil C to nitrogen (N) ratio was significantly affected by treatment in both study sites. At the Southern Appalachian Mountains site, C to N ratio decreased (and therefore soil organic matter quality increased) in the order: mechanical > mechanical+burn = control > burn (fig. 5). At the Ohio Hills site, the magnitude of the difference among treatments was less than it was at the Southern Appalachian Mountains; however, the mechanical treatment still had significantly greater soil C to N ratio than did the other three treatments at the Ohio Hills (fig. 5). The significant effect of the treatments on soil C to N ratio persisted into the fourth posttreatment growing season at the

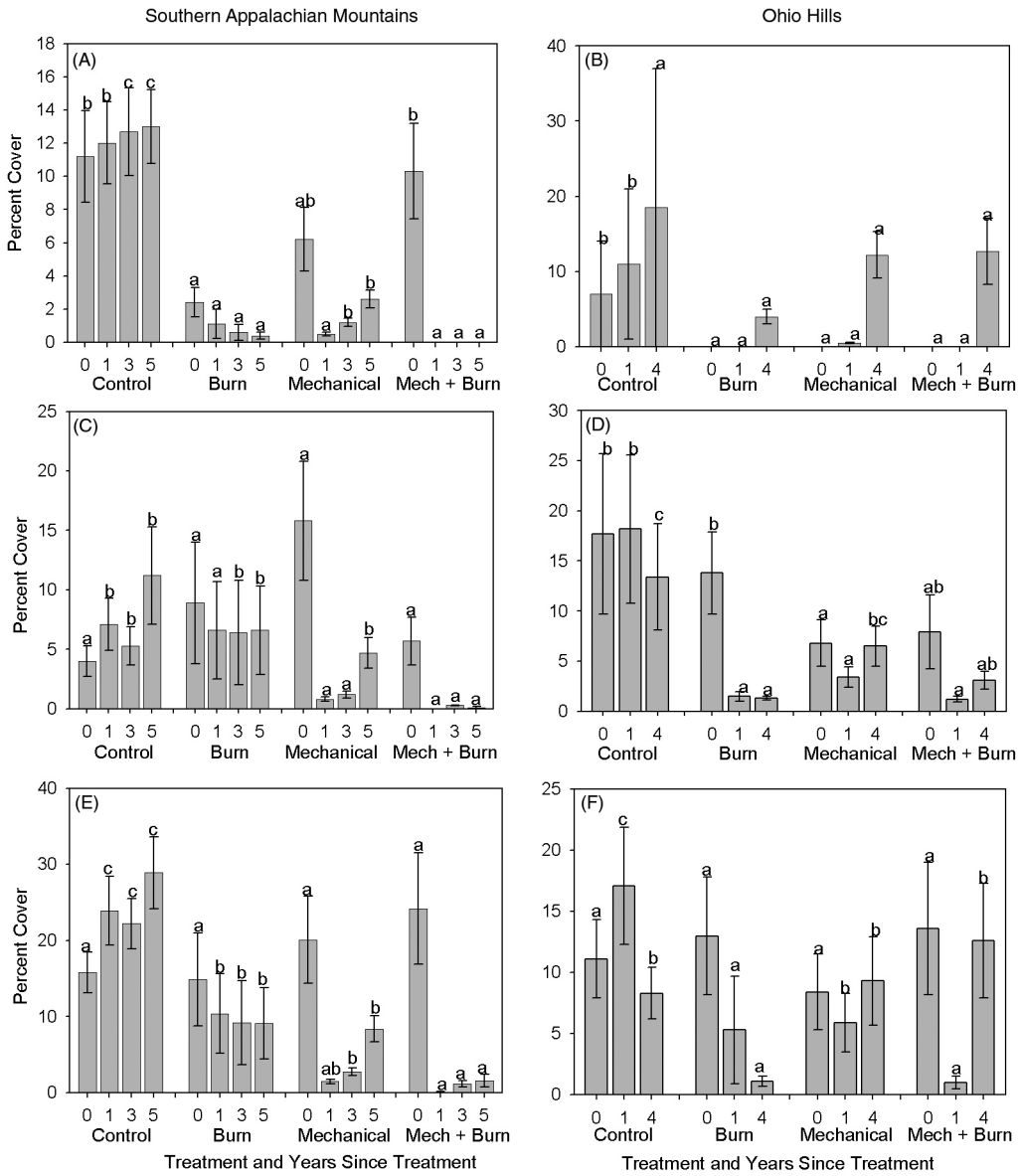

Figure 3—Percent cover of shrubs by treatment and year for select species and species groups at the Southern Appalachian Mountains (A—mountain laurel, B—rhododendron, C—all species) and Ohio Hills (A—*Rubus*, B—*Smilax*, C—all species) sites of the National Fire and Fire Surrogate Study. Error bars indicate differences among years within a treatment. Letters indicate differences among treatments within a year (from Waldrop and others 2008).

Southern Appalachian Mountains, but not in the Ohio Hills. At the Southern Appalachian Mountains site, soils from the mechanical plots had significantly greater C to N ratio during the fourth year than did soils from the other treatments (fig. 5).

Breeding Bird Species Richness

Species richness of breeding birds at the Southern Appalachian Mountains site was not significantly different among treatments during the pretreatment year (2001) or for 2 years after treatments were initiated (fig. 6A). However, richness increased in the burn-only and mechanical+burn treatment areas as overstory trees continued to die and as these treatment areas became more open. Richness was significantly higher in mechanical+burn areas the third year after treatment than in other areas. By the fourth year, both burn treatment areas had significantly higher species richness than did control or mechanical treatment areas and the mechanical+burn treatment had the highest richness of all treatments. Mortality of overstory trees was slower in the burn-only areas than in the mechanical+burn areas suggesting that species richness may continue to increase if trees continue to die.

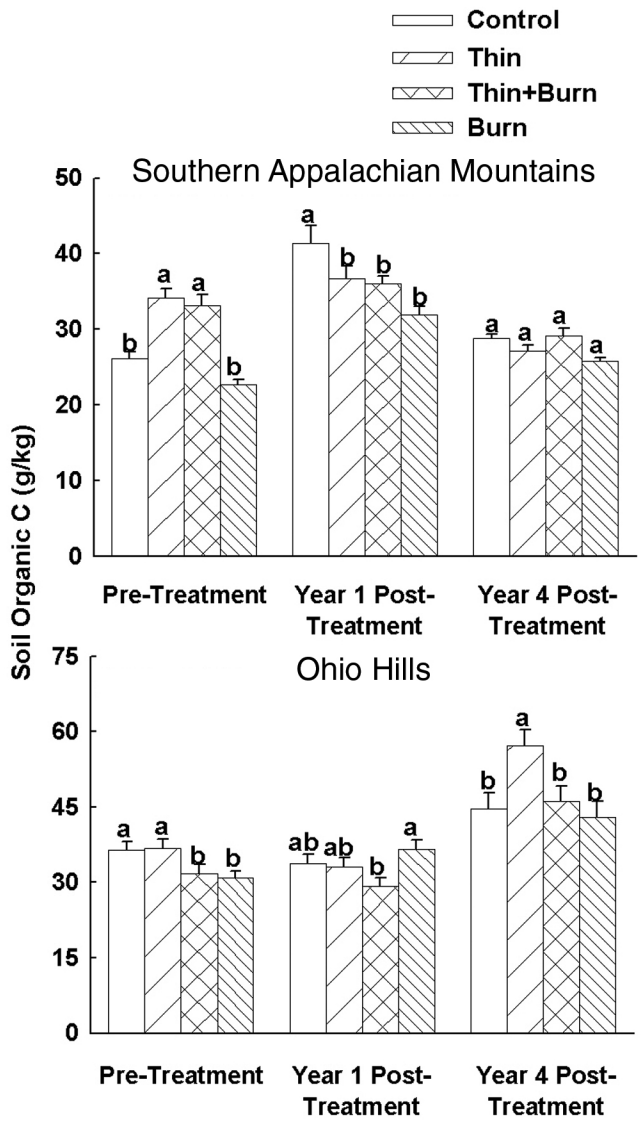

Figure 4—Changes in soil organic carbon (C) content (g C/kg) in relation to four forest restoration alternatives. Histogram bars represent means of n = 60 with standard errors of the means indicated. For site-year combinations in which there were significant treatment effects, histogram bars indicated by the same lowercase letter were not significantly different at $P < 0.05$ (from Boerner and others 2008).

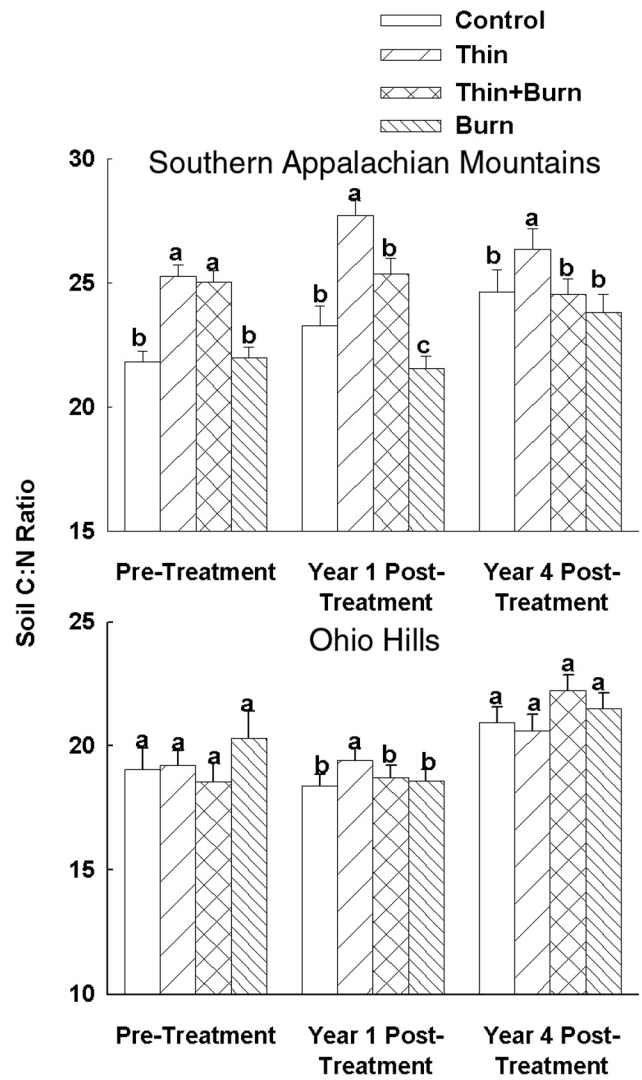

Figure 5—Changes in soil organic matter carbon to nitrogen ratio in relation to four forest restoration alternatives. Format follows figure 4 (from Boerner and others 2008).

Breeding birds responded to reductions in basal area at the Ohio Hills site is a similar manner to that of the Southern Appalachian Mountains site. However, the response to specific treatments differed because of the difference in mechanical treatments. In Ohio, richness was significantly higher in mechanical-only and mechanical+burn sites the first 3 years after treatment (2001 to 2003) (fig. 6B); differences were greatest the third year after treatment. At this site, the canopy was opened by the mechanical treatment (thinning) as opposed to prescribed burning at the Southern Appalachian Mountains site. Breeding bird richness decreased in all four treatment areas between the third and fourth years after treatment in Ohio, but differences between thinned and nonthinned plots remained significant.

DISCUSSION

Restoration of hardwood ecosystems of the Southern Appalachian Mountains and central Appalachian region is challenging because they have been protected from fire for decades, resulting in dramatic changes in stand structure. Treatments selected for the two hardwood sites of the FFS study provide a range of options for restoration which included commercial thinning at the Ohio Hills site, chainsaw felling of saplings and shrubs at the Southern Appalachian

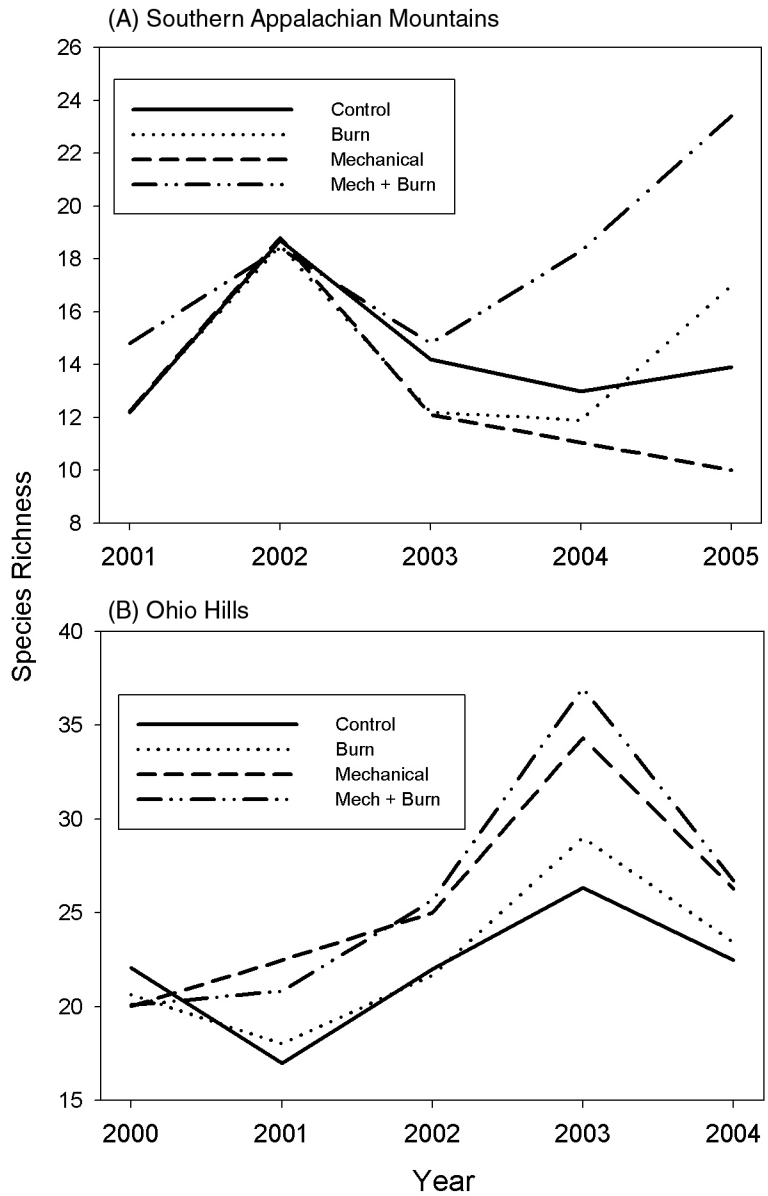

Figure 6—Species richness of breeding birds in four forest restoration alternatives: prescribed burn, mechanical understory reduction, mechanical+burn, and controls for the (A) Southern Appalachian Mountains and (B) Ohio Hills sites of the National Fire and Fire Surrogate Study [Southern Appalachian data were developed from Greenberg and others (2007)].

Mountains site, winter prescribed burning at both sites, and a combination of mechanical and burning treatments at both sites. Each treatment was designed to restore open woodland conditions by altering stand structure (mechanical treatments), reintroducing an ecosystem process (fire), or both. Historical accounts of soil characteristics lead us to postulate that the restoration goal for eastern forests would be soil subsystems that are lower in available nutrients (especially inorganic N), higher in soil organic matter, and lower in both soil organic matter quality and microbial (especially bacterial) activity (Boerner and others 2008). These systems were once driven by low rates of nutrient turnover and from a reservoir of C and N that was persistent and recalcitrant. Objectives for restoration of bird habitat are somewhat less clear. However, vegetation structure has a strong impact on the diversity and composition of bird communities. Disturbance can enhance diversity at stand and landscape scales by creating a mosaic of habitats or vegetation types. A number of bird species require habitat that has been recently disturbed by fire or by large-scale, high-intensity disturbance (Greenberg and others 2007).

None of the treatments at the Southern Appalachian Mountains or Ohio Hills was entirely successful at producing

the stand structure of open woodlands, but each began the process of restoration. The burn-only treatment at both sites created some overstory mortality but stand basal area was reduced only slightly as surviving trees continued to grow. Mortality was continuing at Ohio Hills and may eventually result in more open stands. This result was unexpected as we assumed that most mortality would occur during the first year after treatment. After 6 years, the mechanical treatments were the most effective at opening the overstory but continued mortality in the burn-only plots may lower basal area to levels lower than that of the mechanical treatments. Fire reduced the sapling and shrub layer at both sites. This is especially important at the Southern Appalachian Mountains where dense mountain laurel can act as a vertical fuel. Tree regeneration was abundant at both sites including oaks, red maple, and yellow-poplar. A single burn at Ohio Hills decreased oak regeneration but increased the density of its competitors. At the Southern Appalachian Mountains, two burns seemed to favor oak regeneration by severely reducing density of red maple and yellow-poplar.

The immediate response of the organic matter complex to the restoration treatments was a reduction in soil organic matter quantity by all three treatments in the Southern Appalachian Mountains site and by the combination of thinning and burning in the Ohio Hills site. This was accompanied by a reduction in soil organic matter quality (as indicated by an increase in soil C to N ratio) in response to mechanical treatments in both sites and an improvement in soil organic matter quality as a result of fire alone at the Southern Appalachian Mountains site. By the third- or fourth-growing season after treatment, however, these effects had begun to dissipate and were limited to an elevation in soil organic matter content at the Southern Appalachian Mountains site and a reduction in soil organic matter quality at the Ohio Hills site, both of which were induced by the mechanical treatment.

Species richness of breeding birds did not increase immediately after treatment but seemed to respond positively as hardwood continued to die, especially at the Southern Appalachian Mountains site. The mechanical+burn treatment produced the quickest and largest change in species richness at both study sites. However, the open conditions created by thinning at Ohio Hills and hot prescribed fires at the Southern Appalachian Mountains are ephemeral unless either burning or mechanical fuel-reduction treatments are continued.

CONCLUSIONS

All treatments at both the Southern Appalachian Mountains and Ohio Hills sites effectively altered stand structure, but none created stands with open canopies and absent midstory. The mechanical+burn treatment showed the quickest movement toward restoration goals, but understory vegetation in eastern hardwood ecosystems regenerates and grows quickly without repeated control treatments. In addition, we saw no indication that mechanical/structural restoration actually produced changes in the desired direction for soil organic C. A single fire-based/functional treatment did offer some possibility of restoration progress repeated entries at intervals of 3 to 8 years might be necessary. Bird diversity responded positively to open canopy conditions which were best provided by the mechanical+burn treatment at both study sites, but those habitat conditions are short lived. Our results indicate that restoration of eastern hardwood forests cannot be accomplished with one or two entries. Rather, restoration is likely to be a complex, lengthy, and resource consumptive process.

ACKNOWLEDGMENT

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LITERATURE CITED

Boerner, R.E.J.; Coates, T.A.; Yaussy, D.A.; Waldrop, T.A. 2008. Assessing ecosystem restoration alternatives in eastern deciduous forests: the view from belowground. Restoration Ecology. 16(3): 425–434.

Campbell, J.W.; Hanula, J.L.; Waldrop, T.A. 2007. Effects of prescribed fire and fire surrogates on floral visiting insects of the Blue Ridge Province in North Carolina. Biological Conservation. 134: 393–404.

Coates, T. Adam; Boerner, Ralph E.J.; Waldrop, Thomas A.; Yaussy, Daniel A. 2008. Soil nitrogen transformations under alternative management strategies in Appalachian forests. Soil Science Society of America Journal. 72: 558–565.

Greenberg, Cathryn H.; Forrest, T.G.; Waldrop, Thomas. 2010. Short-term responses of macroarthropods to prescribed fire and mechanical fuel reduction in a Southern Appalachian upland hardwood forest. Forest Science. 56(1): 112–121.

Greenberg, Cathryn H.; Tomcho, Aimee Livings; Lanham, J. Drew [and others]. 2007. Short-term effects of fire and other fuel-reduction treatments on breeding birds in a Southern Appalachian upland hardwood forest. Journal of Wildlife Management. 71(6): 1906–1916.

Iverson, Louis R.; Yaussy, Daniel A.; Rebbeck, Joanne [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: 1–12.

Keenan, Scott C. 1998. Soil survey of Polk County, North Carolina. Washington, DC: U.S. Department of Agriculture, Natural Resources Conservation Service. 218 p.

Kilpatrick, Eran S.; Waldrop, Thomas A.; Lanham, Joseph D. [and others]. 2010. An analysis of fuel reduction treatments on herpetofauna across four southeastern sites of the National Fire and Fire Surrogate Study. Forest Science. 56(1): 122–130.

Schwilk, Dylan W.; Keeley, Jon E.; Knapp, Eric E. [and others. 2009. The National Fire and Fire Surrogate Study: effects of alternative fuel reduction methods on tree size structure and fuels. Ecological Applications. 19(2): 285–304.

Stanturf, John A.; Wade, Dale D.; Waldrop, Thomas A. [and others]. 2002. Fire in southern forest landscapes. In: Wear, D.M, Greis, J., eds. Southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 607–630. Chapter 25.

Sutherland, Elaine Kennedy; Hutchinson, Todd F.; Yaussy, Daniel A. 2003. Introduction, study area description, and experimental design. In: Sutherland, Elaine K.; Hutchinson, Todd F., eds. Characteristics of mixed oak forest ecosystems in southern Ohio prior to the reintroduction of fire. Gen. Tech. Rep. NE-299. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station: 1–16.

Tomcho, A.L. 2004. Effects of prescribed fire and understory removal on bird communities in a Southern Appalachian forest. Clemson, SC: Clemson University. 72 p. M.S. thesis.

Van Lear, David H.; Waldrop, Thomas A. 1989. History, use, and effects of fire in the Southern Appalachians. Gen. Tech. Rep. SE-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 20 p.

Waldrop, Thomas A.; Phillips, Ross J.; Simon, Dean M. 2010. Fuels and predicted fire behavior in eastern hardwood and pine-hardwood stands following fire and fire surrogate treatments. Forest Science. 56(1): 32–45.

Waldrop, Thomas A.; Yaussy, Daniel A.; Phillips, Ross J. [and others]. 2008. Fuel reduction treatments affect stand structure of hardwood forests in western North Carolina and southern Ohio, USA. Forest Ecology and Management. 255: 3117–3129.

Yaussy, Daniel A.; Hutchinson, Todd F.; Sutherland, Elaine Kennedy. 2003. Structure, composition, and condition of overstory trees. In: Sutherland, Elaine K.; Hutchinson, Todd F., eds. Characteristics of mixed oak forest ecosystems in southern Ohio prior to the reintroduction of fire. Gen. Tech. Rep. NE-299. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station: 99–112.

Youngblood, Andrew; Metlen, Kerry; Knapp, Eric E. [and others]. 2005. Implementation of the Fire and Fire Surrogate Study—a national research effort to evaluate the consequences of fuel reduction treatments. In: Peterson, Charles E.; Maguire, Douglas A., eds. Balancing ecosystem values: innovative experiments for sustainable forestry. Gen. Tech. Rep. PNW-GTR-635. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station: 315–321.

INTEREST GROUP OPINIONS ABOUT FUEL REDUCTION IN SOUTHERN APPALACHIA

Carin E. Vadala, Robert D. Bixler, and Thomas A. Waldrop[1]

Abstract—Opinions of interested publics and interest groups (n = 640) about fuel reduction (FR) in the Southern Appalachian Mountains were investigated through social survey using both pictorial and written questions. The study identified three discrete groups based on knowledge of forest history in the Southern Appalachian Mountains, attitudes toward social and ecological changes due to FR, credibility of public land management agencies as managers of forests, aesthetics of FR areas, and recreation activity participation. Results identified three groups of concerned publics labeled as conservation oriented, naïve perceptual, and preservation oriented. The conservation-oriented group was accepting of FR for specific reasons; the naïve-perceptual group disliked even minor charred views and stumps; the preservation group was skeptical of FR and characterized by wanting nature to be left alone. Detail rich results provide guidance in constructing different educational and persuasive messages specific to each of these three groups about forest management through FR in Southern Appalachia.

INTRODUCTION

Concerned publics can help managers develop communication strategies that are responsive to people's values and attitudes toward prescribed fire and mechanical fuel reduction. In a survey of forest managers the two top constraints of implementing fuel-reduction techniques were negative public opinion and the close proximity to residential neighborhoods (Haines and others 2001). Much of the research on support of prescribed fire and fuel reduction has been conducted in the Pacific Northwest (Shindler and Neburka 1997; Toman and others 2004, 2006), yet little is known about the Southern Appalachia. Haines and others' (2001) study, as well as the lack of regional research, demonstrates the need to study perceptions of fuel reduction in the Southern Appalachia.

An understanding of the perceptions of prescribed fire and mechanical fuel reduction will help managers develop communication strategies for stakeholders. Approaches to communication include an understanding of source credibility, characteristics of message receivers, channel of delivery, situational factors, and message content (Ajzen 1992).

Most existing studies about public attitude toward fuel reduction have been conducted in the Western United States. One focus of these studies has been source credibility in terms of trust and what roles forest managers and stakeholders should play in fuel reduction. Local and State management agencies were trusted more so than Federal Agencies (Brunson and Shindler 2004, McCaffrey 2004, Shindler and Toman 2003, Toman and others 2006). Additionally, results documented that landowners perceived their role as limited to managing their own property, while expecting land managers to manage the forested areas surrounding their homes.

Characteristics of persons who might receive messages about fuel reduction have also been studied in Western States, focusing on demographics, attitudes of people in fire prone areas, knowledge of forest and fire ecology, knowledge of agencies, and ecological and aesthetic perceptions of fuel-reduction techniques (Shindler and Neburka 1997; Toman and others 2004, 2006). These studies suggested that residents were experienced and knowledgeable about fuel-reduction methods and outcomes.

Studies about channel of delivery, focused on how to communicate about fuel reduction, have echoed general persuasion studies that multiple approaches are needed (Bonar 2007). Several studies in the Western United States have evaluated communication content and strategy. Respondents endorsed interactive communication over static media. Sources of information perceived as useful ranged from high preference for local fire departments while recognizing that main stream media is sometimes useful (Brunson and Shindler 2004, McCaffrey 2004, Shindler and Toman 2003, Toman and others 2006).

Another aspect of communication research is understanding situational factors, combinations of variables that create interest or concern (Ajzen 1992). For instance, people living in fire prone areas in the West rarely viewed themselves as responsible for fuel reduction, only creating defensible space around their homes (McCaffrey 2004). Yet, these same individuals were insistent that they should have a role in local fire planning. Brunson and Shindler (2004) demonstrated that participants agreed that forest managers should use prescribed fire, mechanical fuel reduction, and livestock grazing as tools for minimizing fuel loads. However, they felt that prescribed fire should be used less often when near residential areas. This is evidence of how fuel-reduction

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strategies insensitive to situational and contextual issues might confuse or even anger involved publics.

As the last component of understanding the persuasion process (Ajzen 1992), message content is important. This step generally comes after there is an understanding of the variation in credibility of sources, potential receivers of the message, preferences and availability of channels, and situational factors. No studies testing message content on fuel reduction could be located.

The purpose of the study was to identify segments of stakeholders involved with forest issues in the Southern Appalachian Mountains. To accomplish this, a deeper understanding of stakeholder's fuel-reduction knowledge, preferences, and attitudes was needed.

METHODS

The primary focus of this study was helping managers understand how and in what way to interact with individuals and their associated interest groups. In order for a manager to interact with the public, it is important for them to recognize that a stakeholder has a set of opinions. Knowledge of these opinions will help managers recognize and constructively respond to those concerns. Purposive sampling was used to limit data collection to publics potentially interested or involved with the topics of forest management and fuel reduction.

Research on opinion surveys has indicated that people without formulated opinions will answer survey questions they know nothing about, based only on the information in the wording of the question. For instance, Bishop (2004) has demonstrated that asking about President George W. Bush's social security initiative provides significantly greater levels of support from Republicans than when a question is asked about a social security initiative without reference to President Bush. Bishop (2004) described numerous other public opinion polls where it was obvious that respondents in random opinion polls readily answered questions about issues they knew nothing about. Considering that fuel reduction in Southern Appalachian Mountains is far less salient to the general public than social security, it seems counterproductive and misleading to survey people uninvolved in the topic. Consequently sampling was designed to minimize participation from disinterested parties. Numerous studies have shown that interest and involvement with a topic are the best predictors of participation, and that public opinion polls based on random samples with aggressive followup procedures to encourage high-participation rates provide distorted results in terms of levels of respondents' awareness and involvement with issues (Bishop 2004).

Sample Population

Three types of potential respondents were approached (table 1). Use of this sampling strategy does not produce accurate population parameters, yet this was not the intention of the study. The study purpose is to identify types of people based on existing interests and involvement with issues related to Southern Appalachian forests.

The initial contact was made by regular mail or Internet survey. All homeowners were contacted by mail. Onsite recreationists were given a choice of mail or Internet survey, and clubs and organizations were contacted through an email invitation to complete a Web-based survey. Only one reminder was sent to potential participants to minimize responses from disinterested participants being badgered into participating. The final sample size was 640, adequate for a segmentation study (Evans and Berman 1994). Hierarchical segment analysis was used to identify meaningful segments. Segments were then further described based on attitudes and beliefs.

Measures

The segments were identified with the use of a knowledge test, visual perceptions of photographs, and by asking a series of questions on the acceptability of forest changes that potentially occur due to fuel-reduction practices. The questions on the acceptability of forest changes were gathered from studies mentioned in the literature review and recent outcomes of ecological effects of fuel reduction in the Southern Appalachian Mountains. The items measured social and ecological aspects of fuel reduction. Visual perceptions or acceptability were measured using a set of eight photographs recently subjected to chainsaw felling of shrubs, fire, both, or neither. Factor analysis was used to create composite variables from the scales. A composite score was created for the knowledge test. The 17 questions addressed knowledge of ecological effects of fire and history of the forests in Southern Appalachia. Responses were coded as not sure = 0, incorrect answers = –1, and correct answers = 1. Therefore,

Table 1—Types of respondents approached to participate in this study

Survey group	Technique used
Homeowners	People who lived within census blocks overlapping U.S. Forest Service land
Recreationists	People participating in hiking, equestrian, mountain biking, hunting, fishing, camping, climbing, picnicking, and Revolutionary War reenacting were approached on public forest lands
Interest groups	Conservation, preservation, hiking, and hunting clubs approached by email

the cumulative score was penalized for missing a question but not if the respondent admitted to not knowing the answer.

RESULTS

The first segment identified through cluster analysis was the "preservation oriented" (PO) grouping (table 2). This group was moderately knowledgeable about the ecological effects of fire and the history of the forests in the Southern Appalachian region. The PO values were evident because this group was characterized by high acceptability of increasing the diversity of nongame animals. The group found an increase in dead and standing downed trees as moderately acceptable and had little acceptability of all other outcomes. The PO group preferred forests with visually thick stands of rhododendron (*Rhododendron maximum* L.) and mountain laurel (*Kalmia latifolia* L.). Forests with evidence of charred stumps and floors relatively free of ground plants were less acceptable to this group.

The second segment is the "conservation oriented" (CO) grouping (table 2). This group was highly knowledgeable on historical ecology of Southern Appalachia and the effects of fire. CO individuals prefer that management use fuel-reduction techniques to reduce rhododendron and wildflowers, increase dead material, prevent damage to residential structures, improve habitat for game and nongame species, leave visible signs of fire, and make it easier to walk through the forest. Visual preferences for CO respondents included photographs with open floors, visible penetration, and those with signs of fuel reduction either by fire or mechanical treatments.

The final segment "naïve perceptual" (NP) tended to have the least amount of knowledge about fire, fuel reduction, and historical ecology (table 2). This group found moderately acceptable outcomes that prevent damage to property, decrease soil and water qualities, reduce rhododendron and mountain laurel, and easier to see and walk through the forest. This group found least acceptable outcomes that include improved game and nongame habitat, residual burn marks, and increased standing dead material. Visual preferences that demonstrated management techniques were lowest for this group.

DISCUSSION

One caution is in order. Because of the type of sample and followup procedures used, the study results should not be interpreted as an opinion poll. Percentages of each group holding a particular view that could be inferred from the results tables do not reflect percentages of opinions present in the general public or the relative percentages of each group in the general population.

This research helped to replicate and extend the literature on stakeholders' attitudes and perceptions of forest managers using fuel-reduction techniques. Results indicate that people

Table 2—Standardized mean comparison of segment membership based upon knowledge of ecology and fuel reduction, acceptability of changes, and perceptual evaluation

Variables (reliability score)	Preservation oriented	Conservation oriented	Naïve perceptual
Knowledge of Southern Appalachian ecology and fuel-reduction effects	Medium –0.20	High 0.82	Low –0.57
Changes/acceptability factors			
Decreased soil and water qualities (0.60)	Low –0.30	High 0.31	Medium 0.08
Reduction in rhododendron, mountain laurel, wildflowers (0.84)	Low –0.45	High 0.83	Medium –0.23
Improve habitat for nongame animals (0.88)	High 0.41	High 0.52	Low –1.1
Easier to see and walk through the forests, new plant growth (0.75)	Low –0.77	High 0.87	Medium 0.12
Prevent damage from wildfires to property (0.83)	Low –0.53	High 0.52	Medium 0.10
Residual burn marks on trees and reduce air quality (0.78)	Low –0.41	High 0.98	Low –0.48
Improve game habitat, turkey, deer, trout, and increase blueberry shrubs (0.69)	Low –0.11	High 0.36	Low –0.21
Increase dead standing and downed trees (0.78)	Medium 0.24	High 0.38	Low –0.61
Perceptual evaluation			
Charred areas evident with sprouting stumps, moderate visibility (0.85)	Medium –0.01	High 0.37	Low –0.33
Forest floor with rhododendron, limited visibility (0.68)	High 0.43	Medium –0.02	Low –0.48
Deep visual penetration, smooth ground surfaces (0.72)	Low –0.06	High 0.23	Low –0.14

support fuel-reduction techniques for various reasons. A deeper understanding of the different segments will help managers to recognize and respond to the concerns of stakeholders based upon existing attitudes and perspectives.

An important part of persuasive communication involves understanding the situational factors that help create interest or concern (Ajzen 1992). At least one study indicates that negative public opinion in the Southeastern United States is the top-ranked barrier to implementing fuel reduction (Haines and others 2001). Thus it is important for managers to understand the characteristics of individuals who possess negative public opinions on fuel reduction. An understanding of characteristics such as attitudes, knowledge of fire ecology, knowledge of agencies, and perceptions of fuel reduction (Shindler and Neburka 1997; Toman and others 2004, 2006) will help managers develop an open and interactive communication process.

APPLICATIONS

This study should help in developing communication strategies in that it provides detailed description of knowledge, attitudes, and perceptions of stakeholders. The NP segment was least to moderately concerned about attitudes, perception, and knowledge. This suggests less interest than the other segments and thus peripheral routes to persuasion may be effective. This group is least likely to respond to messages that are developed with evidence and logic, and care must be used in the construction of such messages. PO and CO publics are more likely to respond to rational arguments that are consistent with existing attitudes, values, and perceptions (Petty and Cacioppo 1986).

This study demonstrated that interested stakeholders vary in their acceptability of fuel-reduction techniques. Managers can use this information to judge whether fuel reduction is the best option depending on the potential characteristics of the technique used and who will directly experience the outcome. CO individuals prefer open space with deep visual penetration and habitat where game species can easily be hunted. PO individuals prefer increases in nongame species. The lack of tolerance by the NP segment for residual burn marks, charred stumps, and standing dead or downed trees suggests minimizing signs of fuel-reduction techniques along roadsides, picnic areas, and trails.

There are several social and ecological outcomes of fuel reduction. Depending on knowledge, attitudes, and perceptions of stakeholders, the outcomes can either be acceptable or unacceptable. The results of this study suggest that PO and CO segments have the knowledge and attitudes in order to respond well to rational arguments about fuel reduction. PO individuals are more likely to respond to ecological arguments involving nongame species while CO individuals will respond to both ecological and utilitarian outcomes. The NP group will most likely be a challenging segment to communicate with due to their lack of knowledge, low attitude, and perceptions of fuel-reduction outcomes. Thus, communication techniques may need to focus on persuasion instead of rational education messages.

This research demonstrates the variability of interested publics and suggests that communication techniques vary depending on the characteristics of the audience. The use of persuasive vs. educational techniques should vary depending on the audience. Attitudes or interested publics need to be accounted for when managers respond to concerns and questions. This research will be useful to help managers identify different segments and shape messages based on interests on concerned stakeholders.

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LITERATURE CITED

Ajzen, I. 1992. Persuasive communication theory in social psychology: a historical perspective. In: Manfredo, Michael J., ed. Influencing human behavior: theory and applications in recreation, tourism, and natural resources management. Champaign, IL: Sagamore Publishing Co.: 1–28.

Bishop, G.F. 2004. The illusion of public opinion: fact and artifact in American public opinion polls. Lanham, MD: Rowman & Littlefield Publishers, Inc. 248 p.

Bonar, S.A. 2007. A conservation professional's guide to working with people. Washington, DC: Island Press. 198 p.

Brunson, M.; Shindler, B. 2004. Geographic variation in social acceptability of wildland fuels management in the Western United States. Society and Natural Resources. 17: 661–678.

Evans, J.R.; Berman, B. 1994. Marketing. New York: Macmillan Publishing Co. [Number of pages unknown].

Haines, T.K.; Busby, R.L.; Cleaves, D.A. 2001. Prescribed burning in the South: trends, purpose and barriers. Southern Journal of Applied Forestry. 25(4): 149–153.

McCaffrey, S.M. 2004. Fighting fire with education: what is the best way to reach out to homeowners? Journal of Forestry. 102(5): 12–19.

Petty, R.E.; Cacioppo, J.T. 1986. Communication and persuasion: central and peripheral routes to attitude change. New York: Springer-Verlag. [Number of pages unknown].

Shindler, B.; Neburka, J. 1997. Public participation in forest planning: 8 attributes of success. Journal of Forestry. 95(1): 17–19.

Shindler, B.; Toman, E. 2003. Fuel reduction strategies in forest communities: a longitudinal analysis of public support Journal of Forestry. 101(6): 8–15.

Toman, E.; Shindler, B.; Brunson, M. 2006. Fire and fuel management communication strategies: citizen evaluations of agency outreach activities. Society and Natural Resources. 19: 321–336.

Toman, E.; Shindler, B.; Reed, M. 2004. Prescribed fire: the influence of site visits on citizen attitudes. J. Environ. Educ. 35(3): 13–17.

SOILS AND WATER

View of the Falls on the Little Missouri River, Caddo Ranger District, Ouachita National Forest, in Montgomery County, Arkansas. (Photo by James M. Guldin)

EROSION RESPONSE OF A HARVESTED PIEDMONT LOBLOLLY PINE PLANTATION IN ALABAMA: PRELIMINARY RESULTS

Emily A. Carter[1]

Abstract—The erosion impact of typical forest management operations in a loblolly pine (*Pinus taeda* L.) plantation in the Piedmont region of Alabama was investigated. Soil loss and runoff were highly variable throughout postharvest and first year after site preparation and planting. Under postharvest conditions, the annual rate of soil loss was 106.5 and 274.4 kg/ha in two locations (DIST1 and DIST2) of the harvest tract while annual rate of runoff was 141.4. and 200.4 mm/ha, respectively. The annual rate of soil loss after site preparation/planting varied by treatment in which orientation of planted beds or no beds were tested for its influence on erosion. The treatments consisted of beds oriented down the slope (DTS), across the slope (ATS), no bedding, and machine planting only (MPO). Soil loss was greatest on sites where beds were oriented DTS followed by sites subjected to MPO with no soil disruption prior to establishment of planting beds (MPO). The annual rate of soil loss from DTS and MPO was estimated to be 14 520 and 774 kg/ha and stands in contrast to erosion rates of 79.2 and 67.0 kg/ha for sites where beds were oriented ATS and where no disturbance took place, respectively. Nutrient mobilization was evaluated by determining chemical constituents of runoff collected after each precipitation event. In the postharvest phase, the total amount of base cations determined in runoff was estimated to be 6.81 and 4.60 kg/ha for DIST1 and DIST2, respectively. Cumulative totals of base cations were estimated to be 9.67 kg/ha in DTS and 3.57 and 3.19 kg/ha in MPO and ATS, respectively. Calcium and potassium were the most abundant elements in runoff while aluminum increased in runoff compared to postharvest conditions.

INTRODUCTION

Soil erosion is an ongoing process in forested landscapes contributing approximately 9.1 by 10^6 Mg of suspended sediment per year to watercourses within the contiguous United States (Fowler and Heady 1981). Forest landscapes often attenuate soil erosion through the rapid infiltration of rainfall, and reduction in rainfall impacts energy by ground cover (Dissmeyer and Foster 1981, Moore and others 1986). However, when ground cover is disturbed, soil is displaced and transported in overland channel flow (rills) or sheet flow (interrills) (Owoputi and Stolte 1995). These processes can act individually or together and can displace and transport significant quantities of soil from a landscape. Some forest operations may have accelerated soil erosion of which harvesting, road building, and mechanical site preparation are the main contributors (Yoho 1980). The final erosion response varies according to the interaction between the type of activity under evaluation and local site conditions including site factors related to rainfall erosivity, soil erodibility, slope length, and steepness and cover condition (Dissmeyer and Foster 1981, Kinnell and Cummings 1993, McIntyre and others 1987).

Erosion rates associated with pine production in the Southeastern United States, primarily loblolly pine (*Pinus taeda* L.), have been previously reported and found to be highly variable (Beasley and others 1986, McBroom and others 2008, Yoho 1980). Machine trafficking during harvesting has resulted in increased runoff due to compacted soil layers (increased bulk density) that results in loss of soil porosity and water infiltration which in turn increases runoff (Greacen and Sands 1980, Shaw and Carter 2002). Further changes may occur during site preparation and planting that includes mechanical manipulations to remediate compacted soil layers but can result in significant losses of sediment and nutrients (Blackburn and Wood 1990, Pye and Vitousek 1985). The resulting planting site may be significantly compromised in subsequent biomass production due to previous loss of soil and nutrients (Merino and others 2005, Van Oost and others 2006).

OBJECTIVE

The objective of the study was to measure sediment loss, runoff, and nutrient mobilization from select areas of a loblolly pine plantation in the Piedmont region of Alabama subjected to harvesting and site preparation/replanting.

MATERIALS AND METHODS

The study site was located in a 20-year-old loblolly pine plantation in Lee County, AL, and encompassed an area approximately 25.4 ha in size (fig. 1). Tree basal area was estimated to be 27.5 m^2/ha of loblolly pine and 4.6 m^2/ha of hardwoods with an expected yield of 202.1 Mg (green)/ha. The primary soil series within the harvest tract was mapped as a Gwinnett sandy loam, a member of the fine, kaolinitic, thermic Rhodic Kanhapludult family (Soil Conservation Service 1981). A harvest operation was conducted in winter/spring 1998 utilizing a conventional harvesting system: a feller buncher working in conjunction with grapple skidders pulling to two separate decks; production averaged approximately 180 Mg (green)/day. The tract remained in a postharvest condition for approximately 14 months until aerial application of herbicides in May 1999, and mechanical site preparation that consisted of shearing planting rows with a V-Blade, followed by a single pass of a Savannah combination plow configured for bedding capabilities in November/December 1999. Pine seedlings were machine planted in late January 2000 by inserting seedlings in slots opened by a small shank and soil repacked by packing wheels.

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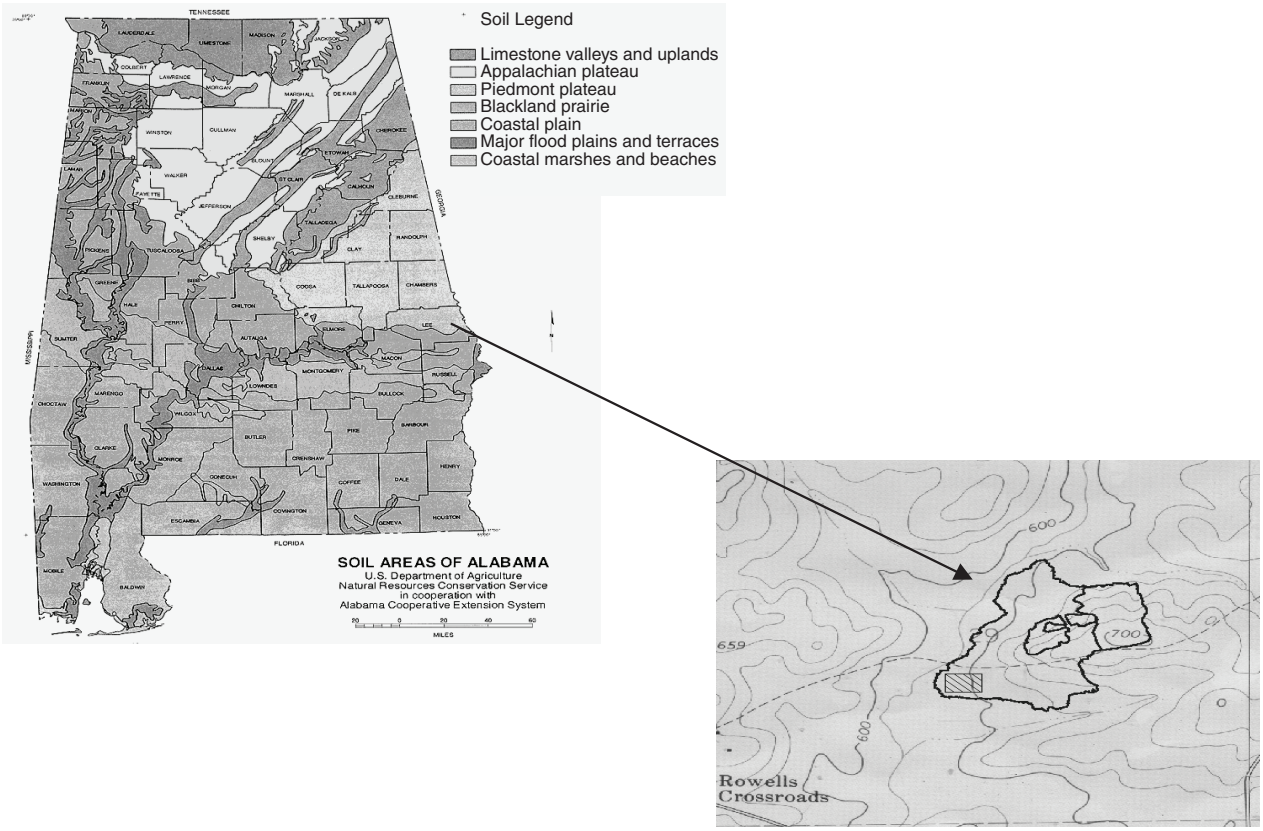

Figure 1—Location of loblolly pine plantation study site subjected to harvesting and site preparation and planting in the Piedmont region of Alabama. *Source:* Natural Resources Conservation Service (www.mo15.nrcs.usda.gov/states/al_soils_graphic.html)

Erosion Collection System

Steel-framed plots, approximately 5.5 by 2 m in size, were installed in select locations of the study site to monitor runoff and sediment production from areas disturbed by harvest and regeneration operations (fig. 2). Each location contained three framed plots that were installed on similar soil types and slope steepness (~10 percent). Runoff and entrained sediment were routed through a PVC pipe to a 210-L collection barrel placed downslope from the plot outlet; runoff was measured and sediment samples were collected after each rainfall event. Plots were installed in the fall of 1998 to assess erosion in two areas of the tract disturbed by harvest activities, labeled DIST1 and DIST2, and a third set installed in an undisturbed forest (CON) adjacent to the harvest tract in winter 1999. Site preparation activities commenced in fall of 1999 at which time each framed plot was removed to accommodate shearing and bedding of the harvest tract, replaced temporarily for 1 month prior to planting, removed to accommodate machine planting, and reinstalled after machine planting of seedlings. Steel frames (three each) were reinstalled after site preparation in a downslope direction to assess the influence of bedding on erosion. Treatments consisted of beds oriented across the slope (ATS), down the slope (DTS), the original undisturbed site (CON), and an area subjected to machine planting only (MPO) with no mechanical site preparation within each frame.

The two sites labeled ATS and MPO were established on postharvest sites DIST1 and DIST2, respectively. A tipping bucket rain gauge and plastic static rain gauge were placed in an elevated section of the harvest tract to measure rainfall quantities and intensities. Descriptions of plot characteristics and sediment and runoff yields for selected time periods have been previously published (Grace 2004).

Sample Collection and Processing

After each rainfall event, runoff was estimated by measuring the collected runoff depth in each barrel, and samples collected to determine the amount of total delivered sediment produced by each storm event. Total delivered sediment was defined as the quantity of suspended and deposited sediment. Suspended sediment was determined by removing a 500-ml aliquot of standing runoff from each collection barrel on each sampling date and processed according to American Public Health Association (1995) method number 2540D. Deposited sediment was collected by bailing the runoff from each collection tank until approximately 7 cm of standing runoff remained and transferring deposited material and runoff to an 18.9-L bucket that was returned to the laboratory for processing. Deposited sediment was sieved to separate sand size material from silt and clay size material (53-µm sieve). These separates were placed in containers and dried

Figure 2—Photographic images of steel-framed plots and runoff collector barrels utilized in estimation of sediment loss, runoff, and nutrient mobilization in a loblolly pine plantation in the Piedmont region of Alabama.

in a forced air oven at 105 °C. Dry weights were recorded for each plot by sampling date and averaged over the three plots for each sampling date.

At the time of the sampling for suspended sediment analysis, a separate aliquot (500 ml) of suspended material sample was collected for nutrient analysis. The sample was sent to an independent laboratory for analysis and select nutrients measured by plasma emission spectrometry (ICP-AES) (Soltanpour and others 1996). Runoff samples were filtered through a 0.45-µm membrane filter to separate solution from suspension phase prior to analysis for base cations, aluminum, and nitrate-nitrogen.

Statistical Analyses

The experimental design consisted of a randomized block design with three replications in which three treatments were evaluated in the postharvest phase and four treatments in the postsite preparation and planting portion. A PROC MIXED (SAS Institute Inc., Cary, NC) procedure was used to evaluate the significance of treatment on total sediment production and runoff during postharvest and postsite preparation phases. Treatment was determined to be a fixed effect, and day of year was designated as the random effect. Means were separated by least squares. Similar procedures were used to evaluate nutrient mobilization expressed as annual mean loss of base cations (BASECATS), aluminum (Al), and nitrate-nitrogen (NO_3-N).

RESULTS AND DISCUSSION

Post Harvest Results

The amount of sediment loss and runoff monitored during a portion of the postharvest phase (January to October 1999) by sampling date is depicted in figure 3; cumulative totals and mean estimations of sediment and runoff by location are included in table 1. In general, sediment loss and runoff were higher in areas (DIST1 and DIST2) that had experienced machine trafficking during harvest operations compared to the CON area (table 1). The amount of soil loss and runoff varied during the course of the time period under evaluation. In this study, differences in the erosion response of the two disturbed areas (DIST1 and DIST2) were noted with runoff and sediment yields higher from DIST2. The average quantity of sediment removed during the postharvest phase was significantly higher ($P < 0.001$) from DIST2 than from DIST1 and CON. Similarly, runoff was significantly different ($P < 0.001$) under postharvest conditions, and significant differences were detected among treatments. A potential explanation for this result may be spatial variability in the soil surface physical condition due to changes in soil structure in response to machine movements during harvest operations. Previous research has noted soil erosion to be influenced by the impact of machine traffic on soil structure through reductions in porosity and water infiltration (Meyer and Harmon 1989, Voorhees and others 1979, Wendt and others 1986). Machine traffic during harvesting was more intensive in portions of the harvest tract and less intense in other areas and may have altered soil physical conditions and response accordingly (Carter and others 2000).

Sediment collected after each rainfall event was separated into sand and silt/clay fractions and the silt/clay fraction was higher proportion regardless of site disturbance (table 1).

Site Preparation and Planting Results

The implementation of site preparation/planting increased sediment loss and runoff in the first year compared with the postharvest period (table 1). Among the treatments, soil loss and runoff were greatest from DTS (log scale to include all data) followed by MPO and ATS, and as with the postharvest condition, soil loss and runoff varied by storm event (figs. 4A and 4B). Soil loss from ATS corresponded closely to quantities measured in DIST1 while sediment yields measured from MPO increased in comparison to the previous site condition (DIST2). As was observed with postharvest results, the sediment fraction had higher levels of silt/clay than sand with the exception of DTS (table 1). Cumulative runoff production was greatest from DTS, as would be expected, followed by MPO and ATS; the lowest runoff production occurred in CON (table 1). Sediment loss ($P < 0.0001$) and runoff ($P < 0.001$)

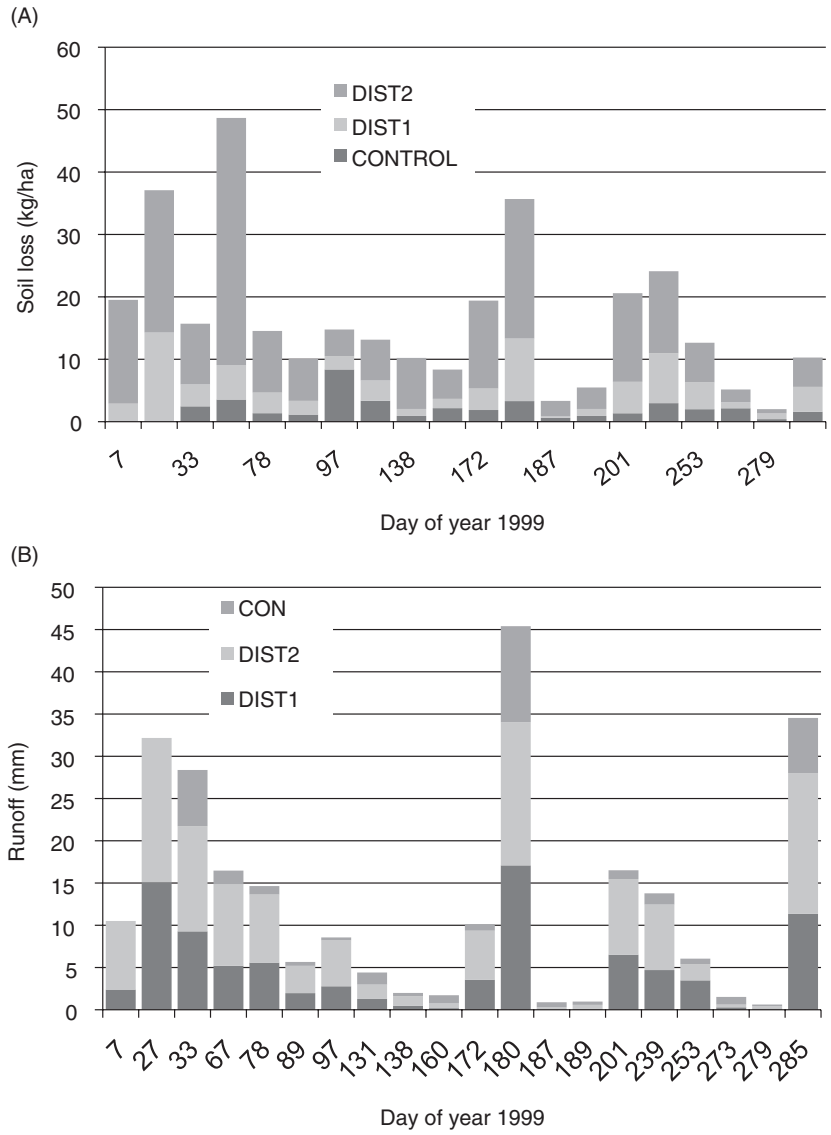

Figure 3—Sediment loss (kg/ha) (A) and runoff (mm) (B) quantities from a harvested loblolly pine plantation in the Piedmont region of Alabama. (DIST1 = plot, DIST2 = plot, CON = undisturbed forest)

were found to be significantly different for treatments under evaluation with DTS results significantly greater than ATS, MPO, and CON. No significant differences were detected among the three remaining treatments, but the higher levels of sediment loss and runoff associated with DTS may have skewed the data, obscuring significant differences.

The amount of sediment loss and runoff in response to site preparation may be related to surface conditions and bed orientation. The highest amount of sediment loss and runoff occurred in plots in which beds were positioned DTS, which permitted runoff to be channeled rapidly downslope. Soil loss and runoff quantities measured in DTS underscored the influence of slope, lack of vegetative cover, and soil erodibility on erosion potential (Kinnell and Cummings 1993, Meyer and Harmon 1989, Stein and others 1986, Van Oost and others 2006). Surface soil left unprotected is prone to erosion through the disruption of soil aggregates by rainfall and subsequent release of soil particles; this is especially evident in soils dominated by silt and clay size fractions similar to the textural composition of our study site (Burroughs and others 1992, Miller and Baharuddin 1987). In contrast, sediment loss and runoff were substantially lower from ATS plots where shorter runoff distances between beds intercepted water flow and potentially reduced sediment loss. Runoff results from MPO indicated levels elevated in comparison to ATS but substantially more sediment loss and may be the result of the tillage effect imposed during planting of seedlings.

Table 1—Cumulative, annual rate, mean soil loss, and runoff as a result of harvesting and site preparation/planting operations in a loblolly pine plantation, Alabama

	Sediment					Runoff		
Treatment[a]	Total	Sand	Silt/clay	Rate	Mean[b]	Total	Rate	Mean[b]
	kg/ha			*kg/ha/year*	*kg/ha*	*mm/ha*	*mm/ha/year*	*mm/ha*
1999								
DIST1	79.5	27.9	54.2	106.5	4.09 a	113.1	141.4	4.58 c
DIST2	212.5	97.9	114.2	274.4	10.64 b	160.3	200.4	6.34 a
CON	40.5	12.8	24.4	43.4	2.21 a	49.3	61.6	1.83 b
2000								
ATS	81.0	37.4	42.8	79.2	7.31 b	66.1	64.8	33.4 b
DTS	12558	6438	6086	14520	1566 a	210.0	205.8	105.2 a
MPO	726.9	302.4	419.5	774.1	86.41 b	88.6	86.8	54.2 b
CON	55.9	16.5	33.5	67.0	8.64 b	32.6	32.0	26.1 b

DIST1 and DIST2 = postharvest disturbance; CON = control; ATS = across the slope; DTS = down the slope; MPO = machine plant only.

[a] Means by treatment year followed by similar letters were not significant at the $P = 0.05$ level.

[b] Means of sediment and runoff were based on $n = 58$ observations from postharvest data and $n = 71$ observations from postsite preparation and replanting data.

Sufficient surface soil was disturbed in this process than when exposed to rainfall, and soil particles were entrained by runoff and transported downslope. Soil disturbances resulting from tillage have often been linked to higher erosion rates, and the increased soil loss in MPO may have resulted from the loosening of an erodible soil (Van Oost and others 2006). Soil loss and runoff in CON would be expected to be less than other treatments and the results of this study generally confirm this expectation.

Nutrient Mobilization

Results of runoff water analyses of each rain event (figs. 5A and 5B) and the mean annual accumulation of select elements are included in table 2. During the postharvest period, nutrient quantities varied by treatment with DIST1 having the largest displacement of calcium (Ca), magnesium (Mg), and potassium (K) while DIST2 was higher in sodium (Na) and Al; CON levels never exceeded DIST1 and DIST2 in the time period examined. Nutrient concentrations expressed as BASECATS were found to be variable throughout the postharvest phase (fig. 5A); similar variability in Al response was observed (data not shown). Among the individual postharvest treatments, the best relationship between BASECATS concentration and runoff was determined to be DIST2:

$$r^2 = 0.30;\ y = 0.0003x^2 - 0.0747x + 7.0807 \qquad (1)$$

and between exchangeable Al and runoff in CON:

$$r^2 = 0.63;\ y = 0.0016x + 0.0285 \qquad (2)$$

Postharvest treatment levels of BASECATS ($P < 0.10$) exceeded CON, and significant differences were detected between DIST1 and CON when mean values were compared (table 2).

In the initial phases of the postsite preparation/planting period, nutrient mobilization was highest in DTS among postsite preparation treatments and lowest in CON (table 2). Runoff nutrient levels varied when ATS and MPO were compared with Ca and Mg (slightly) higher in ATS while K, Na, and Al were higher in runoff from MPO. The amount of nutrient displacement varied greatly by storm event (fig. 5B), but an overall relationship between runoff and nutrient concentration (BASECATS) for all treatments was detected.

$$r^2 = 0.56;\ y = -3.0838\ln(x) + 18.48 \qquad (3)$$

The relationship between runoff and Al was detected for CON only.

$$r^2 = 0.92;\ y = 0.0258x + 0.0241 \qquad (4)$$

Cursory examination of the relationship between runoff and sediment load indicated no relationship under postharvest

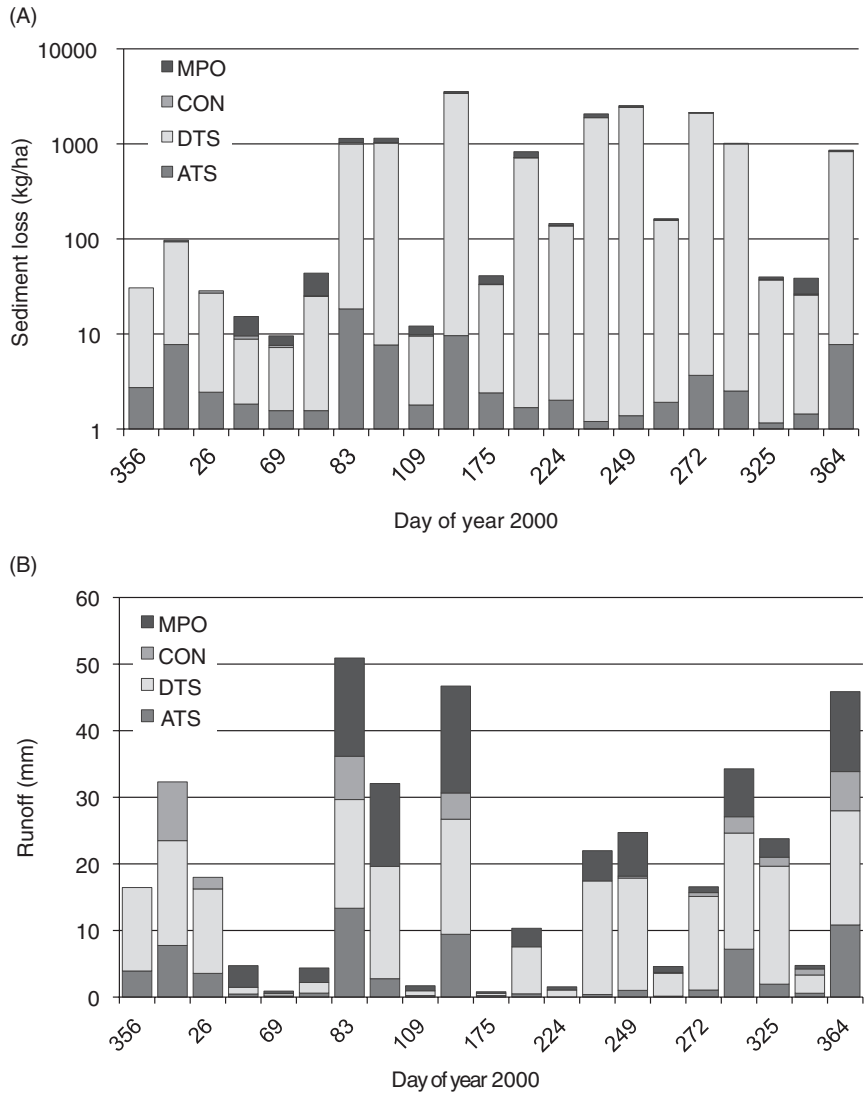

Figure 4—Sediment loss (kg/ha) (A) and runoff (mm) (B) from a loblolly pine plantation subjected to site preparation/planting in the Piedmont region of Alabama. (MPO = machine planting only, CON = undisturbed forest, DTS = down the slope, ATS = across the slope)

conditions while sediment concentration and runoff under postsite preparation conditions appeared to be related.

$$r^2 = 0.46;\ y = 7.4775x^2 - 61.894x + 53.62 \qquad (5)$$

Treatments were highly significant ($P < 0.001$) for the mean quantity of BASECATS displaced with DTS significantly different from other treatments when mean values were compared and MPO significantly different from CON (table 2).

Nutrients entrained in runoff from sites that have been subjected to mechanical manipulations have been previously reported in forested and agricultural settings (Blackburn and Wood 1990, Kleinman and others 2006, Pye and Vitouseck 1985). The mechanism by which nutrients are mobilized by runoff may be the result of a process by which surface water via rainfall, runoff or infiltration mixes with soil constituents, thereby, transferring soil nutrients to soil solution and eventually to surface runoff. The depth of mixing is believed to be 3 to 4 mm with upward movement of nutrients to the mixing zone possible (Kleinman and others 2006, Zhang and others 1997). The overall result of nutrient mobilization would be the formation of zones or areas of nutrient enrichment or depletion in a dissected landscape such as was evaluated in this study with the potential to impact site productivity (Ni and Zhang 2007, Papiernik and others 2009). Studies of highly weathered soils under

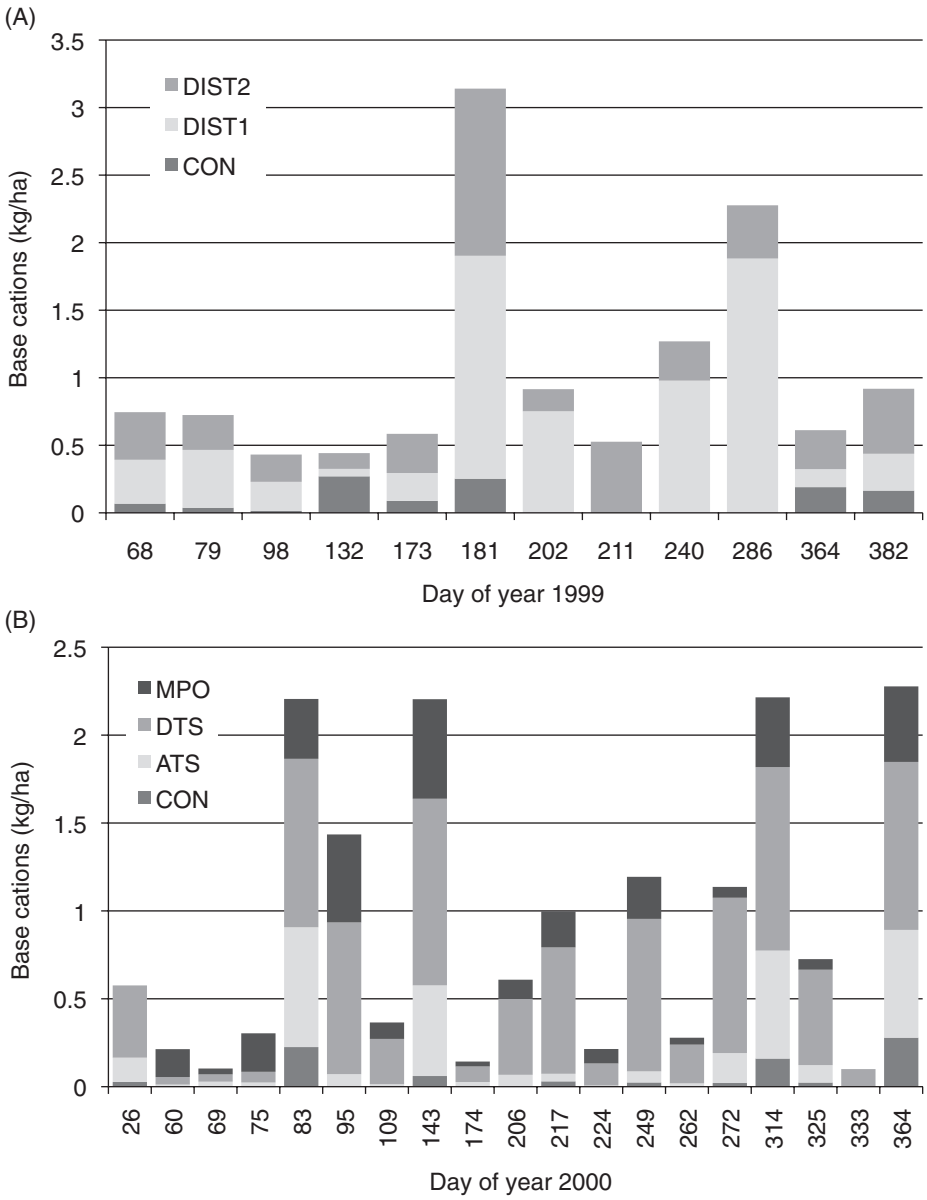

Figure 5—Base cation (kg/ha) mobilization in response to harvest operations (A) and site preparation/planting (B) of a loblolly pine plantation in the Piedmont region of Alabama. (DIST1 = plot, DIST2 = plot, CON = undisturbed forest, MPO = machine planting only, CON = undisturbed forest, DTS = down the slope, ATS = across the slope)

cropping systems similar to the soils of this study have indicated that crop productivity was affected by the degree of erosion and the decline was primarily linked to a lack of phosphorus (P) availability (Stone and others 1985, Thomas and others 1989). Soils of this region are typically deficient (unless the site has a prior history of agricultural production) in available nutrients as typified by a previously uncropped site in Georgia where effective cation exchange capacity (ECEC) values, or nutrient holding capacity, ranged between 2 and 3 cmol(+)/kg (Sumner and others 1986). It should be noted that the quantities of BASECATS displaced by runoff were greater under postharvest conditions compared to site preparation treatments with the exception of DTS. Additionally, NO_3-N quantities, measured only during site preparation treatments, were highest in DTS but overall were very small.

Table 2—Mean accumulations and concentrations of base cations, aluminum, and nitrate by treatment in response to harvest, site preparation, and planting operations in a loblolly pine plantation, Alabama

	Nutrients								Runoff					
Treatment[a]	Ca	Mg	K	Na	Total	Al[b]	BASECATS[b]	NO_3	Ca	Mg	K	Na	Al	NO_3
	kg/ha								*mg/l*					
1999														
CON	0.43	0.11	0.36	0.19	1.09	0.06 b	0.14 b	ND	2.61	0.57	1.87	1.10	0.11	ND
DIST1	3.14	0.82	2.06	0.79	6.81	0.31 b	0.63 a	ND	3.03	0.76	1.87	0.81	0.24	ND
DIST2	1.44	0.41	1.65	1.10	4.60	0.40 a	0.38 ab	ND	1.30	0.37	1.31	0.85	0.94	ND
2000														
CON	0.29	0.10	0.38	0.10	0.67	0.29 b	0.05 b	0.03 b	11.62	1.79	3.04	2.03	0.87	0.53
ATS	1.14	0.37	1.35	0.33	3.19	0.49 b	0.18 bc	0.16 b	6.56	1.20	3.58	1.37	0.84	0.48
DTS	2.45	1.08	4.65	1.49	9.67	6.29 a	0.50 a	0.95 a	3.00	0.78	2.77	0.99	2.63	0.75
MPO	1.01	0.36	1.73	0.47	3.57	2.02 b	0.21 c	0.13 b	5.32	0.94	2.98	1.05	1.13	0.35

Ca = calcium; Mg = magnesium; K = potassium; Na = sodium; Al = aluminum; BASECATS = base cations; NO_3 = nitrate; CON = control; DIST1 and DIST2 = postharvest disturbance; ATS = across the slope; DTS = down the slope; MPO = machine plant only.

[a] Means by treatment year followed by similar letters were not significant at the $\alpha = 0.05$ level.

[b] Means of aluminum and BASECATS for postharvest and postsite preparation/planting was based on $n = 31$ and $n = 71$ observations; $n = 36$ for nitrate-nitrogen results.

SUMMARY

Soil loss and runoff in response to harvesting, site preparation, and planting in a Piedmont site in Alabama exhibited a high degree of variability throughout the study period. Cumulative annual soil loss during postharvest in one treatment (DIST2) slightly exceeded previously reported estimates (~224 kg/ha/year) for harvesting and thinning activities in the Southeast. Site preparation treatments DTS and MPO resulted in substantially higher sediment losses in comparison to ATS and CON. Annual cumulative runoff rates during postharvest and site preparation varied with the highest rates of 200 mm/ha/yr associated with DIST2 and DTS. Nutrient mobilization appeared to be consistent with runoff losses reported in previous studies. The greatest sediment and nutrient mobilization occurred where beds were oriented in the DTS while MPO surprisingly contributed more sediment and nutrient mobilization than expected.

LITERATURE CITED

American Public Health Association. 1995. Standard methods for the examination of water and wastewater. 19th ed. Washington, DC. http://www.standardmethods.org/. [Date accessed: July 22, 2011].

Beasley, R.S.; Granillo, A.B.; Zillmer, V. 1986. Sediment losses from forest management: mechanical vs. chemical site preparation after clearcutting. Journal of Environmental Quality. 15: 413–416.

Blackburn, W.H.; Wood, J.C. 1990. Nutrient export in stormflow following forest harvesting and site preparation in east Texas. Journal of Environmental Quality. 19: 402–408.

Burroughs, E.R.; Luce, C.H.; Phillips, F. 1992. Estimating interrill erodibility of forest soils. Transactions of the American Society of Agricultural Engineers. 35: 1489–1495.

Carter, E.A.; McDonald, T.P.; Torbert, J.L. 2000. Assessment of soil strength variability in a harvested loblolly pine plantation in the Piedmont region of Alabama. New Zealand Journal of Forest Science. 30: 237–249.

Dissmeyer, G.E.; Foster, G.R. 1981. Estimating the cover management factor (C) in the universal soil loss equation for forest conditions. Journal of Soil and Water Conservation. 36: 235–240.

Fowler, J.M.; Heady, E.O. 1981. Suspended sediment production potential on undisturbed forest land. Journal of Soil and Water Conservation. 36: 47–50.

Grace, J.M., III. 2004. Soil erosion following forest operations in the southern Piedmont of central Alabama. Journal of Soil and Water Conservation. 59: 160–166.

Greacen, E.L.; Sands, R. 1980. Compaction of forest soils: a review. Australian Journal of Soil Research. 18: 163–189.

Kinnell, P.I.A.; Cummings, D. 1993. Soil/slope gradient interactions in erosion by rain-impacted flow. Transactions of the American Society of Agricultural Engineers. 36: 381–387.

Kleinman, P.J.A.; Srinivasan, M.S.; Dell, C.J. [and others]. 2006. Role of rainfall intensity and hydrology in nutrient transport via surface runoff. Journal of Environmental Quality. 35: 1248–1259.

McBroom, M.W.; Beasley, R.S.; Chang, M. [and others]. 2008. Storm runoff and sediment losses from forest clearcutting and stand re-establishment with best management practices in east Texas, USA. Hydrological Processes. 22: 1509–1522.

McIntyre, S.C.; Lance, J.C.; Campbell, B.L. [and others]. 1987. Using cesium-137 to estimate soil erosion on a clearcut hillside. Journal of Soil and Water Conservation. 42: 117–120.

Merino, A.; Balboa, M.A.; Rodriguez Soalleiro, R.; Alvarez Gonzalez, J.G. 2005. Nutrient exports under different harvesting regimes in fast-growing forest plantations in southern Europe. Forest Ecology and Management. 207: 325–339.

Meyer, L.D.; Harmon, W.C. 1989. How row-sideslope length and steepness affect sideslope erosion. Transactions of the American Society of Agricultural Engineers. 32: 639–644.

Miller, W.P.; Baharuddin, M.K. 1987. Particle size of interrill eroded sediments from highly weathered soils. Soil Science Society of America Journal. 51: 1610–1615.

Moore, I.D.; Burch, G.J.; Wallbrink, P.J. 1986. Preferential flow and hydraulic conductivity of forest soils. Soil Science Society of America Journal. 50: 876–881.

Ni, S.J.; Zhang, J.H. 2007. Variation of chemical properties as affected by soil erosion on hillslopes and terraces. European Journal of Soil Science. 58: 1285–1292.

Owoputi, L.O.; Stolte, W.J. 1995. Soil detachment in the physically based soil erosion process: a review. Transactions of the American Society of Agricultural Engineers. 38: 1099–1110.

Papiernik, S.K.; Schumacher, T.E.; Lobb, D.A. [and others]. 2009. Soil properties and productivity as affected by topsoil movement within and eroded landform. Soil and Tillage Research. 102: 67–77.

Pye, J.M.; Vitousek, P.M. 1985. Soil and nutrient removals by erosion and windrowing at a Southeastern U.S. Piedmont site. Forest Ecology and Management. 11: 145–155.

Shaw, J.N.; Carter, E.A. 2002. Timber harvesting effects on spatial variability of Southeastern U.S. Piedmont soil properties. Soil Science. 167: 288–302.

Soil Conservation Service. 1981. Soil survey of Lee County, Alabama. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 100 p.

Soltanpour, P.N.; Johnson, G.W.; Workman, S.M. [and others]. 1996. Inductively coupled plasma emission spectroscopy. In: Sparks, D.L., ed. Methods of soil analysis—part 3 chemical methods. Soil Science Society of America Book Series: 5. Madison, WI: American Society of Agronomy; Soil Science Society of America. [Number of pages unknown].

Stein, O.R.; Neibling, W.H.; Logan, T.J. [and others]. 1986. Runoff and soil loss as influenced by tillage and residue cover. Soil Science Society of America Journal. 50: 1527–1531.

Stone, J.R.; Gilliam, J.W.; Cassel, D.K. [and others]. 1985. Effect of erosion and landscape position on the productivity of Piedmont soils. Soil Science Society of America Journal. 49: 987–991.

Sumner, M.E.; Shahandeh, H.; Bouton, J. [and others]. 1986. Amelioration of an acid soil profile through deep liming and surface application of gypsum. Soil Science Society of America Journal. 50: 1254–1258.

Thomas, P.J.; Simpson, T.W.; Baker, J.C. 1989. Erosion effects on productivity of Cecil soils in the Virginia Piedmont. Soil Science Society of America Journal. 53: 928–933.

Van Oost, K.; Govers, G.; De Alba, S. [and others]. 2006. Tillage erosion: a review of controlling factors and implications for soil quality. Progress in Physical Geography. 30: 443–466.

Voorhees, W.B.; Young, R.A.; Lyles, L. 1979. Wheel traffic considerations in erosion research. Transactions of the American Society of Agricultural Engineers. 22: 786–790.

Wendt, R.C.; Alberts, E.E.; Hjelmfelt, A.T., Jr. 1986. Variability of runoff and soil loss from fallow experimental plots. Soil Science Society of America Journal. 50: 730–736.

Yoho, N.S. 1980. Forest management and sediment production in the South—a review. Southern Journal of Applied Forestry. 4: 27–35.

Zhang, X.C.; Norton, D.; Nearing, M.A. 1997. Chemical transfer from soil solution to surface runoff. Water Resources Research. 33: 809–815.

EVALUATION OF ROAD APPROACHES TO FOUR DIFFERENT TYPES OF STREAM CROSSINGS IN THE VIRGINIA PIEDMONT

Matthew B. Carroll, W. Michael Aust, C. Andrew Dolloff, and Robert M. Shaffer[1]

Abstract—Erosion potential was estimated for road approaches during 4 phases of a timber harvesting scheduled for 23 stream crossings in the Virginia Piedmont. The objectives of this study were to: (1) examine four different types of stream crossing structures (steel bridges, pole bridges, standard culverts, and reenforced fords) in order to determine if the type of stream crossing affects erosion potential and (2) evaluate the potential erosion associated with the stream crossing approaches using the Water Erosion Prediction Project (WEPP) for forest roads and the forestry version of the Universal Soil Loss Equation (USLE). An unbalanced replication resulted in six replications of each crossing, except pole bridges (7) and fords (4). Results indicate that any of the stream crossings may be appropriate if located, installed, and maintained properly. However, we found that approaches associated with culverts had the potential for the highest soil loss rates as estimated by both WEPP (46.2 tons per acre per year) and USLE (85.8 tons per acre per year). Both of these models showed a general decrease in the potential for erosion from the during harvest phase to the postroad closure phase.

INTRODUCTION

Stream crossings can produce a number of water-quality pollutants, but sediment is usually the primary concern. Research indicates that roads create more pollution, in the form of sediment, than harvesting activities. Furthermore, stream crossings are the most frequent sources of sediment introduction (Rothwell 1983). Road construction and associated stream crossings are common activities for conventional harvest operations. Sediment produced at stream crossings originates from two primary sources: the stream crossing structure itself and the road approaches to the crossing (Taylor and others 1999). Locating the least steep approaches for stream crossings and choosing good locations are common best management practices (BMP) recommended for minimizing sediment pollution. The potential for water-quality impacts other than sediment also exist at stream crossings. Nutrients attached to sediment particles, which are transported directly to stream systems, may also present additional nonpoint source problems in forested watersheds (Grace 2005).

METHODS

Study Site Description

Stream crossings evaluated in this study were restricted to the Piedmont region. The Piedmont developed due to erosion and has a gentle slope from the mountains to the Coastal Plain (Daniels and others 1973). The interior of the province typically has a gently rolling landscape with moderate relief bounded by steeper, deeper valleys of the modern streams (Daniels and others 1973).

Most study sites were located on private properties that were under contract or land owned by MeadWestvaco or Huber Engineered Woods. Stands harvested ranged from mixed hardwood with white oak (*Quercus alba*) and yellow-poplar (*Liriodendron tulipifera*) to loblolly pine (*Pinus taeda*) plantations. A range of road classes were used to acquire all four types of stream crossings, ranging from skid trails (class IV roads) to permanent haul roads (class II to III roads).

Data Collection

Field visits were conducted during four different phases of the harvesting operation: prereopening/preinstallation, postinstallation/preharvest, during harvest, and postroad closure. Stream crossings were associated with permanent haul roads, temporary haul roads, or skid trails.

Data were collected to predict erosion from both the entrance and exit approach to the stream crossing. Weather information, slope length, slope width, slope percent, slope shape, road management, and soil texture were collected to estimate the approach erosion values with the Water Erosion Prediction Project (WEPP) (Forsyth and others 2005). The Universal Soil Loss Equation (USLE) model was also used to predict erosion from the approaches. Estimated soil erosion is represented by the following equation for USLE:

Estimated soil erosion = A (tons per acre per year) = $RKLSCP$

where
- R = rainfall and runoff index
- K = soil erodibility
- LS = slope length and steepness
- CP = cover-management practice factor for untilled and tilled forest land

The CP factor has several subfactors that influence the estimate such as bare soil, residual binding, soil reconsolidation, canopy, steps, onsite storage, invading vegetation, and high organic matter content (for untilled only) (Dissmeyer and Foster 1984).

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Erosion prediction models were used to estimate the amount of sediment being contributed from road approaches each year on a per-acre basis. WEPP version 2006.5 is a computer-based model published by the U.S. Department of Agriculture, Agricultural Research Service. This model is used to estimate sheet and rill erosion (Forsyth and others 2005). Inputting weather station, slope, road management, and soil texture information in this program allows it to predict erosion (tons per acre per year). The program was run to predict erosion for a 10-year period and obtains an average soil loss value. The USLE manual was the other main source of information to calculate the predicted soil loss. This model is effective for predicting sheet and rill erosion on forest land (Dissmeyer and Foster 1984).

Analyses

Data analysis was performed using the Number Cruncher Statistical System (Hintze 2004). Analysis of variance tests were done at the $\alpha = 0.10$ level. The Tukey-Kramer multiple comparison test was used to show significant differences of the four types of stream crossings at the $\alpha = 0.10$ level.

RESULTS

Evaluation of the erosion rates associated with the approaches to the various stream crossings using the WEPP model indicated no significant differences between the four stream crossing types for the preinstallation phase (*P*-value = 0.201), postinstallation/preharvest phase (*P*-value = 0.89), or postroad closure phase (*P*-value = 0.15). However, the during harvest phase resulted in significant differences between the four stream crossings (table 1) (*P*-value = 0.07). During harvest, culvert crossing approaches resulted in significantly more estimated erosion (46.2 tons per acre per year) than the ford, pole bridge, or steel skidder bridge (18.6, 21.6, and 29.7 tons per acre per year, respectively) (table 1). Higher estimates at the preinstallation phase may be due to preexisting road construction conditions for culvert, ford, and steel bridge stream crossings.

Estimation of the erosion rates associated with the approaches to the studied stream crossings using the USLE model indicated no significant differences between the four stream crossing types for the preinstallation phase (*P*-value = 0.16). However, the preharvest/postinstallation phase (*P*-value = 0.08) and the during harvest phase (*P*-value = 0.0006) resulted in significant differences between the four stream crossings (table 2) Also, the postroad closure phase resulted in significant differences in approaches among the four crossings (table 2) (*P*-value = 0.055). During harvest, approaches associated with culvert crossings resulted in significantly more estimated erosion (85.8 tons per acre per year) than the ford, pole bridge, or steel skidder bridge (23.4, 4.5, and 18.7 tons per acre per year, respectively) (table 2). Culverts, fords, and steel bridges showed a decrease in estimated erosion (50.5, 20.6, and 15.6 tons per acre per year, respectively) at the postroad closure phase. Pole bridge approaches increased from the during harvest phase estimated erosion rate of 4.5 to 10.3 tons per acre per year following road closure. Although significant differences of approaches were realized for the preharvest/postinstallation phase, the Tukey-Kramer multiple comparison test was unable to detect groups due to a limited sample size.

DISCUSSION

Water Erosion Prediction Project Estimates of Approach Erosion

Failure to detect differences in erosion estimates between treatments prior to installation of the crossings indicates that the subsequent treatments were being installed on relatively similar sites, which can be expected due to low disturbance before construction or harvesting activities. Each of the four types of stream crossings had at least one crossing that was installed with preexisting road conditions. Ford crossings had more preexisting crossings and approaches than any other crossing type. These preexisting conditions probably contribute to the higher levels of estimated erosion rates at the prereopening/preinstallation phase (table 1). Field observation and evaluation showed that the WEPP model projected a large amount of annual soil loss on approaches due to cover management practices and slope grade and length, during harvest. Absence of rock or gravel, except within the streamside management zone (SMZ) where the stream crossings were installed, caused higher erosion potential

Table 1—Mean values of the four stream crossing types during each sampling period as predicted by the WEPP model

	Sampling periods			
Stream crossing type	Prereopening/ preinstallation	Postinstallation/ preharvest	During harvest	Postroad closure
	tons per acre per year (tonnes/ha/year)			
Culverts	10.7 (24.0) ns	26.2 (58.7) ns	46.2 (103.5) a[a]	24.4 (54.7) ns
Fords	22.2 (49.7) ns	15.8 (35.4) ns	18.6 (41.7) b	19.9 (44.6) ns
Pole bridges	6.2 (13.9) ns	23.0 (51.5) ns	21.6 (48.4) b	11.8 (26.4) ns
Steel bridges	11.9 (26.7) ns	22.1 (49.5) ns	29.7 (66.5) ab	25.5 (57.1) ns

WEPP = Water Erosion Prediction Project; ns = none significant.

[a] Lower case letters indicate statistical significance at the $\alpha = 0.10$ level.

Table 2—Mean values of the four stream crossing types during each sampling period as predicted by the USLE model

Stream crossing type	Sampling periods			
	Prereopening/ preinstallation	Postinstallation/ preharvest	During harvest	Postroad closure
	tons per acre per year (tonnes/ha/year)			
Culverts	3.8 (8.5) ns	34.4 (77.1) ns	85.8 (192.2) a[a]	50.5 (113.1) a
Fords	2.7 (6.0) ns	9.8 (22.0) ns	23.4 (52.4) b	20.6 (46.1) ab
Pole bridges	0.1 (0.22) ns	1.7 (3.8) ns	4.5 (10.1) b	10.3 (23.1) b
Steel bridges	2.2 (4.9) ns	34.2 (76.6) ns	18.7 (41.9) b	15.6 (34.9) ab

USLE = Universal Soil Loss Equation; ns = none significant.

[a] Lower case letters indicate statistical significance at the $\alpha = 0.10$ level.

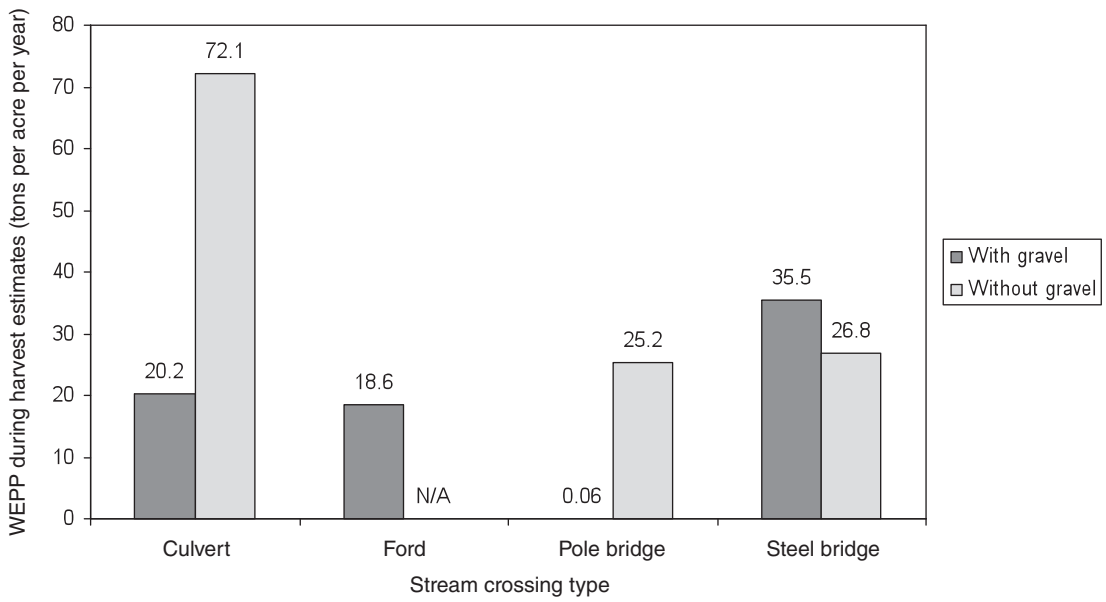

Figure 1—Water Erosion Prediction Project (WEPP) during harvest estimated erosion rates showing the differences in erosion rates for gravel/rock application to road approaches for each of the four types of stream crossings.

for some crossings (fig. 1). After harvest activities included implementing BMPs and reestablishing vegetation. Most stream crossing approaches decreased in WEPP erosion potential from the during harvest phase to the postroad closure phase with the exception of the ford stream crossing, which slightly increased (table 1).

Universal Soil Loss Equation Estimates of Approach Erosion

All stream crossings showed low-erosion potential, <4 tons per acre per year (9 mt/ha/year) (table 2) at the preinstallation/ prereopening phase. Postinstallation/preharvest mean erosion estimate values showed a *P*-value of 0.079 which revealed significance among crossing types. However, data recorded for this phase of harvest were limited due to factors such as immediate use of stream crossings and preexisting conditions of previously used crossings. USLE mean erosion estimates displayed a significant difference among stream crossing approaches during harvest. Approaches to stream crossing erosion means decreased from the during harvest phase to the after harvest phase with the exception of road approaches associated with pole bridge crossings (table 2). Possible explanations of this increase for pole bridge approaches from 4.5 tons per acre per year (10.1 mt/ha/year) during harvest to 10.3 tons per acre per year (23.1 mt/ha/year) after harvest are increases in bare ground and removal of natural vegetation. Often logging contractors will remove “rub” trees which are commonly used to change the direction of a skidder’s load of timber to minimize stream channel contact. This removal of trees adjacent to the approach decreases the amount of cover.

CONCLUSIONS

The evaluation of erosion potential from road approaches leading to 23 stream crossings throughout the harvest process allows the following conclusions to be drawn:

- Approaches associated with culvert stream crossings provide the highest potential for soil erosion of the four types of stream crossings studied as estimated by both WEPP—road model—and USLE—forestry version.
- Implementing BMP practices to reduce bare soil, increase the residual natural vegetation, and minimize slope length can help in maintaining low potentials for estimated erosion on an annual basis.

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LITERATURE CITED

Daniels, R.B.; Allen, B.L.; Bailey, H.H.; Beinroth, F.H. 1973. Physiography. In: Buel, S.W., ed. Soils of the Southern States and Puerto Rico. South. Coop. Ser. Bull. 174. [Place of publication unknown]: U.S. Department of Agriculture: 3–23.

Dissmeyer, George E.; Foster, George R. 1984. A guide for predicting sheet and rill erosion on forest land. Tech. Publ. R8-TP 6. [Atlanta]: U.S. Department of Agriculture Forest Service, Southern Region. 40 p.

Forsyth, A.R.; Bubb, K.A.; Cox, M.E. 2005. Runoff, sediment loss and water quality from forest roads in a southeast Queensland Coastal Plain *Pinus* plantation. Forest Ecology and Management. 221: 194–206.

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

Hintze, J. 2004. NCSS 2004. Kaysville, UT: NCSS, Inc. http://www.ncss.com.

Rothwell, R.L. 1983. Erosion and sediment production at mid-stream crossings. Forestry Chronicle. 23: 62–66.

Taylor, Steven E.; Rummer, Robert B.; Yoo, Kyung H. [and others]. 1999. What we know—and don't know—about water quality at stream crossings. Journal of Forestry. 97(8): 12–17.

INFLUENCE OF FOREST ROADS STANDARDS AND NETWORKS ON WATER YIELD AS PREDICTED BY THE DISTRIBUTED HYDROLOGY-SOIL-VEGETATION MODEL

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Abstract—Throughout the country, foresters are continually looking at the effects of logging and forest roads on stream discharge and overall stream health. In the Pacific Northwest, a distributed hydrology-soil-vegetation model (DHSVM) has been used to predict the effects of logging on peak discharge in mountainous regions. DHSVM uses elevation, meteorological, vegetation, and soil data to model the hydrology of the catchment explicitly on a grid cell by grid cell scale. The model is unique in its ability to consider the impacts of road networks on catchment hydrology due to the addition of a road and channel network algorithm. This is of critical importance because it has long been recognized that forest roads can have very large impacts on water yields and water quality. The primary objectives of this study are to determine whether or not DHSVM can be applied to the gentler slopes of the Appalachian Mountains and, if so, determine which types of roads and road networks have the smallest effect on stream discharge. Calibration of the model will be done using historical data collected from the Coweeta Long-Term Ecological Research Station in the Blue Ridge Mountains of North Carolina. Forest road parameters that will be considered in this study include road density and road standards. This type of information will be useful to watershed managers and watershed planners for minimizing the impacts of forest roads.

INTRODUCTION

Across the United States, extensive changes in land use have been occurring. Between 1973 and 2000, approximately 800 km^2 of forest and agriculture land were developed or mechanically disturbed in the Blue Ridge Mountains (Taylor and others 2007). These massive changes in landcover have generated public anxieties over potential environmental impacts. Extractive resources practices, such as forestry and mining, have been at the forefront of these concerns. In July 2001, extensive flooding occurred in southern West Virginia and southwestern Virginia. National Weather Service stations recorded maximum rainfall intensities of 50 mm/hour, and damages were estimated to be around $150 million (Eisenbies and others 2007). As a result of the devastation, West Virginia Governor Bob Wise created a Flood Investigation Advisory Committee that was designed to evaluate the impacts of logging and mining on flooding. In 2002 the Flood Advisory Technical Taskforce recommendations were released. The comprehensive report detailed guidelines for mining and logging activities and aimed at reducing their environmental impact.

Because of such highly publicized flood events, there is often a public misconception about the effects that forest harvesting can have on flooding (McCutcheon 2006). Following harvest, stream discharge often increases slightly due to decreases in evapotranspiration (Cornish and Vertessy 2001). However, many studies have found that discharge returns to preharvest conditions within 5 years of logging (Hewlett and Helvey 1970, Hornbeck and others 1970, Swank and others 2001).

While forest systems will regenerate after a harvest and often return to preexisting conditions, logging can create some permanent impacts on the ecosystem, mainly in the form of road networks. Forest roads can affect the natural movement and extent of runoff by redistributing the water through the road network. The nonvegetated corridors can act like stream channels, intercepting overland flow and rerouting it downslope. The compacted surfaces of forested roads can also reduce infiltration, thus impacting the health and vitality of the ecosystem. In a particular catchment, a road segment may act as a barrier, corridor, sink, or source for both water and sediments (Jones and others 2000). All of these functions can alter the stream hydrograph, both in quantity and in timing.

Many studies have been conducted that attempt to determine the effects of forest harvesting and road construction on stream discharge. However, it can be difficult to separate the two effects because harvesting and road construction are often done simultaneously. Bowling and Lettenmaier (2002) used a distributed hydrology-soil-vegetation model (DHSVM) to simulate the effects of forest roads on streamflow in two catchments in western Washington. The drainage areas ranged from 2.3 to 2.8 km^2 and the road lengths were 10.7 and 11.4 km, respectively. The authors found that over the 11-year simulation phase, roads networks increased the 10-year return period flood by 8 to 10 percent. With mature vegetation, it was found that forest roads could increase 10-year return peak flow rates by as much as 22 percent (Bowling and Lettenmaier 2002). The authors discovered that roads redistribute water throughout the basin, resulting in drier zones directly beneath the road (due to compaction) and saturated zones in areas surrounding culvert drainage points (Bowling and Lettenmaier 2002).

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There are many environmental concerns over the construction of forest roads, including forest fragmentation, increased sedimentation and thus decreased water and stream quality, increased fire hazards, and invasive species dispersal. As of 2007, there were approximately 386,000 miles of roads located within the National Forest System. The 2008 fiscal year budget for the U.S. Forest Service appropriates over $225 million to the management of the road network; this is an 8-percent increase from the 2007 budget (U.S. Department of Agriculture 2007). Due to their increasing importance as well as their mounting scrutiny, it is critical for forest roads to be properly designed, installed, and maintained.

The purpose of this study is to assess the capability of a DHSVM to predict the impacts of forest roads and forest road networks on peak stream discharge. The specific objectives of this research are to: (1) determine whether or not DHSVM can be used in the gentler slopes and deeper soil terrains of the Appalachian Mountains and (2) assess which types of road networks have the most impact on stream discharge.

BACKGROUND

Road Standards

Forest roads are often categorized into four different road classes, which elucidate the overall quality of the road—the lower the class number, the higher the road quality. Some design specifications for classifying a forest road include the cut-and-fill slope ratios, road grade, number and angle of switchbacks, surfacing, water control features, and number and type of stream crossings (Walbridge 1997). An improperly designed road can negatively affect drainage and water control, water quality, traffic, and erosion. In comparison, an accurately placed and planned road can increase the access to and the value of property, decrease harvesting days and reduce costs, comply with Federal regulations, and increase the emergency access.

The Distributed Hydrology-Soil-Vegetation Model

DHSVM is a physically based model that represents watershed processes at the scale of a digital elevation model (Wigmosta and others 2002). The use of distributed models is becoming increasingly important in hydrologic predictions because they allow the user to utilize available spatial data for both input and testing. DHSVM has been used to analyze the effects of land use change, harvesting, and road networks on streamflow in forested, mountainous environments throughout the world (Bowling and Lettenmaier 2002, Bowling and others 2000, Cuo and others 2006, Doten and Lettenmaier 2004, Doten and others 2006, VanShaar and others 2002). It has also been used in hydrologic modeling (Haddeland and Lettenmaier 1995, Kenward and Lettenmaier 1997, Westrick and others 2002, Wigmosta and Lettenmaier 1999, Wigmosta and others 1995), and to look at the interactions between climate and climate change and hydrology (Arola and Lettenmaier 1996, Leung and Wigmosta 1999, Wigmosta and others 1995). DHSVM is described in complete detail in Wigmosta and others (1994) and Wigmosta and others (2002).

The DHSVM is complex and requires a number of spatial data inputs as well as defined parameters. Due to this fact, calibration of the model for parameters that cannot be measured is necessary. Bowling and Lettenmaier (2002) used a calibration period of 3 years for their study on the effect of forest road systems on forested catchments in the Pacific Northwest. When calibrating the model for discharge, the authors had to adjust lateral hydraulic conductivity (LAI), deep layer soil depth, height of road cuts, and decrease in LAI. In general, the model underpredicted the base flows and overpredicted storm peaks (Bowling and Lettenmaier 2002).

Cuo and others (2006) used DHSVM to simulate the effects of road networks on hydrological processes in northern Thailand. During calibration, the authors found that the model adequately replicated soil moisture and depth but only accurately simulated streamflow for 2 of the 3 calibration years. This was attributed to year to year changes in land cover that were not reproduced in the model (Cuo and others 2006). In an early presentation of DHSVM, Wigmosta and others (1994) present model calibration data for a forested basin in Montana. For discharge, the model had a daily simulation root mean square area of 1.2 mm and a R^2 of 0.95. The model slightly overpredicted hydrograph recession for the growing season and undersimulated low flow during the dormant season (Wigmosta and others 1994).

METHODS

Site Description

Model calibration will be conducted using data from the Coweeta Long-Term Ecological Research (LTER) Station in Macon County, NC (35°03' N, 83°25' W). The Coweeta River Basin is located in the Nantahala National Forest in western North Carolina, which is in the Blue Ridge physiographic province (fig. 1). The Coweeta LTER was established in 1932 for the purpose of studying streamflow and erosion in an ecological context (Douglass and Hoover 1988). The LTER consists of two adjacent bowl-shaped basins—the Coweeta Basin and the Dryman Fork Basin. This study will be concentrated on the Coweeta Basin. Together, the basins are comprised of over 50 watersheds, ranging from 3 to 760 ha, with the total LTER measuring approximately 2185 ha. The region is mountainous and elevations range from 675 to 1592 m at the top of Albert Mountain. Sideslopes depend on the mountain but generally range from 50 to 60 percent (Swank and Crossley 1988). Based on the Thornthwaite

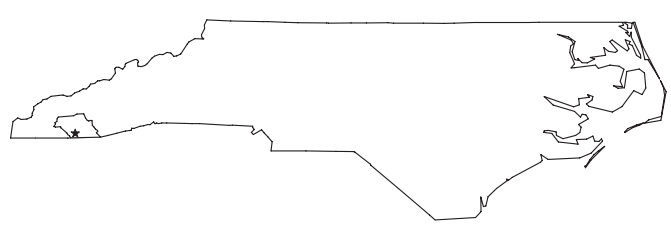

Figure 1—The Coweeta Long-Term Ecological Research Station is located in Macon County, NC.

original scheme, Coweeta's climate is classified as wet and mesothermal with adequate rainfall. Precipitation is most abundant during the winter months; a yearly average measured at one climate station was 1820 mm. Average temperatures range from winter lows of –4 °C to summer highs of 23 °C (Swift and others 1988). A soils map of the area was created by the Soil Conservation Service (now the Natural Resources Conservation Service) in 1985. Major soil series include Inceptisols, which are characterized by little profile development, and Ultisols, which are older and more weathered (Swank and Crossley 1988). Forest vegetation in the Appalachians can be very diverse, but major overstory species at Coweeta include *Acer rubrum*, *Nyssa sylvatica*, *Carya* spp., *Quercus prinus*, and *Oxydendrum arboreum*. Major understory influences include *Rhododendron* spp. and *Kalmia* spp. (Bolstad and others 1997).

Calibration and Validation

The DHSVM has been used to analyze streamflow in forested, mountainous terrain, most commonly in the Pacific Northwest (Bowling and Lettenmaier 2002, Doten and Lettenmaier 2004, Doten and others 2006, Leung and Wigmosta 1999, Wigmosta and Lettenmaier 1999, Wigmosta and others 1994) and Canada (Wigmosta and Perkins 2001; Whitaker and others 2002, 2003). In the Cascade Range in Washington and Oregon, elevations can range from sea level to over 2000 m, and slopes can reach upwards of 100 percent. Here in the Southern Appalachian Mountains, elevations are much lower and slopes have gentler gradients. In the Coweeta watershed, maximum elevation change is approximately 915 m, and sideslope slopes are 50 to 60 percent (Swift and others 1988). In order to determine whether or not DHSVM is applicable to gentler terrains with deeper soil systems, calibration and validation of the model are necessary. The model will be standardized using observed stream discharge data from the Coweeta LTER. Availability of data and location of weirs restricts the study to the northern portion of the Coweeta Basin (fig. 2).

Calibration of the model will be run for the 2001 and 2002 water years or from October 1, 2000, to September 30, 2002. Modeled stream discharge data will be contrasted with observed data to ensure that the model reaches a steady state before all further runs. A correlation coefficient (R^2) of at least 0.8 will be needed to consider the calibration successful. A validation period will occur from October 1, 2002, to September 30, 2007, to guarantee the quality of the parameters and the model data.

Road Density Experiment

Determining the impact of road density on stream discharge can be difficult because so many road factors can have an effect on watershed processes. Such road impact factors include the spatial location of the road with relation to streams, the gradient or slope of the road, the surfacing material of the road, road design features (such as insloped, outsloped, and crowned roads), the water control features of the road, number of stream crossings, and length of the road. Bernard (2006) developed an approach to determine the impact of road networks on sedimentation potential by

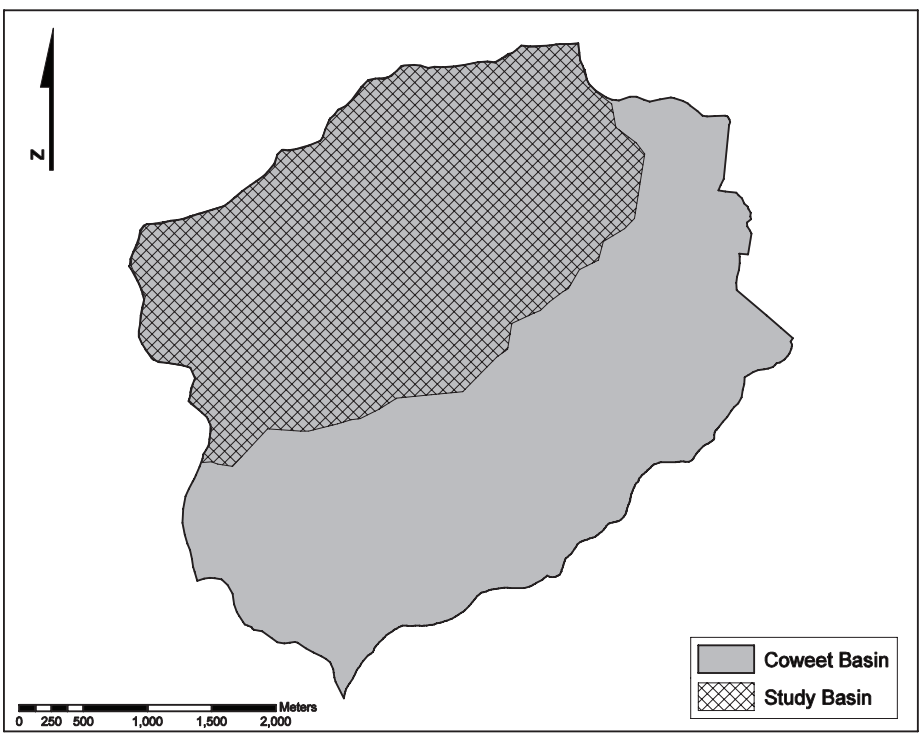

Figure 2—The study site will be confined to the northern portion of the Coweeta Basin.

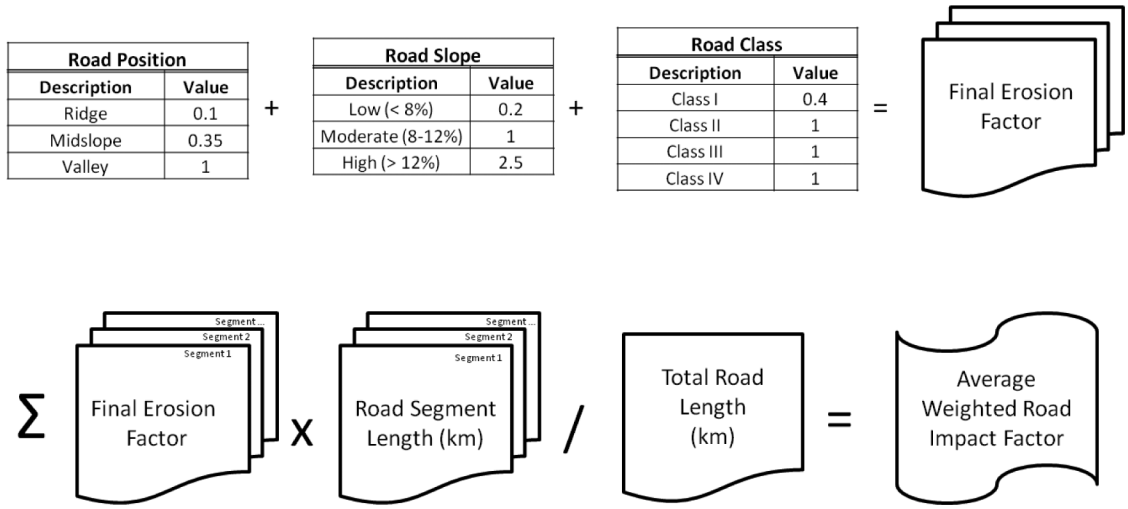

Figure 3—Flowchart of final erosion factor and final watershed road impact factor. Adapted from Bernard (2006).

amalgamating road position, slope, and class into a final erosion factor. A final watershed road impact factor is found by integrating road segment length into the model (Bernard 2006). For the purposes of this project, an average final erosion factor will be used. A flowchart of the road impact factor is depicted in figure 3. It is assumed that the road class factor encompasses other water control features on the road, as well as the design and surface material of the road.

A road-density experiment will be conducted on the northern portion of the Coweeta Basin. The experiment on road density effects on water discharge will be a completely randomized split-plot design. Instead of using whole plots, various road layouts will be designed. The two treatment factors are road density and the erosion factor (EF). The road-density experiment will include three treatments, plus the control, which are explained in table 1. For each road density, two road networks will be designed. Within the specified density and layout, the average final EFs will be varied by modifying the road class. All model variables other than road density and EF will remain as they were set during the validation run. The road effects on streamflow will be analyzed and variables of interest include intensity and timing of the storm hydrograph. Data will be analyzed using SAS 9.1. A general analysis of variance will be run using PROC GLM.

Table 1—Treatment descriptions for road-density experiment

Treatment	Road density	Replications
	------- *m/ha* -------	
Control	40	3
1	0	3
2	20	3
3	60	3

CONCLUSIONS

Flooding has and will continue to be an important consideration when evaluating the effects of changes in land use. For forested watersheds, forest roads have the potential to modify flooding potential in both positive and negative ways (Eisenbies and others 2007). Our study was designed to assess the DHSVM's utility for evaluating the influence of road standards and network density on potential flooding. The model appears to include variables that could reasonably be expected to relate to water movement associated with forest roads, but it requires data that are generally only available from research watersheds. Also, the model is relatively nonuser-friendly, so our calibration attempts are incomplete. We hope to remedy the validation problems and develop a more user-friendly interface as our next steps.

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LITERATURE CITED

Arola, A.; Lettenmaier, D.P. 1996. Effects of subgrid spatial heterogeneity on GCM-scale land surface energy and moisture fluxes. American Meteorological Society. 9: 1339–1349.

Bernard, A.M. 2006. Geospatial modeling of forest road networks and their effects on stream macroinvertebrate communities. Blacksburg, VA: Virginia Polytechnic Institute and State University. 110 p. M.S. thesis.

Bolstad, P.V.; Swank, W.T.; Vose, J.M. 1997. Predicting Southern Appalachian overstory vegetation with digital terrain data. Landscape Ecology. 13: 271–283.

Bowling, L.C.; Lettenmaier, D.P. 2002. Evaluation of the effects of forest roads on streamflow in Hard and Ware Creeks, Washington. Water Resour. Ser. Tech. Rep. 155. Seattle: University of Washington, Department of Civil and Environmental Engineering. 202 p.

Bowling, L.C.; Storck, P.; Lettenmaier, D.P. 2000. Hydrologic effects of logging in western Washington, United States. Water Resources Research. 36(11): 3223–3240.

Cornish, P.M.; Vertessy, R.A. 2001. Forest age-induced changes in evapotranspiration and water yield in a eucalypt forest. Journal of Hydrology. 242: 43–63.

Cuo, L.; Giambelluca, T.W.; Ziegler, A.D.; Nullet, M.A. 2006. Use of distributed hydrology soil vegetation model to study road effects on hydrologic processes in Pang Khum Experimental Watershed, northern Thailand. Forest Ecology and Management. 224: 81–94.

Doten, C.O.; Bowling, L.C.; Lanini, J.S. [and others]. 2006. A spatially distributed model for the dynamic prediction of sediment erosion and transport in mountainous forested watersheds. Water Resources Research. 42: 15.

Doten, C.O.; Lettenmaier, D.P. 2004. Prediction of sediment erosion and transport with the distributed hydrology—oil-vegetation model. Water Resour. Ser. Tech. Rep. 178. Seattle: University of Washington, Department of Civil and Environmental Engineering. 70 p.

Douglass, J.E.; Hoover, M.D. 1988. History of Coweeta. In: Swank, W.T.; Crossley, D.A., eds. Forest hydrology and ecology at Coweeta. New York: Springer-Verlag: 17–31.

Eisenbies, M.H.; Adams, M.B.; Aust, W.M.; Burger, J.A. 2007. Bibliography of forest water yields, flooding issues, and the hydrologic modeling of extreme flood events. Gen. Tech. Rep. NRS-8. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 44 p.

Haddeland, I.; Lettenmaier, D.P. 1995. Hydrologic modeling of boreal forest ecosystems. Water Resour. Ser. Tech. Rep. 143. Seattle: University of Washington, Department of Civil and Environmental Engineering. 138 p.

Hewlett, J.D.; Helvey, J.D. 1970. Effects of forest clear-felling on the storm hydrograph. Water Resources Research. 6(3): 768–782.

Hornbeck, J.W.; Pierce, R.S.; Federer, C.A. 1970. Streamflow changes after forest clearing in New England. Water Resources Research. 6(4): 1124–1132.

Jones, J.A.; Swanson, F.J.; Wemple, B.C.; Snyder, K.U. 2000. Effects of forest roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology. 14(1): 76–85.

Kenward, T.; Lettenmaier, D.P. 1997. Assessment of required accuracy of digital elevation data for hydrologic modeling. Water Resour. Ser. Tech. Rep. 153. Seattle: University of Washington, Department of Civil and Environmental Engineering. 131 p.

Leung, L.R.; Wigmosta, M.S. 1999. Potential climate change impacts on mountain watersheds in the Pacific Northwest. 35(6): 1463–1471.

McCutcheon, S.C. 2006. Hydrology of Eastern U.S. forested mountain watersheds: rigorous testing of the curve number method for flood analysis. Testimony to the West Virginia Forest Management Review Commission. Athens, GA: University of Georgia, School of Forest Resources. 11 p.

Swank, W.T.; Crossley, D.A. 1988. Introduction and site description. In: Swank, W.T.; Crossley, D.A., eds. Forest hydrology and ecology at Coweeta. New York: Springer-Verlag: 3–16.

Swank, W.T.; Vose, J.M.; Elliott, K.J. 2001. Long-term hydrologic and water quality responses following commercial clearcutting of mixed hardwoods on a Southern Appalachian catchment. Forest Ecology and Management. 143: 163–178.

Swift, L.W.; Cunningham, G.B.; Douglass, J.E. 1988. Climatology and hydrology. In: Swank, W.T.; Crossley, D.A., eds. Forest hydrology and ecology at Coweeta. New York: Springer-Verlag: 35–55.

Taylor, J.L.; Kurtz, J.; Napton, D.E. 2007. Status and trends of Eastern United States land cover. Washington, DC: U.S. Department of the Interior. [Number of pages unknown].

U.S. Department of Agriculture Forest Service. 2007. Fiscal year 2008: President's budget. [Washington, DC]. 60 p.

VanShaar, J.R.; Haddeland, I.; Lettenmaier, D.P. 2002. Effects of land-cover changes on the hydrological response of interior Columbia River basin forested catchments. Hydrological Processes. 16: 2499–2520.

Walbridge, T.A. 1997. The location of forest roads. Blacksburg, VA: Virginia Polytechnic Institute and State University. 91 p.

Westrick, K.J.; Sorck, P.; Mass, C.F. 2002. Description and evaluation of a hydrometeorological forecast system for mountainous watersheds. Weather and Forecasting. 17: 250–262.

Whitaker, A.; Alila, Y.; Beckers, J. 2002. Evaluating peak flow sensitivity to clear-cutting in different elevation bands of a snowmelt-dominated mountainous catchment. Water Resources Research. 38(9): 1172–1189.

Whitaker, A.; Alila, Y.; Beckers, J.; Toews, D. 2003. Application of the distributed hydrology soil vegetation model to Redfish Creek, British Columbia: model evaluation using internal catchment data. Hydrological Processes. 17: 199–224.

Wigmosta, M.S.; Lettenmaier, D.P. 1999. A comparison of simplified methods for routing topographically driven subsurface flow. Water Resources Research. 35(1): 255–264.

Wigmosta, M.S.; Leung, L.R.; Rykiel, E.J. 1995. Regional modeling of climate-terrestrial ecosystem interactions. Journal of Biogeography. 1: 453–465.

Wigmosta, M.S.; Nijssen, B.; Storck, P. 2002. The distributed hydrology soil vegetation model. In: Singh, V.P.; Frevert, D., eds. Mathematical models of small watershed hydrology and applications. Highlands Ranch, CO: [Publisher unknown]: 7–42.

Wigmosta, M.S.; Perkins, W.A. 2001. Simulating the effects of forest roads on watershed hydrology. In: Wigmosta, M.S.; Burges, S.J., comps., eds. Land use and watersheds: human influence on hydrology and geomorphology in urban and forest areas. Washington, DC: American Geophysical Union: 127–143.

Wigmosta, M.S.; Vail, L.W.; Lettenmaier, D.P. 1994. A distributed hydrology-soil-vegetation model for complex terrain. Water Resources Research. 30(6): 1665–1679.

THE EFFECTIVENESS OF STREAMSIDE MANAGEMENT ZONES IN CONTROLLING NUTRIENT FLUXES FOLLOWING AN INDUSTRIAL FERTILIZER APPLICATION

Joseph M. Secoges, W. Michael Aust, and John R. Seiler[1]

Abstract—Many State best management practice programs recommend streamside management zone (SMZ) widths based on limited or inadequate data with regard to nutrient fluxes from silvicultural activities. Diammonium phosphate and urea were applied to subwatersheds of 2- to 3-year-old loblolly pines (*Pinus taeda*) upslope from 12 SMZ study areas in Buckingham County, VA. Three replications of four SMZ treatments (30.5 m, 15.2 m nonthinned, 15.2 m thinned, and 7.6 m) were studied using ionic exchange membranes. Our hypothesis is that nitrogen and phosphorous levels will decrease in a forested SMZ setting as distance from the harvest boundary to the creek increases. Furthermore, we hypothesize that wider, unthinned SMZs are more likely to prevent nutrients from reaching the creek than narrower and/or thinned SMZs. Preliminary results indicate stream water quality is unaffected by fertilization at all SMZ width treatments. Nutrient movement in the upper soil and litter layer through the various SMZ widths will be discussed.

INTRODUCTION

Public concern over the health of the Nation's waters led to the development of the Federal Water Pollution Control Act of 1972 (P.L. 92.500) and subsequent amendments, which are commonly known as the Clean Water Act (Ice and others 1997). The Clean Water Act "set water quality standards for all contaminants in surface waters" (U.S. Environmental Protection Agency 2008). In order to comply with guidelines established by the Clean Water Act, the State of Virginia established best management practices (BMP) for forestry operations (Virginia Department of Forestry 2002). BMP guidelines are environmentally and economically significant because they influence the potential management on 6.2 million ha (61 percent of the total land base) of potentially commercial forest land in Virginia (Virginia Department of Forestry 2002). Virginia landowners implement BMPs voluntarily except within Chesapeake Bay Preservation Area where BMP usage is required. Furthermore, all silvicultural practices must not produce water toxicity levels higher than national standards or cause increased sedimentation as regulated by the Virginia Silvicultural Water Quality Law (Virginia Department of Forestry 2002).

Streamside management zones (SMZ) are commonly used BMPs for preventing water-quality degradation from forest silviculture operations (Aust and Blinn 2004). Streamside forests help prevent excess sediment and nutrients from reaching the stream, protect streams from thermal pollution, correct negative aquatic effects of pesticides, and help generate food sources that promote aquatic productivity and diversity (Welsch 1996). Additional principle functions of riparian areas are to stabilize streambanks, provide a source of spawning gravel, moderate riparian microclimates, and provide wildlife habitat (O'Laughlin and Belt 1995). Walbridge (1993) noted that forested wetlands provide eight biogeochemical functions—sediment deposition, denitrification, sulfate reduction, phosphorous sorption, nutrient uptake, decomposition of waste organics, sorption of heavy metals, and retention of toxics—that improve water quality and two biogeochemical functions—carbon storage and methane production—that influence global atmospheric changes. During the first year following a harvest in Mississippi, Keim and Schoenholtz (1999) found that streams in logged watersheds without SMZ implementation had approximately three times the sediment concentration as found in nonharvested watersheds and concluded that SMZs are most effective for controlling sedimentation when the forest floor is undisturbed. Adequate buffer width depends on the condition of the buffer, e.g., amount of vegetation and soil disturbance, the relative functional value of the water body, e.g., disturbance regime and plant origin, and the impact potential of adjacent land, e.g., park land vs. residencies or farms (Castelle and others 1994).

Various widths of SMZs have been recommended and implemented based on the level of protection desired for a particular type of water body and the percent slope of adjacent lands (Virginia Department of Forestry 2002). However, recommended widths are arbitrary guidelines most often determined by politics and established with little or no scientific basis with regard to effectiveness of various widths in controlling targeted pollutants (Castelle and others 1994). Additionally, science-based width recommendations were established mostly for perceived adequate sediment control rather than for nutrients. A 15.2-m wide SMZ is the most commonly utilized for a typical upland Piedmont forested stream.

OBJECTIVES

The primary goal of this study was to determine SMZ width necessary for prevention of nitrogen (N) and phosphorous (P) from reaching a creek following typical diammonium

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phosphate (DAP) and urea (UREA) application on a young loblolly pine (*Pinus taeda*) stand. Previous studies indicate that forested riparian areas act as nutrient sinks (Fox and others 2007, Lowrance and others 1984, O'Laughlin and Belt 1995). Jacobs and Gilliam (1985) found that Coastal Plain buffer strips of <15.2 m significantly reduced nitrate levels in agricultural drainage water before it reached the stream. However, the typical SMZ width needed to prevent increased N and P levels in upper Piedmont streams of a forested setting is unclear. This project is aimed at identifying appropriate SMZ width by examining water movement above the soil surface, within the primary rooting zone, and through deeper subsurface flow. Establishing the transportation patterns of N and P through a SMZ could help forest land managers appropriately designate future harvest boundaries. This study examined three SMZ widths and differences between a nonmanaged and a thinned 15.2-m (50-foot) SMZ, currently the most common SMZ width in Virginia.

METHODS

Twelve study subwatersheds were located in Buckingham County (37°33'00" N, 78°33'30" W) in the upper Piedmont of central Virginia. The subwatersheds are part of a larger watershed research project described by Lakel and others (2006a). The upper Piedmont has rolling terrain and typical elevations ranging from 60 to 460 m above mean sea level (Lakel and others 2006a). Typical land management actions coincide with industrial forest operations and include loblolly pine plantations, clearcutting, ground skidding, fire breaks, chemical application, and controlled fire. Annual precipitation for this region is 107 cm. Average winter (December to February) temperature is 3.3 °C while average growing season (April to September) temperature is approximately 21 °C (Wiseman and Seiler 2004). Soils are generally highly eroded due to previous abusive agricultural activities. Shallow Ultisols with thin surface Ap horizons over subsurface argillic horizons are typical. The Ap horizon is usually low in organic matter and eluvial E horizons are generally slight or absent. Soil textures are frequently gravelly loam to gravelly sandy loam over 1 to 1 clay subsoils (Wiseman and Seiler 2004). Riparian areas on the test sites often have higher levels of organic matter, sand, and coarse fragments than the harvested treatment area.

All sites were on MeadWestvaco properties and were clearcut harvested between summer 2003 and spring 2004 using standard equipment and systems (ground-based harvesting with rubber-tired feller bunchers and skidder) and then site-prep burned by fall of 2004. Within each of the 12 study watersheds (first-order intermittent and perennial streams), a smaller contributing subwatershed (zero order, ephemeral drain) was selected in the clearcut area for the study of fertilizer nutrient movement through various SMZ widths. These subwatersheds ranged from 0.2 to 1.4 ha in size. Treatments were arranged in a completely random design with three replications of the following four treatment widths described by Lakel and others (2006b):

1. 7.6-m (25-foot) wide SMZ (with varying degrees of SMZ harvest but no management)
2. 15.2-m (50-foot) wide SMZ with no SMZ harvest
3. 15.2-m (50-foot) wide SMZ with 50 percent SMZ harvest
4. 30.5-m (100-foot) wide SMZ with no SMZ harvest

DAP and UREA fertilizers were applied to the 12 subwatersheds at common industrial rates of 140.3 and 250.4 kg/ha (125 and 223 pounds per acre), respectively, by ATV and by hand. Application yielded 28.1 kg/ha (25 pounds per acre) elemental P and 140.3 kg/ha (125 pounds per acre) elemental N. DAP and UREA are the primary fertilizer compounds used for forestry fertilization. Slopes, soils, and vegetation were relatively constant among treatments; therefore, it was hypothesized that differences in SMZ width and harvest level would impact the amount of fertilizer nutrient capable of moving from the ephemeral watersheds into the larger study streams.

Ionic Exchange Membranes

Gaskin and others (1989) state that lateral near-surface flow is an important path of nutrient movement in surface horizons. Fluxes of SO_4, Cl, NO_3-N, K, Ca, Mg, and H are greatest in the B/A horizon and decrease with depth. The greatest lateral flow is higher in the soil profile under dryer conditions (Gaskin and others 1989). Denitrification potential is highest in the top 2 cm of surface soil and occurs at higher rates when the soil has higher levels of organic matter (Ambus and Lowrance 1991). Historically, cation and anion exchange resin bags have been used to quantify nutrient movement in upper soil horizons. However, ion-exchange membranes (Ionics, Inc., Watertown, MA), also known as IEMs, are becoming more popular due to their many advantages over resin bags (Elliot 2006). In contrast to the resin bag form, the membrane form offers advantages because their flat structure ensures a constant surface area and better contact with the soil (Huang and Schoenau 1996). Diffusion problems are reduced because the two-dimensional structure ensures more surface area will be in contact with the soil which undergoes minimal disturbance during installation. Furthermore, the IEM is physically and chemically durable and has a high correlation with soil solution P at low-solution concentrations (Cooperband and Logan 1994). IEMs were installed horizontally in the A/B soil horizon (1 to 10 cm deep) as well as the litter layer/A horizon fringe. The litter layer/A horizon interface was studied because a major source and sink of plant nutrients is the litter layer. In regards to N, the largest proportion of water soluble N and supply rates of organic N can be found in the lowest, most decomposed horizon of the litter layer.

IEMs (6.35 cm by 6.35 cm) were installed symmetrically across the width of each SMZ. A set of four membranes included a cation and anion membrane which were inserted in a slit under the O horizon in addition to a cation and anion membrane which were inserted in the A or top of the B horizon (1 to 10 cm below the soil surface) with a gardening trowel. Membranes were placed as flat as possible to minimize the chance for preferential flow. Before the membranes were placed in the field they were rinsed with deionized water to remove excess NaCl solution. Initially

membranes were removed/reinstalled approximately every 2 weeks, but the timeline was altered based on the membranes saturation potential and movement of fertilizer nutrients across the membrane's surface. The number of membranes installed in each treatment was as follows:

Treatment 1 (7.6 m) – 12 membranes per replication
Treatment 2 (15.2 m) – 12 membranes per replication
Treatment 3 (15.2 m) – 20 membranes per replication
Treatment 4 (30.5 m) – 24 membranes per replication

Data were analyzed as a completely randomized design with three replications of four treatments. When treatment differences (alpha = 0.1) were found treatment differences were separated using Tukey's mean separation test at a 0.1 alpha level.

RESULTS AND DISCUSSION

Cation and anion membranes were utilized at various time intervals in the field for over a year to serve as an index for ammonium and nitrate movement at the litter layer-soil layer interface and at the A/B soil horizon. Our results are for four periods that compare normal and "worst-case" conditions: prefertilization, immediately following fertilization, following a heavy rainfall, and 10 months after fertilization. During these four periods, spatial differences among membranes positioned in the clearcut, at the dripline of the SMZ, and at the creek were examined among treatments. One would expect nitrate and ammonium levels to be greatest in the clearcut and at the SMZ dripline following fertilization with decreasing nutrient levels existing at the creek as SMZ width increases. Furthermore, elevated nutrient levels following fertilization are expected to eventually return to prefertilization levels.

Prior to Fertilization

Membranes analyzed prior to fertilization indicate that ammonium and nitrate levels were <2.5 mg/m^2/day for any given treatment (fig. 1). There is no significant difference among cation or anion membrane locations prior to fertilization. Nitrate and ammonium values in the clearcut were naturally higher than at the SMZ dripline and stream positions most likely due to increased decomposition and microbial fixation—an example of the assart effect.

Immediately Following Fertilization

Results from membranes analyzed immediately following fertilization indicate that ammonium levels ranged from 24 to 230 mg/m^2/day in the clearcut but <5 mg/m^2/day at the SMZ dripline and <2.5 mg/m^2/day at the creek for any given treatment. There are significant differences among cation membrane locations immediately following fertilization. The clearcut positions on all four treatments, in addition to the SMZ dripline position on the 30.5-m treatment, are significantly related (fig. 2).

The clearcut position on the 7.6-m, 15.2-m, and 15.2-m thin treatment are also statistically similar (fig. 2). Nitrate levels were much lower than ammonium levels immediately after fertilization and ranged from only 3 to 9 mg/m^2/day in the clearcut but <0.5 mg/m^2/day at the SMZ dripline and <0.4 mg/m^2/day at the creek for any given treatment. There are significant differences among anion membrane locations immediately following fertilization. These statistical differences were only apparent in the clearcut areas across the four treatments (fig. 2).

Following Heavy Rainfall

Membranes analyzed following a heavy rainfall indicate that ammonium levels ranged from 30 to 112 mg/m^2/day in the clearcut but <5 mg/m^2/day at the SMZ dripline and <1 mg/m^2/day at the creek for any given treatment. There are significant differences among cation membrane locations following heavy rainfall. Significant differences were between the clearcut position and the dripline and stream positions (fig. 3).

Nitrate levels were slightly higher than ammonium following a heavy rainfall and ranged from 54 to 165 mg/m^2/day in the clearcut but <28 mg/m^2/day at the SMZ dripline and <6 mg/m^2/day at the creek for any given treatment.

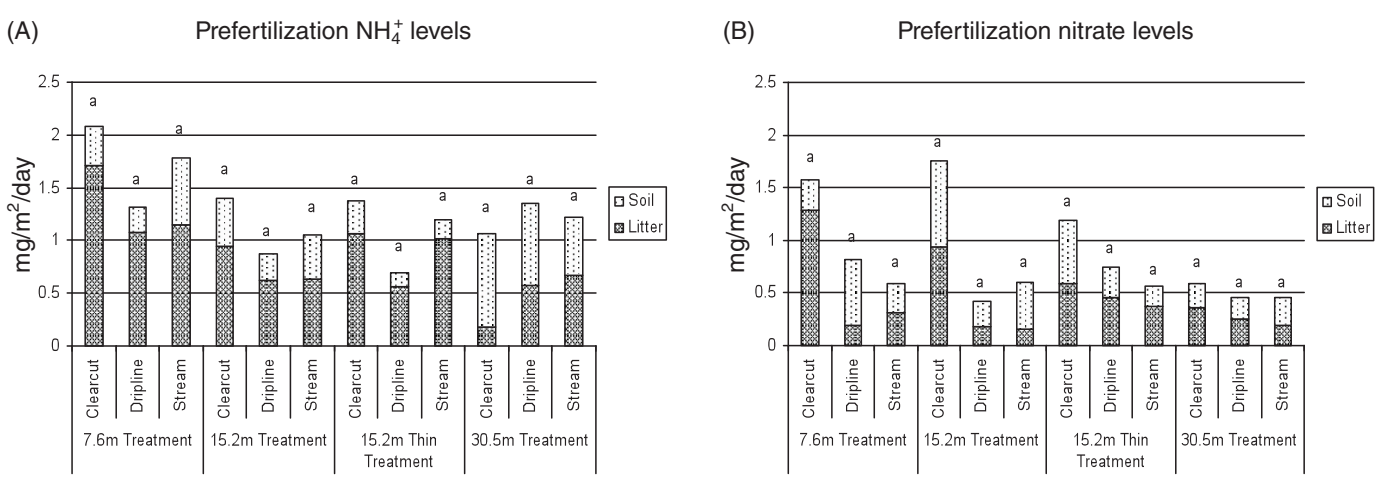

Figure 1—(A) Average ammonium and (B) nitrate levels for all treatments prior to fertilization.

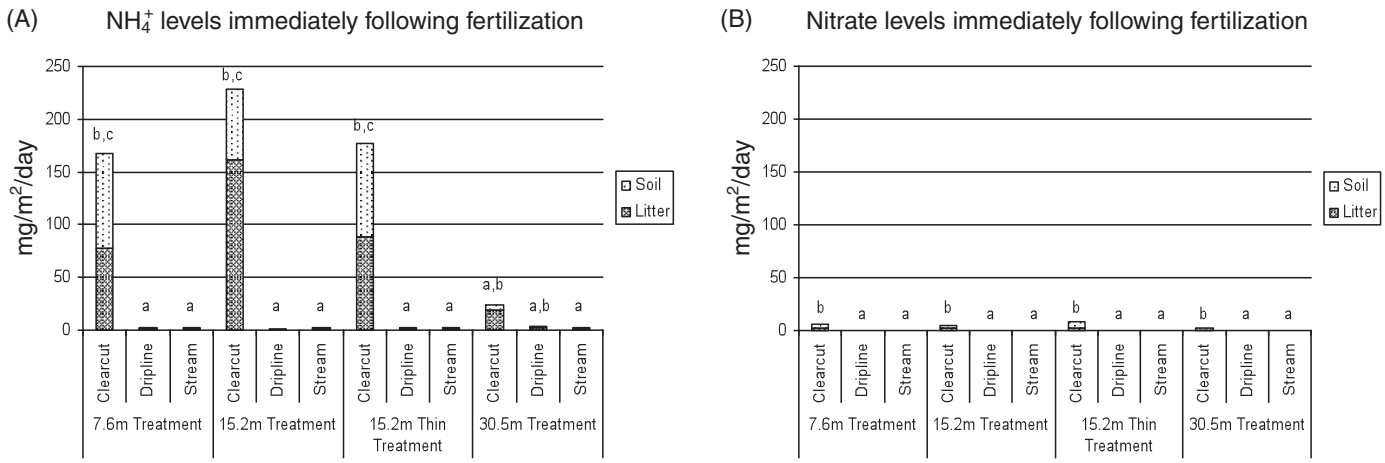

Figure 2—(A) Average ammonium and (B) nitrate levels for all treatments immediately following fertilization.

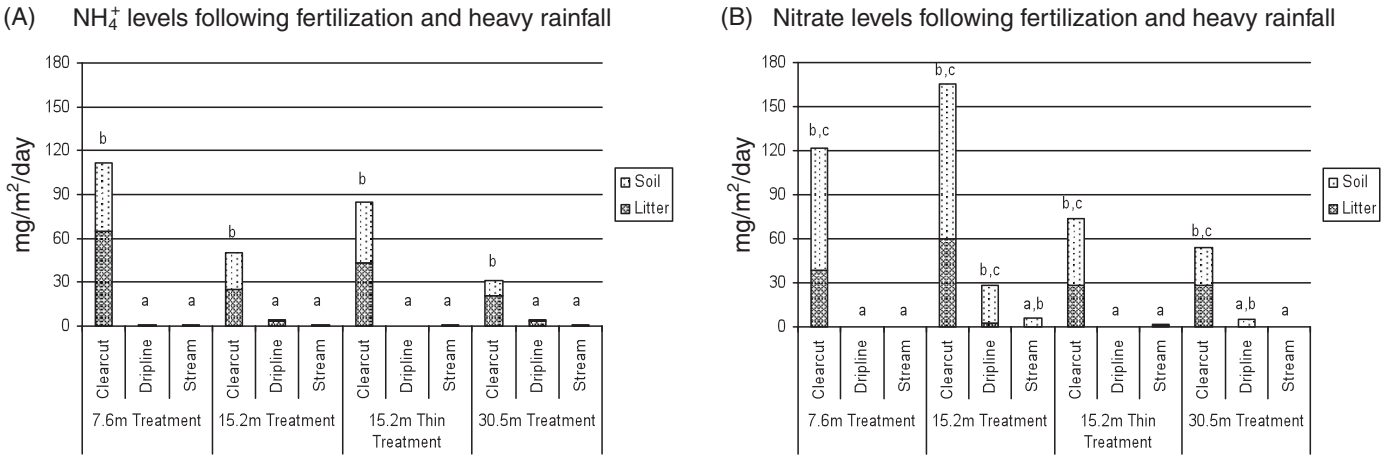

Figure 3—(A) Average ammonium and (B) nitrate levels for all treatments following a heavy rainfall.

There are significant differences among anion membrane locations following heavy rainfall (fig. 3). There is a statistical difference between the clearcut positions and the SMZ dripline position on the 15.2-m treatment and all the other SMZ dripline and stream positions. Furthermore, the stream position on the 15.2-m treatment as well as the SMZ dripline position on the 30.5-m treatment are statistically similar to the clearcut positions and 15.2-m treatment SMZ dripline. These relationships suggest that the largest movement of nitrogen in the upper soil layer and litter layer occurs as nitrate after a heavy rainfall.

Ten Months Following Fertilization

Membranes analyzed 10 months following fertilization indicate that ammonium levels ranged from only 1 to 9 mg/m^2/day in the clearcut but <4.5 mg/m^2/day at the SMZ dripline and <4 mg/m^2/day at the stream for any given treatment. There are no significant differences among cation membrane locations 10 months after fertilization (fig. 4).

As with the cation membranes there are no significant differences among anion membrane locations 10 months following fertilization (fig. 4). Nitrate levels 10 months following fertilization ranged from only 12 to 45 mg/m^2/day in the clearcut but <4 mg/m^2/day at the SMZ dripline and <2 mg/m^2/day at the creek for any given treatment.

CONCLUSIONS

Some preliminary conclusions can be drawn from this data. First, high levels of ammonium and nitrate found in the clearcut after fertilization are not evident at the SMZ dripline or the stream positions. Values at SMZ dripline locations reached 28 mg/m^2/day for nitrate on the 15.2-m treatment following a heavy rainfall but never got above 5 mg/m^2/day for any treatment at any time throughout the year-long study period. Furthermore, stream location values never got above 6 mg/m^2/day for either ammonium or nitrate at any treatment.

Second, there appears to be lag time for nitrate to become present after fertilization most likely due to the need for

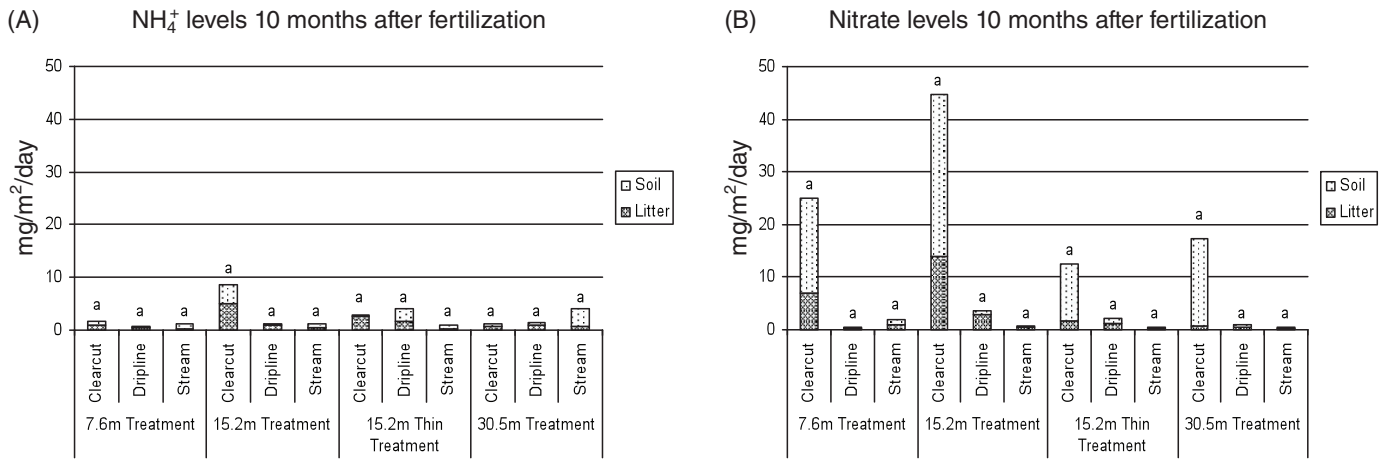

Figure 4—(A) Average ammonium and (B) nitrate levels for all treatments 10 months after fertilization.

mineralization of organic N. Average nitrate levels in the clearcuts 2 weeks after fertilization were <9 mg/m^2/day. Meanwhile average ammonium ranged between 24 and 230 mg/m^2/day in the clearcuts 2 weeks after fertilization. By 10 months, ammonium and nitrate levels in the clearcut are returning to prefertilization levels. After 10 months, values are still higher than prior to fertilization but appear to approach prefertilization levels.

It may seem reasonable to suggest that a 7.6-m wide SMZ with forested vegetation is sufficient for removing N in these Piedmont sites due to lack of ammonium and nitrate found near the creeks. However, it appears that water movement below the litter layer and at shallow soil layers is not an important pathway for N movement because position value differences did not seem to exist among treatments. Furthermore, after fertilization, ammonium and nitrate levels were seldom higher at the stream or even the SMZ dripline than they were prior to fertilization. It seems more reasonable to suggest that if the nitrate and ammonium are moving out of the clearcut, then it is probably in ground water deeper in the soil profile. Therefore, further analysis on subsurface N movement is certainly necessary before an SMZ width can be recommended based on the impact from industrial fertilizer application.

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LITERATURE CITED

Ambus, P.; Lowrance, R. 1991. Comparison of denitrification in two riparian soils. Soil Science Society of America. 55(4): 994–997.

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

Castelle, A.; Johnson, A.; Conolly, C. 1994. Wetland and stream buffer size requirements—a review. Journal of Environmental Quality. 23: 878–882.

Cooperband, L.R.; Logan, T.J. 1994. Measuring in situ changes in labile soil phosphorous with anion-exchange membranes. Soil Science Society of America. 58(Jan–Feb): 105–114.

Elliot, J.R. 2006. Effects of a control release nitrogen fertilizer and thinning on the nitrogen dynamics of a mid-rotation loblolly pine stand in the Piedmont of Virginia. Blacksburg, VA: Virginia Tech. 166 p. M.S. thesis.

Fox, T.R.; Allen, H.L.; Albaugh, T.J. [and others]. 2007. Forest fertilization and water quality in the United States. Better Crops. 91(1): 7–9.

Gaskin, J.W.; Dowd, J.F.; Nutter, W.L.; Swank, W.T. 1989. Vertical and lateral components of soil nutrient flux in a hillslope. Journal Of Environmental Quality. 18(Oct–Dec): 403–410.

Huang, W.Z.; Schoenau, J.J. 1996. Microsite assessment of forest soil nitrogen, phosphorous, and potassium supply rates in-field using ion exchange membranes. Communications in Soil Science and Plant Analysis. 27(15–17): 2895–2908.

Ice, G.G.; Stuart, G.W.; Waide, J.B. [and others]. 1997. Twenty-five years of the Clean Water Act: how clean are forest practices. Journal of Forestry. 95(7): 9–13.

Jacobs, T.C.; Gilliam, J.W. 1985. Riparian losses of nitrate from agricultural drainage waters. Journal of Environmental Quality. 14(4): 472–478.

Keim, R.F.; Schoenholtz, S.H. 1999. Functions and effectiveness of silvicultural streamside management zones in loessial bluff forests. Forest Ecology and Management. 118(1–3): 197–209.

Lakel, W.A.; Aust, W.M.; Dolloff, C.A. 2006a. Effects of forestry streamside management zones on water quality in Virginia. The 2006 Council on Forest Engineering. [Number of pages unknown].

Lakel, W.A.; Aust, W.M.; Dolloff, C.A. 2006b. Seeing the trees along the streamside. Journal of Soil and Water Conservation. 61(1): 22–29.

Lowrance, R.R.; Todd, R.L.; Asmussen, L.E. 1984. Nutrient cycling in an agricultural watershed: I. Phreatic movement. Journal of Environmental Quality. 13(1): 22–27.

O'Laughlin, J.; Belt, G.H. 1995. Functional approaches to riparian buffer strip design. Journal of Forestry. 93(2): 29–32.

U.S. Environmental Protection Agency. 2008. History of the Clean Water Act. Laws, regulations, guidance, and dockets. November 26, 2008. http://www.epa.gov/regulations/laws/cwahistory.html. [Date accessed: July 25, 2011].

Virginia Department of Forestry. 2002. Virginia's forestry best management practices for water quality. http://cnre.vt.edu/forestupdate/presentations/STHM/Irvine%20Riparian%20Buffers,%20BMPs%20and%20Water%20Quality.pdf. [Date accessed: July 24, 2011].

Walbridge, M.R. 1993. Functions and values of forested wetlands in the Southern United States. Journal of Forestry. 91(5): 15–19.

Welsch, D.J. 1996. Riparian forest buffers: function and design for protection and enhancement of water resources. Publ. NA-PR-07-91. [Broomall, PA]: U.S. Department of Agriculture Forest Service, Northern Area State and Private Forestry. 24 p.

Wiseman, P.E.; Seiler, J.R. 2004. Soil CO2 efflux across four age classes of plantation loblolly pine (*Pinus Taeda* L.) on the Virginia Piedmont. Forest Ecology and Management. 192(2–3): 297–311.

DETERMINATION OF FIELD-EFFECTIVE SOIL PROPERTIES IN THE TIDEWATER REGION OF NORTH CAROLINA

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Abstract—Soils vary spatially in texture, structure, depth of horizons, and macropores, which can lead to a large variation in soil physical properties. In particular, saturated hydraulic conductivity (K_{sat}) and drainable porosity are critical properties required to model field hydrology in poorly drained lands. These soil-property values can be measured by several methods; however, larger scale, "field-effective" values may be needed in developing, calibrating, and validating models, such as DRAINMOD. In this investigation, field-effective soil-property values were estimated from water table and outflow measurements from a 3-year field experiment on poorly drained loblolly pine (*Pinus taeda* L.) plantation watersheds in eastern North Carolina and tested against two additional estimation methods. Field estimates for K_{sat} were compared to estimates determined using the auger-hole method and estimates determined from soil cores by the constant-head method. The field-effective K_{sat} of the surface layer was estimated at 140 and 90 cm/hour for the unthinned and thinned condition, respectively. These values are greater than values obtained from soil cores and not different from values obtained from the auger-hole method which had mean conductivities of 100 and 80 cm/hour for unthinned condition, respectively. The thinned condition had K_{sat} values of 32 and 17 cm/hour based on soil cores and the auger-hole method, respectively. The differences between the field-based values and the constant-head method and auger-hole method may be a result of heterogeneity of soils or overestimations in the field-effective results.

INTRODUCTION

Hydrologic models have, in recent years, received increased utilization as management planning tools to evaluate alternative management scenarios. Models have become essential tools in evaluating and forecasting effects of manmade disturbances, land use changes, and climate change. In the absence of long-term data for a specific scenario, as is typically the case, validated models can provide forest managers vital information to evaluate alternatives. Information attained from model predictions assist in evaluating the effects of changes on attributes ranging from site productivity to water quality (Cho and others 2009, Croke and others 2004, Sun and others 2006). In forest management, hydrologic models have shown promise in predicting hydrology and the influence of forest operations on hydrologic responses (Amatya and Skaggs 2001; Amatya and others 2000, 2004; Lovejoy and others 1997; Parsons and Trettin 2001; Sun and others 1998). However, accurate model predictions are dependent on quality site characterization.

Previous work has shown that site characteristics can greatly influence hydrology and hydrologic responses within a watershed (Detenbeck and others 2005; Lide and others 1995; Sun and others 2001, 2006). Site characteristics such as climate, slope, vegetation, topographic relief, soils, and geographic location affect the hydrograph resulting from a given precipitation event. Among these variables, soil properties are the most difficult to quantify for a given site due to their inherent spatial variability, disturbances associated with land use, and possible errors associated with measurement methods (Grace and others 2006a, Skaggs and others 2008). Soils are highly variable spatially in depth of horizons, structure, texture, organic matter content, and water release or holding properties (Cohen and others 2008, Wei and others 2008). This inherent high variability of soil properties within a watershed is compounded by variability caused by disturbances in the form of mechanized operations. These disturbances can result in compaction and rutting in the areas adjacent to and under tires (Carter and McDonald 1998, Grace and others 2006a) which can account for as much as 20 percent of the disturbed area following operations in a watershed (McDonald and others 1998, Stuart and Carr 1991). Compacted zones can further increase soil variability within the watershed which, in turn, can affect the soil water relationships in the soil.

Soil properties related to soil water movement, such as saturated hydraulic conductivity (K_{sat}), volume drained (V_d), and drainable porosity (f) are especially troublesome to measure *in situ* or by cores in the forested setting due to the existence of roots, voids left from root systems, and buried debris. In addition, hydraulic conductivity is typically measured and modeled as homogenous within layers in the soil profile. Previous work has shown that like most soil properties, K_{sat} and f are highly variable in an area as small as a m^2 (Grace and others 2006a). Consequently, point K_{sat} and f measurements may fail to accurately describe the field conditions from a modeling standpoint due to scaling issues. For example, Bierkens and van der Gaast (1998) presented errors associated with neglecting stochastic upscaling methods. Soil properties are typically determined *in situ* or laboratory determined based on soil cores or measurements collected from randomly selected locations. Core samples and *in-situ* measurements, such as the auger-hole method, are point values that may only represent a small portion of the site under consideration. Soil core samples represent the core scale, and the auger-hole method represents soil properties at the model block scale as described by Bierkens

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and van der Gaast (1998). However, hydrologic modeling is often performed at the local or regional scale which is a considerably larger scale than represented by cores or blocks. This fact presents the need for upscaling, deriving soil properties from smaller scale measurements, or determining soil properties from field hydrologic measurements.

Modeling the hydrology of a site often requires the best possible characterization, or field-effective values, for the soil properties associated with soil water movement for the site under consideration. Soil-property values can be determined from several methods; however, field-effective values are often needed in developing, calibrating, and validating models, such as DRAINMOD. The objective of this paper is to utilize water table and outflow measurements from a 3-year field experiment on loblolly pine (*Pinus taeda* L.) plantation watersheds in eastern North Carolina to estimate K_{sat}, V_d, and f. This alternative procedure and estimates were tested for differences with two alternative estimation methods—estimates determined from the auger-hole method and estimates determined from soil cores by the constant-head method.

METHODS

Site Description

The field experiment was located at approximately 35° latitude and 76° longitude in the Tidewater region near Plymouth, NC. The experiment was conducted on a 56-ha poorly drained loblolly pine plantation watershed owned and managed by Weyerhaeuser Company. The watershed was isolated from the surrounding forest lands by forest roads and a collector ditch. Soils on the study sites are mapped as primarily Belhaven muck soil series, an organic, shallow water table soil. Soils are highly organic with soil organic matter contents of 80 percent or greater in the top soil horizon (O_a horizon). Site elevation was 4.1 to 4.5 m above sea level with an average slope of <0.1 percent. The watershed was delineated into a 40-ha subwatershed (WS5) to receive a thinning treatment and a 16-ha subwatershed (WS2) that served as a control. These watersheds were separated from a hydrologic standpoint using earthen berms in the collector ditch.

Watersheds were instrumented with water table wells, storm water samplers, and up- and downstream stage recorders. The water table was monitored continuously with pressure transducers and hourly measurements recorded using dataloggers for each study watershed. Similarly, discharge was monitored continuously using submerged pressure transducers in combination with Stevens chart recorder. Precipitation information was collected with a tipping bucket rain sensor located within 0.5 km of the study sites. Descriptions of the measurement systems, collected data, and results from the 3-year study period have been reported (Grace and Skaggs 2006; Grace and others 2006b, 2007).

V_d, f, and K_{sat} values determined based on field experimental data were compared to values measured using soil cores and the auger-hole method reported by Grace and others (2007a) using SAS TTEST (alpha = 0.05) procedures (SAS 2004). The hypothesis was that no mean difference exists in soil-property values for the three methods utilized in this work. Subsequent model predictions based on values determined for the methods were evaluated using the Nash-Sutcliffe model efficiency (ME) coefficient (Nash and Sutcliffe 1970).

Volume Drained

V_d determinations used drainage, water table depth (WTD), and rainfall data collected during the study period from the field experiment. This procedure involved performing a mass balance for rainfall events for the system as the water table changes from time T_1 to time T_2 (fig. 1). The mass balance is given as:

$$\Delta V_a = D + ET - R \tag{1}$$

where
R = rainfall (cm)
D = system drainage (cm)
ET = evapotranspiration losses (cm)
ΔV_a = change in V_d (cm) during the time period.

Record during and surrounding rain events was used in calculations of V_d, ET was assumed to be zero during those times. In addition, deep seepage was assumed negligible and excluded in this determination.

This equation expressed in terms of system flux and f is given by:

$$f\ \Delta WTD = q\ \Delta t - R \tag{2}$$

or

$$f = R - \sqrt{\frac{q\,\Delta t}{\Delta \mathrm{WTD}}} \tag{3}$$

where
f = drainable porosity
q = system flux (cm/hour)
Δt = change in time ($T_2 - T_1$)
ΔWTD = incremental change in water table depth at the midpoint from time T_1 to time T_2 (fig. 1)

Determination of the drainable porosity, f, from the equations above assumes the unsaturated zone is drained to equilibrium with the water table at all times. The assumption that the water content distribution at any time is similar to the distribution of the stationary water table and the profile approximately drained to equilibrium has been demonstrated for drained profiles by Skaggs and Tang (1976) and Tang and Skaggs (1978). However, it is understood that representing the water table profile as linear has a degree of error associated with the volume estimates as the water table recedes (development of the elliptical water table profile) i.e., lag time associated with midpoint water table change. This lag time is the period when water stored (bank storage) in the initially flat water table drains before the midpoint water table recedes and takes on the theoretical elliptical shape explained by the relationships. The f values determined from the drainage rate and WTD measurements in the relationships above can be underestimated due to the fact that the difference in V_d as the midpoint water table recedes under theoretical elliptical profile (fig. 2A) is <V_d calculated

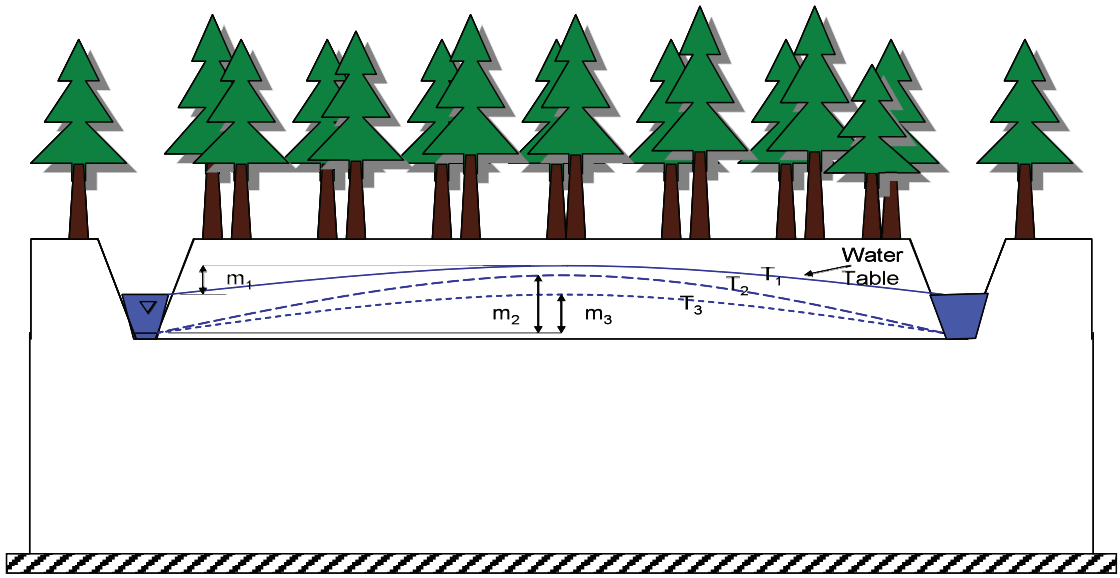

Figure 1—Water table profile following rain events as the water table recedes during a drainage event.

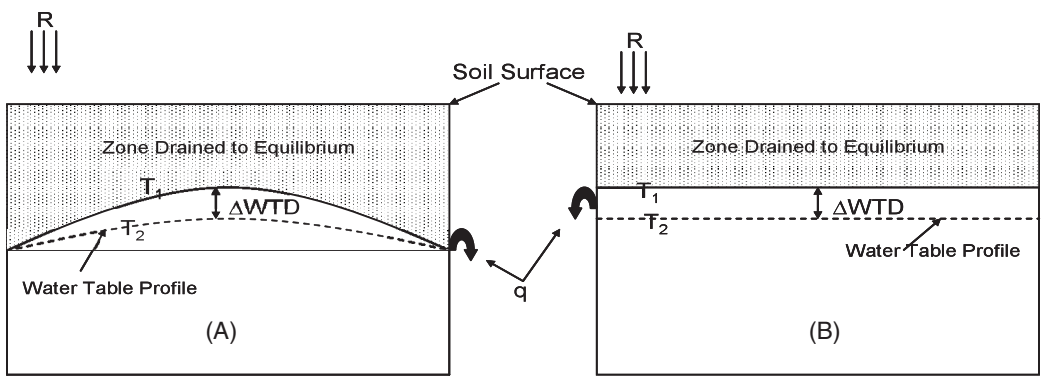

Figure 2—Diagram of theoretical elliptical water table profile during drainage events (A) and water table profile represented in the mass balance equations (B).

from mass balance relationships (fig. 2B) (McCarthy and Skaggs 1991). The elliptical water table profile was verified by observations from the watersheds in this investigation based on water table measurements in the lateral ditch, 1 m from the lateral ditch, 3 m from the lateral ditch, and 50 m from the lateral ditch (midpoint measurement).

Saturated Hydraulic Conductivity

K_{sat} was determined by first defining the relationship between drainage flux and height of the water table above the drain, $q(m)$, using procedures presented by Skaggs and others (2008). The $q(m)$ relationships for the unthinned and thinned conditions are plotted along with the main drainage curve (MDC) as defined by Skaggs and others (2008) (fig. 3). The MDC represents the $q(m)$ relationship for the profile in this investigation in the absence of rainfall based on solutions to the Boussinesq equation (Youngs 1999). Once the $q(m)$ relationship was defined, water table and drainage record from the field experiment were used to calculate field-effective K_{sat} (K_e) using the following expression based on the drainage equation derived by Hooghoudt (1940):

$$K_e = \sqrt{q\,L^2 \Big/ 4m\left(2d_e + m\right)} \tag{4}$$

where

q = flux (cm/hour)

d_e = equivalent depth of the impermeable layer below the depth of the parallel drains (cm) (for ditches, d_e is the equivalent depth below the water surface in the ditches)

L = distance between parallel drains (cm)

K_e = effective lateral hydraulic conductivity in the soil profile (cm/hour)

m = height of the water table above the water in the parallel drains at the point midway between the drains (cm)

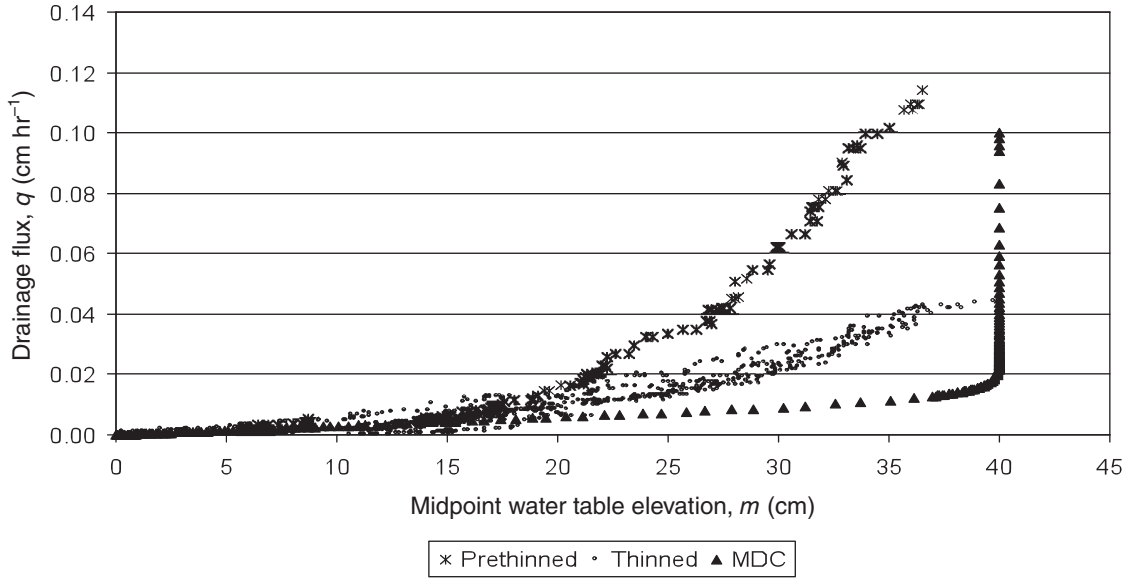

Figure 3—Relationship between observed drainage flux, q, and water table elevation above the ditch water level, m, during the study period for the prethinned and thinned condition. The main drainage curve (MDC) is presented as predicted by solutions to the Boussinesq equation.

These calculations assume that *ET* is negligible and the existence of steady state conditions. This method of using field measurements to calculate field effective K_{sat} relationships on agricultural lands has been presented as valid for similar field scale drainage systems during periods of low *ET* (Skaggs 1976). These methods, however, can result in an overestimation of hydraulic conductivity relationships due to *ET* losses (Skaggs 1976). In this analysis, minimizing the effect of *ET* on field-effective values required concentrating calculations during fall and winter periods and/or during rainfall events.

RESULTS AND DISCUSSION

The relationships between V_d and change in WTD for WS2 and WS5 over the 3-year study period can be seen in figure 4. V_d relationships for WS2 and WS5 were developed over a relatively narrow range of WTDs (45 to 85 cm). The f from the soil surface to a depth of 45 cm was assumed linear, i.e., assumed to have the same porosity as the 45- to 85-cm range that the relationships characterize. The V_d relationships developed for the WS2 and WS5 prethinned watersheds were not significantly different at the 0.05 level of significance based on t-tests ($P = 0.23$). Data from these prethinned conditions were combined to represent the unthinned condition for further analysis. V_d relationships developed for the watersheds based on field experiment data were statistically similar at the 0.05 level of significance based on t-tests between the unthinned condition and unthinned soil core data previously reported ($P = 0.080$). The V_d values from the field experiment and from soil cores were similar for the thinned condition based on statistical tests ($P = 0.353$).

The V_d for WS2 and WS5 based on relationships developed here give f of 0.14, 0.15, and 0.14 for WS2, the unthinned condition (WS2 and prethinned WS5 data combined) and the thinned WS5, respectively (fig. 4). Grace and others (2007) reported f values based on soil cores of 0.21 and 0.15 for WS2 WTD < and >60 cm (average of 0.18), respectively. WS5 f values based on soil cores were 0.15 and 0.10 for WTDs < and >60 cm (average of 0.13), respectively. Both methods used to develop V_d relationships have limitations. The V_d relationships developed from soil cores, as discussed previously, can misrepresent the site due to spatial variability of the soil medium. However, determinations of V_d relationships by this using observed drainage and water table response relationships are also limited by the drainage system characteristics. That is, V_d due to changes in WTD can only be determined when the water table elevation is above the ditch depth (in this case 85 cm belowground surface elevation). The water table for both WS2 and WS5 ranged between 45 and 85 cm below average ground surface elevation during drainage events, limiting the developed relationships to this range.

Field-effective values for K_{sat} were also determined for WS2 and WS5 based on the collected hydrology record (precipitation, WTD, ditch stage, and flow rate). The WTD vs. field-effective hydraulic conductivity relationships were presented graphically for the thinned and unthinned condition over the study period (fig. 5). Based on the field-effective K_{sat} values, the K_{sat} values of the surface layer for the unthinned and thinned condition were determined assuming the conductivity values obtained from soil cores in the deeper layers. The conductivity above the range of flow measurements (zero to 40 cm for the unthinned condition and zero to 30 cm for the thinned condition) was assumed similar to values in the adjacent layer (40 to 50 cm for the unthinned condition and 30 to 40 for the thinned condition). These

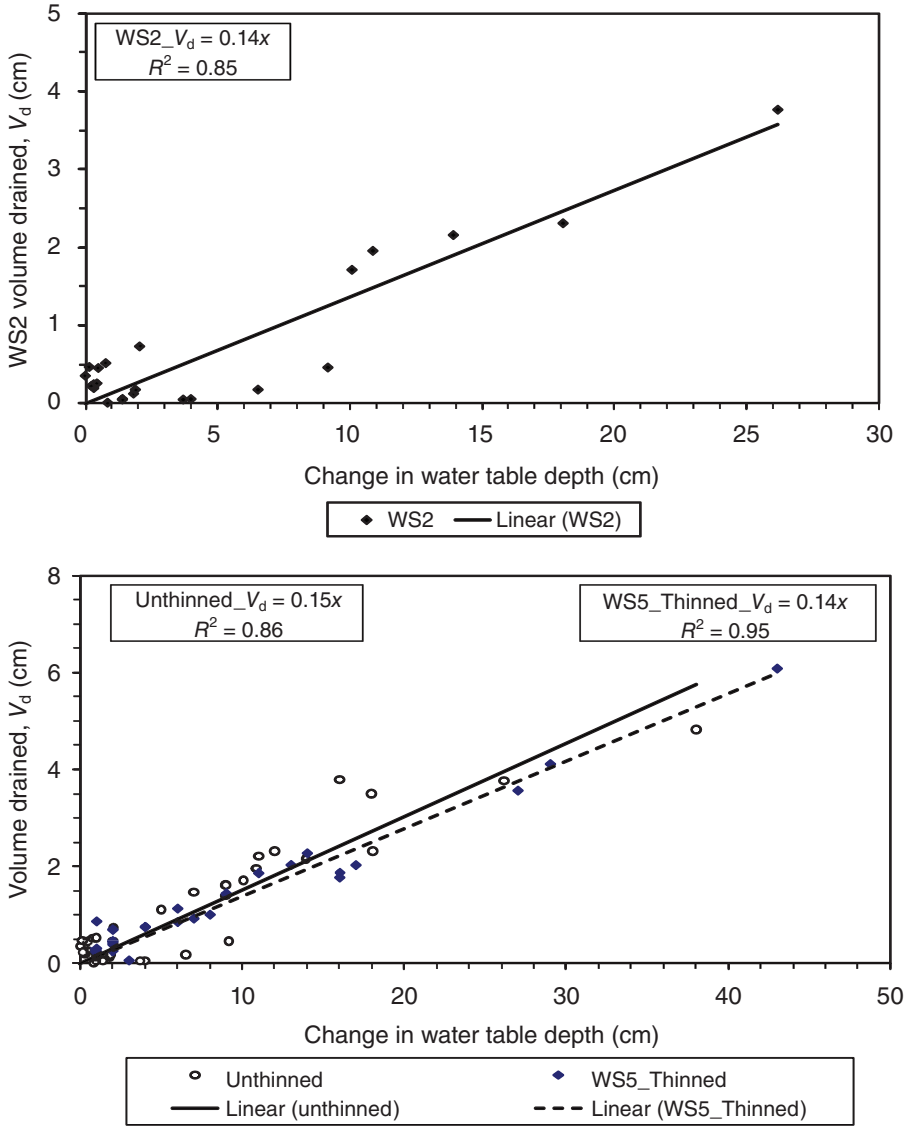

Figure 4—Drainable porosity relationship for WS2 and WS5 watersheds over the 3-year study period (2000–02) based on field-effective V_d computed from measured water table drawdown relationship.

assumptions are supported by the fact that: (1) the flow range is concentrated in the top 40 to 80 cm of the soil profile as presented in figure 5 and (2) previous work (Grace and others 2007) found no differences in cores taken at the 5- to 15-cm and 30- to 45-cm depths.

The mean K_{sat} of the surface layer is estimated at 140 and 90 cm/hour based on the field measurements for the unthinned and thinned condition, respectively. These values are greater than values reported based on soil cores (P = 0.002) but not different from values based on the auger-hole method (P = 0.578) which had mean conductivities of 100 and 80 cm/hour for unthinned condition, respectively (Grace and others 2007). Similarly, mean K_{sat} value for the thinned condition based on field experiment data was greater than values reported based on soil cores (K_{sat} = 32 cm/hour; P < 0.0001) and the auger-hole method (K_{sat} = 17 cm/hour; P = 0.005). The K_{sat} values determined from these field measurements are also greater than values reported for similar soils in the Tidewater region which had values similar to those determined from soil cores (Broadhead and Skaggs 1989). The upper limit of K_{sat} values determined from soil cores by the constant-head method and the auger-hole method was 500 and 170 cm/hour for these watersheds, respectively. These values are similar to the upper limit for field-effective K_{sat} for the unthinned and thinned watershed was 540 and 210 cm/hour, respectively.

The differences between the field-effective K_{sat} values and those obtained by the other two methods used may

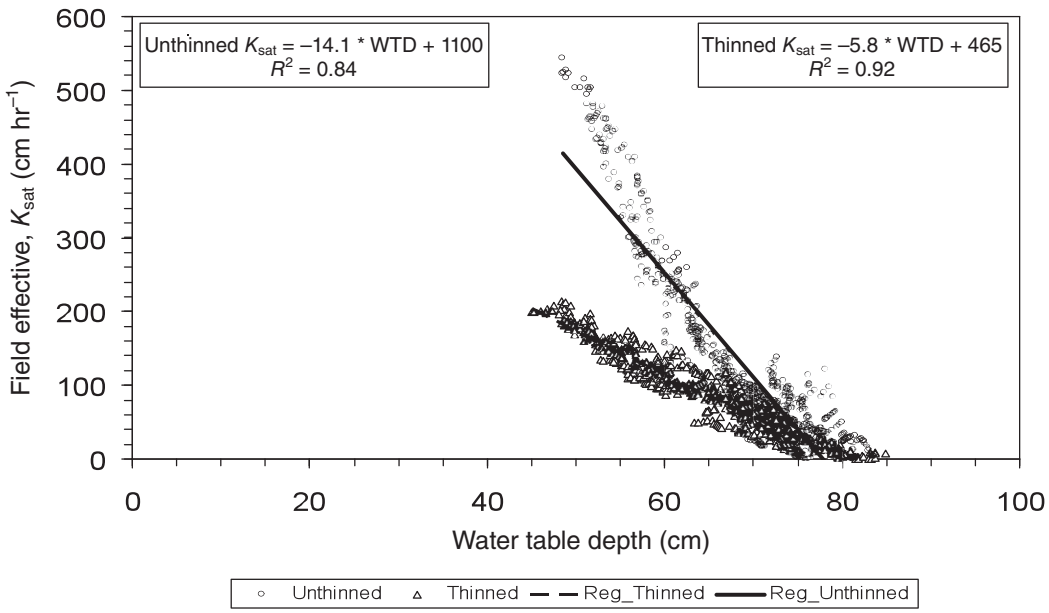

Figure 5—Relationship of observed water table depth and computed field-effective K_{sat} for the unthinned condition and for the thinned condition. Field-effective K_{sat} computed from water table and outflow measurements.

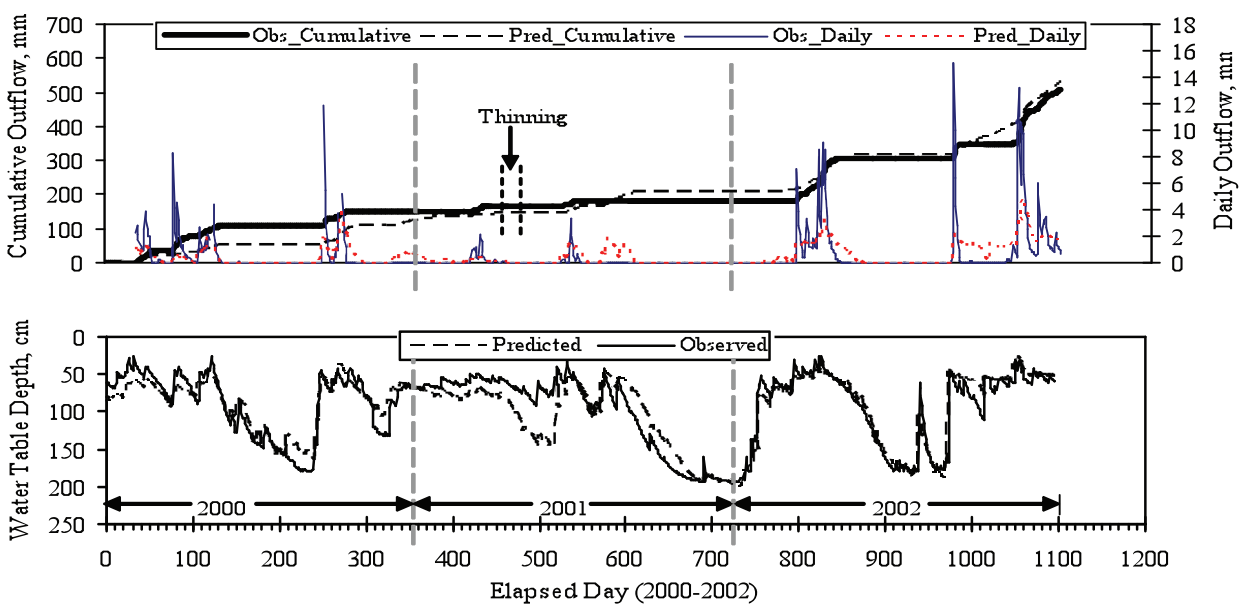

Figure 6—Observed and predicted WS5 cumulative outflow, daily outflow, and daily water table depth during the study period. Predictions based on field-effective V_d and K_{sat} relationships.

be a result of heterogeneity of soils discussed earlier or overestimations in the field-effective results. These factors likely contributed to the observed differences in methods; however, the values determined from the auger-hole and constant-head methods are consistent (within 20 percent) (P = 0.150). The field-effective values may be overestimated due to the influence of *ET* or deep seepage which was assumed negligible in determining field-effective K_{sat} values from field measurement as discussed earlier. Skaggs (1976) presented vertical losses, *ET*, and deep seepage as components that can cause significant overestimation errors in determinations from these field-based measures. *ET* losses

in the determinations made here were assumed negligible but may have been greater than estimated during the periods of record. The errors created by *ET* losses would likely result in the higher K_{sat} values seen in these calculations.

The field-effective values reported here and values previously reported based on soil cores and auger-hole tests were used in predictions of the hydrology from the poorly drained watersheds in this investigation. DRAINMOD predictions based on field-effective and soil core soil input files were compared statistically (table 1) and graphically (figs. 6 and 7). Cumulative outflow was overpredicted using soil inputs from both methods; however, predictions using values based on soil cores and auger-hole determinations were closer to the observed outflow. A deeper water table was also predicted using values from both methods. The absolute average daily difference (AADD) in outflow was similar for the predictions at 0.45 and 0.46 mm. Based on the results of the predictions, the field-effective soil property inputs resulted in more efficient predictions of outflow with a ME of 0.45 compared to the values representing soil cores which had a ME of 0.42. Both these predictions only resulted in fair agreement between

Table 1—Predictions and statistics for outflow and water table depth components based on values determined based on soil cores and field-effective values

Parameter	Observed	Predicted with core values	Predicted with field effective values
Precipitation, mm	3294	3294	3294
Cumulative outflow, mm	508	527	531
Daily average WTD, cm	95	100	97
ET, mm	2789	2771	2670
AADD outflow, mm		0.45	0.46
AADD WTD, cm		15	14
Outflow ME[a]		0.42	0.45
WTD ME[a]		0.84	0.82

WTD = water table depth; ET = evapotranspiration; AADD = absolute average daily difference.

[a] Nash-Sutcliffe model efficiency coefficient (ME).

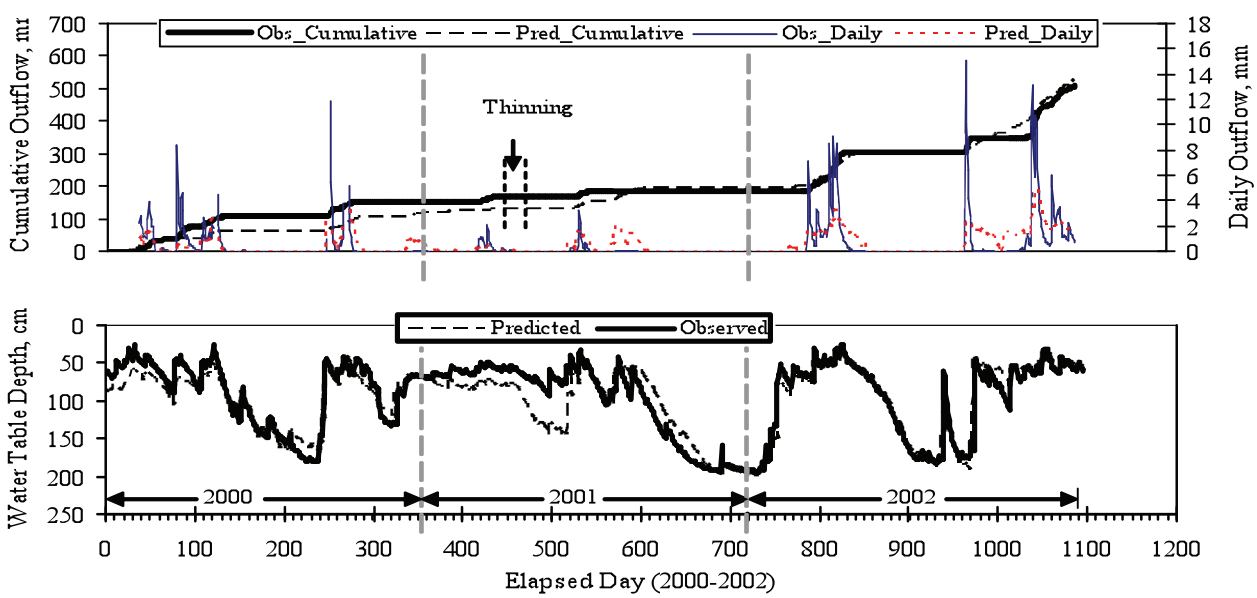

Figure 7—Observed and predicted WS5 cumulative outflow, daily outflow, and daily water table depth during the study period. Predictions based on reported soil core and auger-hole V_d and K_{sat} values.

predicted and observed outflow based on the ME values reported here. Field-effective soil property inputs resulted in a slightly less efficient prediction of WTD (ME = 0.82) in comparison to predictions based on soil property inputs from soil core values (ME = 0.84).

The differences in the predictions are illustrated in the graphical representation of outflow and WTD for the simulation period. Predictions based on field-effective values (fig. 6) show a deeper water table than that of predictions based on soil inputs from soil core values (fig. 7). Field-effective K_{sat} values resulted in a wetter site based on cumulative outflow record (table 1) and more responsive outflow pattern (fig. 6) than found for the soil core input values.

Daily peak outflow rates were in fair agreement to those observed during the study period for predictions based on these field-effective values. However, the deeper water table during most periods following thinning predicted using field-effective values indicate that V_d relationships may have been overestimated in predictions. In contrast, soil inputs based on soil core values resulted in better water table agreement as illustrated in figure 7 and supported by a stronger coefficient of model efficiency (ME = 0.84) (table 1). The predicted outflow hydrograph shows longer duration drainage events with decreased peak outflow rates for predictions based on soil core values. Cumulative outflow predictions based on soil core values did have better agreement during wet periods over the study period than predictions with field-effective soil-property values based on this analysis. These results indicate that true soil-property values for V_d, f, and K_{sat} likely lies somewhere between values obtained from soil cores and auger-hole tests and those obtained in this determination based on outflow and water table record.

CONCLUSIONS

Precipitation, outflow, and water table data were utilized to calculate field-effective soil property inputs for V_d, f, and K_{sat} for artificially drained pine plantation watersheds in this investigation. V_d and subsequent f relationships estimated in this work were similar to those determined based on soil cores collected in the field experiment. The K_{sat} values determined by the constant-head method was less than those presented based on outflow and water table response in the field experiment. Analysis revealed that the field-effective K_{sat} values determined were not different from the values measured using the auger-hole method for the unthinned condition. The field-effective conductivity values determined likely overestimated field conductivities whereas conductivity determined from the constant-head and auger-hole methods likely underestimated the values. The predictions resulting from these alternatives presented in this work appear to support this assumption. The K_{sat} values determined as field-effective values resulted in increased cumulative outflow and a deeper water table for the majority of the study period for both watersheds. The K_{sat} of the surface soil layer lies between the values determined from the small scale tests, constant-head and auger-hole methods, and the field-effective values presented. However, the values are likely closer to the first than the latter. Due to this large variation in the K_{sat} values from the different methods, K_{sat} was regarded as a critical calibration parameter in modeling efforts for these and similar watersheds (Grace and Skaggs 2006).

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LITERATURE CITED

Amatya, D.M.; Chescheir, G.M.; Fernandez, G.P. [and others]. 2004. DRAINWAT—based methods for estimating nitrogen transport in poorly drained watersheds. Transactions of the ASAE. 47(3): 677–687.

Amatya, D.M.; Gregory, J.D.; Skaggs, R.W. 2000. Effects of controlled drainage on storm event hydrology in a loblolly pine plantation. Journal of the American Water Resources Association. 36(1): 175–190.

Amatya, D.M.; Skaggs, R.W. 2001. Hydrologic modeling of a drained pine plantation on poorly drained soils. Forest Science. 47(1): 103–114.

Bierkens, M.F.P.; van der Gaast, J.W.J. 1998. Upscaling hydraulic conductivity: theory and examples from geohydrological studies. Nutrient Cycling in Agroecosystems. 50: 193–207.

Broadhead, R.G.; Skaggs, R.W. 1989. A hydrologic model for the artificially drained North Carolina peatlands. In: Dodd, V.A.; Grace, P.M., eds. Land and water use. Proceedings of the eleventh international congress on Agricultural Engineering: [Number of pages unknown].

Carter, E.A.; McDonald, T.P. 1998. Interaction among machine traffic, soil physical properties and loblolly pine root proliferation in a Piedmont soil. In: Waldrop, T.A., ed. Proceedings of the ninth biennial southern silvilcultural research conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 368–372.

Cho, J.; Barone, V.A.; Mostaghimi, S. 2009. Simulation of land use impacts on groundwater levels and streamflow in a Virginia watershed. Agricultural Water Management. 96(1): 1–11.

Cohen, M.J.; Dunne, E.J.; Bruland, G.L. 2008. Spatial variability of soil properties in cypress domes surrounded by different land uses. Wetlands. 28(2): 411–422.

Croke, B.F.W.; Merritt, W.S.; Jakeman, A.J. 2004. A dynamic model for predicting hydrologic response to land cover changes in gauged and ungauged catchments. Journal of Hydrology. 291(2): 115–131.

Detenbeck, N.E.; Brady, V.J.; Taylor, D.L. [and others]. 2005. Relationship of stream flow regime in the western Lake Superior basin to watershed type characteristics. Journal of Hydrology. 309: 258–276.

Grace, J.M., III; Skaggs, R.W. 2006. Water balance on drained plantation watersheds in North Carolina. In: Proceedings of the international conference, hydrology and management of forested wetlands. St. Joseph, MI: American Society of Agricultural and Biological Engineers: 58–67.

Grace, J.M., III; Skaggs, R.W.; Cassel, D.K. 2006a. Soil physical changes associated with forest harvesting operations on an organic soil. Soil Science Society of America Journal. 70(2): 503–509.

Grace, J.M., III; Skaggs, R.W.; Chescheir, G.M. 2006b. Hydrologic and water quality effects of thinning loblolly pine. Transactions of the ASABE. 49(3): 645–654.

Grace, J.M., III; Skaggs, R.W.; Cassel, D.K. 2007. Influence of thinning loblolly pine (*Pinus taeda* L.) on hydraulic properties of an organic soil. Transactions of the ASABE. 50(2): 517–522.

Hooghoudt, S.B. 1940. Bijdragen tot de kennis van enige natuurkundige grootheden van de ground. No. 7 Versl. Landb. Onderz. 42: 449–541.

Lide, R.M.; Pinder, V.J., III; Beatty, L. 1995. Hydrology of a Carolina Bay located on the upper Coastal Plain of western South Carolina. Wetlands. 15: 47–57.

Lovejoy, S.B.; Lee, J.G.; Randhir, T.O.; Engel, B.A. 1997. Research needs for water quality management in the 21st century: a spatial decision support system. Journal of Soil and Water Conservation. 52(1): 18–22.

McCarthy, E.J.; Skaggs, R.W. 1991. A simplified model for predicting drainage rates for changing boundary conditions. Transactions of the ASAE. 34(2): 443–448.

McDonald, T.; Carter, E.; Taylor, S.; Tobert, J. 1998. Relationship between site disturbance and forest harvesting equipment traffic. In: Proceedings, SOFOR GIS '98, 2nd southern forestry GIS conference: 85–92.

Nash, J.E.; Sutcliffe, J.V. 1970. River flow forecasting through conceptual models: part 1—a discussion of principles. Journal of Hydrology. 10: 282–290.

Parsons, J.E.; Trettin, C.C. 2001. Simulation of hydrology of short rotation hardwood plantations. In: Proceedings, 2001 ASAE annual international meeting. Pap. 012108. St. Joseph, MI: American Society of Agricultural and Biological Engineers: [Number of pages unknown].

SAS Institute Inc. 2004. SAS online. Doc 9.1.3. Cary, NC: SAS Institute Inc.

Skaggs, R.W. 1976. Determination of the hydraulic conductivity—drainable porosity ratio from water table measurements. Transactions of the ASAE. 19(1): 73–84.

Skaggs, R.W.; Chescheir, G.M.; Amatya, D.M.; Diggs, J.D. 2008. Effects of drainage and forest management practices on hydraulic conductivity of wetland soils..In: Farrell, C.; Feehan, J., eds. Proceedings 13th international peat congress; after wise use—the future of peatlands. Jyvaskyla, Finland: International Peat Society: 452–456. Vol. 1.

Skaggs, R.W.; Tang, Y.K. 1976. Saturated and unsaturated flow to parallel drains. Journal of Irrigation and Drainage Division of ASCE. 102(IR2): 221–238.

Stuart, W.B.; Carr, J.L. 1991. Harvesting impacts on steep slopes in Virginia. In: McCormick, L.H.; Gottschalk, K.W., eds. Proceedings 8th central hardwood conference. Gen. Tech. Rep. NE-148. Morgantown, WV: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station: 67–81.

Sun, G.; Callahan, T.J.; Pyzoha, J.E.; Trettin, C.C. 2006. Modeling the climatic and subsurface stratigraphy controls on the hydrology of a Carolina Bay wetland in South Carolina, USA. Wetlands. 26(2): 567–580.

Sun, G.; McNulty, S.G.; Shepard, J.P. [and others]. 2001. Effects of timber management on the hydrology of wetland forests in the Southern United States. Forest Ecology and Management. 143: 227–236.

Sun, G.; Riekerk, H.; Comerford, N.B. 1998. Modeling the hydrologic impacts of forest harvesting on Florida flatwoods. Journal of the American Water Resources Association. 34(4): 843–854.

Tang, Y.K.; Skaggs, R.W. 1978. Subsurface drainage in soils with high hydraulic conductivity layers. Transactions of the ASAE. 21(3): 515–521.

Wei, J. Xiao, D.; Zeng, H.; Fu, Y. 2008. Spatial variability of soil properties in relation to land use and topography in a typical small watershed of the black soil region, northeast China. Environmental Geology. 53: 1663–1672.

Youngs, E.G. 1999. Non-steady flow to drains. In: Skaggs, R.W.; van Schilfgaarde, J., eds. Agricultural drainage. Madison, WI: American Society of Agronomy: 265–296.

FOREST HEALTH

Aerial photo of oak mortality caused by the red oak borer on the Ozark National Forest. (Photo by Fred Stephen)

USING A GIS-BASED SPOT GROWTH MODEL AND VISUAL SIMULATOR TO EVALUATE THE EFFECTS OF SILVICULTURAL TREATMENTS ON SOUTHERN PINE BEETLE-INFESTED STANDS

Chiao-Ying Chou, Roy L. Hedden, Bo Song, and Thomas M. Williams[1]

Abstract—Many models are available for simulating the probability of southern pine beetle (*Dendroctonus frontalis* Zimmermann) (SPB) infestation and outbreak dynamics. However, only a few models focused on the potential spatial SPB growth. Although the integrated pest management systems are currently adopted, SPB management is still challenging because of diverse land ownership, dynamic forest landscapes, and uncertainty in spatial infestation pattern. In this study, we incorporated Geographical Information System (GIS)-based spot growth model into a three-dimensional visualization by using the visual simulator. The GIS maps of possible infestations were generated and used as the basis of three-dimensional visualizations to simulate spatial patterns of spot growth under various silvicultural treatments, including thinning, species mixtures, and different ages of stands. The results indicated these management practices, especially the thinning treatment, can reduce SPB infestation, particularly on the number of trees killed, but this does not necessarily result in a reduction of the infested area. We believe that GIS-based three-dimensional visualizations could provide more realistic landscapes without the spatial and temporal limitations for improving the SPB management decisionmaking process.

INTRODUCTION

The outbreaks of southern pine beetle [*Dendroctonus frontalis* Zimmermann (Coleoptera: Scolytidae)] (SPB) have caused severe ecological and economical damages (Price and others 1998). Consequently, the cause and spread of SPB has been studied extensively and its impacts are quite well understood. Owing to these efforts, existing models can predict both the probability of infestation and spot growth well for managers to make management plans with a minimum cost (Clarke 2001, Hedden 1985, Stephen and Lih 1985). However, although there are several regression and mechanistic models developed for simulating SPB spot growth, most models focus on predicting the number of trees killed in an infestation, and only few models were developed for spatial spot growth within forest stands.

In order to control and reduce the damage from SPB, the integrated pest management (IPM) system was used. IPM can help us to reduce pest populations and maintain them at levels below causing ecological and economic damage through the following main strategies (Edmonds and others 2000, Hedden 1978, Vité 1990): (1) developing a damage threshold where pest suppression is considered necessary, (2) establishing a detection-population monitoring system, and (3) developing silvicultural techniques to lower the population by interfering with the host selection behaviors of dispersing beetles. Despite the implementation of these effective management strategies responding to bark beetle (*Ips typographus* L.) hazards, there are still millions of acres of forests impacted by SPB infestations every year (Clarke 2001, Oliver and others 1994, Stephens and Ruth 2005).

Some challenges still constrain the land managers' ability to accomplish comprehensive IPM programs (Clarke 2001, Coster 1981, Stark and others 1985, Stephens and Ruth 2005). The major constraints are: (1) diverse land ownership—they are responsible for the pest detection and control on their lands with varying objectives and economic resources (Clarke 2001, Oliver and others 1994, Pyne and others 1996) and make it difficult to manage or control SPB outbreaks before or after infestations occurred when this damage covered widespread forest lands; (2) dynamic forest landscapes—although SPB hazards are more serious on pine (*Pinus* spp.) forest stands than other types of forest, these pine stands are dynamic and regional stand conditions are highly variable to make higher uncertainties when detecting the infestation regions and spreading patterns of SPB outbreaks; (3) the uncertainty effect from pest strategies—it is difficult to predict consequences of specified management strategies, such as buffer strips or mechanical thinning, for lowering SPB hazards on ecological or social impacts, and it's even more challenging to determine how efficient to achieve a specified operation (Almo 2006, Coster 1981, Martell 2001). Due to these constraints, forest managers and researchers are still challenged to control SPB hazards, to recover these damaged forests, and to determine the best restoration strategy for forest ecosystem and public awareness (Moore and others 1999, Pollet and Omi 2002, Stephens 1998, Waters 1985). For these reasons, an improved IPM system is needed for combining different objectives and resources from diverse land ownerships; organizing dynamic temporal and spatial data and information; monitoring, analyzing, and evaluating the ecological and social impacts from alternative management operations; and finally representing a comprehensive and sophisticated communication to ameliorate the decisionmaking process (Coster 1981, Sheppard and Salter 2004).

The main study objective was to provide three-dimensional visualizations to make the SPB management decisionmaking processes more effective. Three-dimensional visualizations are spatial representations and understandable

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communication techniques to help us to present different management alternatives and allow observation of forest landscapes without temporal and spatial limitations (McGaughey 1998, Orland 1994, Song and others 2006). Moreover, we used the Geographical Information System (GIS) maps of probable infestations as the basis of visual simulator to generate three-dimensional visualizations. Consequently, we aim to support a visual communication technique not only to deliver the complex information to different stakeholder groups with varying needs and degrees of knowledge on forest science, but also to delineate spatial and temporal changes in forest landscapes resulting from the multiple purposes and alternative SPB management operations (Sheppard and Salter 2004, Song and others 2006).

SIMULATION APPROACH

In this study, we simulated the spread probabilities of SPB spots in loblolly pine (*Pinus taeda*) stands within the southeastern Piedmont region of the United States. Spot growth was mapped by GIS-based SPB spot growth model using ArcObjects and Microsoft® Visual Basic for Applications in ArcGIS. Then, GIS maps of probable spot spread could be modified by initial stand characteristics for different SPB management practices, e.g., thinning, different species mixtures, and different ages of stands. Finally, these GIS maps of probable infestations were used as the basis of three-dimensional visualizations to simulate the trends of spot growth by using the Visual Nature Studio (VNS) software package (3D Nature 2002).

The Simulation of GIS-Based Southern Pine Beetle Spot Growth Model

Hedden and Billings (1979) developed the SPB spot growth model based on simple regression approach. We translated it into a GIS model. Because the SPB spot growth model was not originally designed for GIS base, we had to follow three steps to realize it (fig. 1, Chou and others 2008). In the first step, we determined the stand conditions, including species composition, stand area, stand density, average diameter at breast height (d.b.h.) and stand height, and spatial pattern of trees. The main patterns were uniform, random, and cluster, and these stand patterns would interact with silvicultural treatment effects. Silvicultural treatments influence the spatial pattern of trees and species mixture. In the second step, we calculated the dynamic of number of killed trees using SPB spot growth model for different affected stages during the period of infestation. In the last step, we built a spatial spreading module mainly based on the effects of nearest neighborhood, the susceptible species to pine beetle, season, and wind direction. Using the spatial spreading module within the ArcGIS programming environment, we could estimate the spatial arrangement of these infected trees as GIS maps. Therefore, according to this approach, we can generate any specified stand pattern by assigning the parameters of stand conditions.

Three-Dimensional Visualization of Southern Pine Beetle Spot Growth

A flowchart of the visual simulator (fig. 2) was generated for simulating the three-dimensional visual animations of

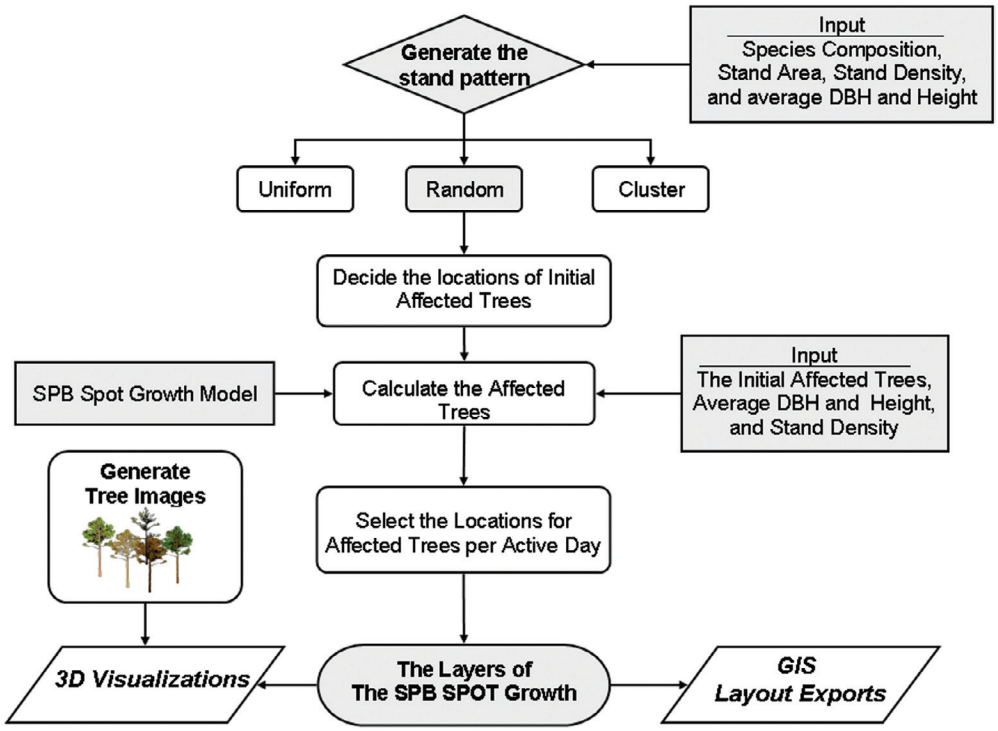

Figure 1—The simulation framework of the GIS-based southern pine beetle (SPB) spot growth model.

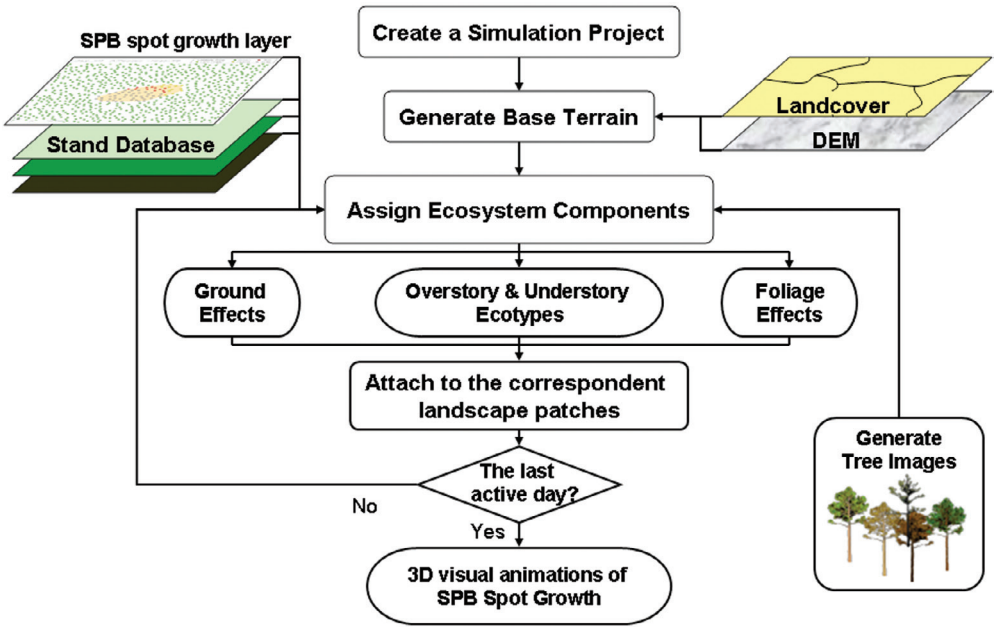

Figure 2—The flowchart of the visual simulator within Visual Nature Studio. (SPB = southern pine beetle, DEM = digital elevation model)

SPB spot growth. The environment of three-dimensional visualization is VNS, a three-dimensional, photorealistic, and landscape-visualization simulation software package (3D Nature 2002). Within it, we can directly import the landcover map and digital elevation model as our terrain base. Then, using the GIS maps of SPB spot growth, stand database, and specified tree images to assign the landscape patches (within VNS, they are called "ecosystem components"). The stand database includes stand density, stand average height, and species composition to support the required parameters for simulations. Moreover, we created specified photorealistic images to represent varied foliage effects from different affected stages and species with diverse colors and crown shapes (fig. 3).

We then assigned these "ecosystem components" by three divisions. First, the "ground effect" is visualized for the soil, litter, and other surface materials. Second, species composition, stand density, and average height are assigned for the "overstory and understory ecotype." And third, specified photorealistic tree images are linked to the appropriate tree "foliage effects." After attaching these ecosystem components to the correspondent landscape patches and stands, we can generate one scenario of SPB stop growth for 1 day. In order to simulate the trend of spot growth, we have to generate the three-dimensional visualizations for the whole period of spot growth by repeating the process of "assign ecosystem components" until the last day of the SPB spot growth simulation. Finally, we created the three-dimensional visual animations of SPB spot growth under the specified stand condition and generated other alternative SPB management operations by these procedures.

Comparing the Simulations of GIS Maps and Three-Dimensional Visualizations from Different Silvicultural Strategies

Following the above approach, we generated GIS maps and three-dimensional visualizations for comparing infestation sizes and spreading trends of SPB spot growth under different silvicultural strategies. First, we simulated stands [site index (SI) = 70, age = 40 years, height = 65 feet] with different stand densities, including low [basal area (BA) = 90 square feet per acre, d.b.h. = 7.76 inches], medium (BA = 120 square feet per acre, d.b.h. = 8.46 inches), and high (BA = 180 square feet per acre, d.b.h. = 9.15 inches) stand densities to see if stands with a higher density would cause more widespread damage. Second, we simulated a pure pine stand [natural loblolly pine (*Pinus taeda*)] and a mixed forest stand [mixture of loblolly pine, yellow-poplar (*Liriodendron tulipifera*), and white oak (*Quercus alba*)] within the same stand condition (SI = 70, age = 40 years, height = 65 feet, BA = 180 square feet per acre, d.b.h. = 9.15 inches) to determine whether different species compositions could affect the trends of spot growth. And third, we simulated infestation growth in a young loblolly pine plantation (about 15 years old, d.b.h. = 5.46 inches, height = 40 feet, SI = 55, BA = 120 square feet per acre) and mature loblolly pine stand (about 40 years old, d.b.h. = 8.46 inches, height = 65 feet, SI = 70, BA = 120 square feet per acre) to determine whether young stands are more resistant to SPB damage than mature stands. Using these simulation procedures, we compared both the number of trees killed and the infested area during 50 days among different management scenarios.

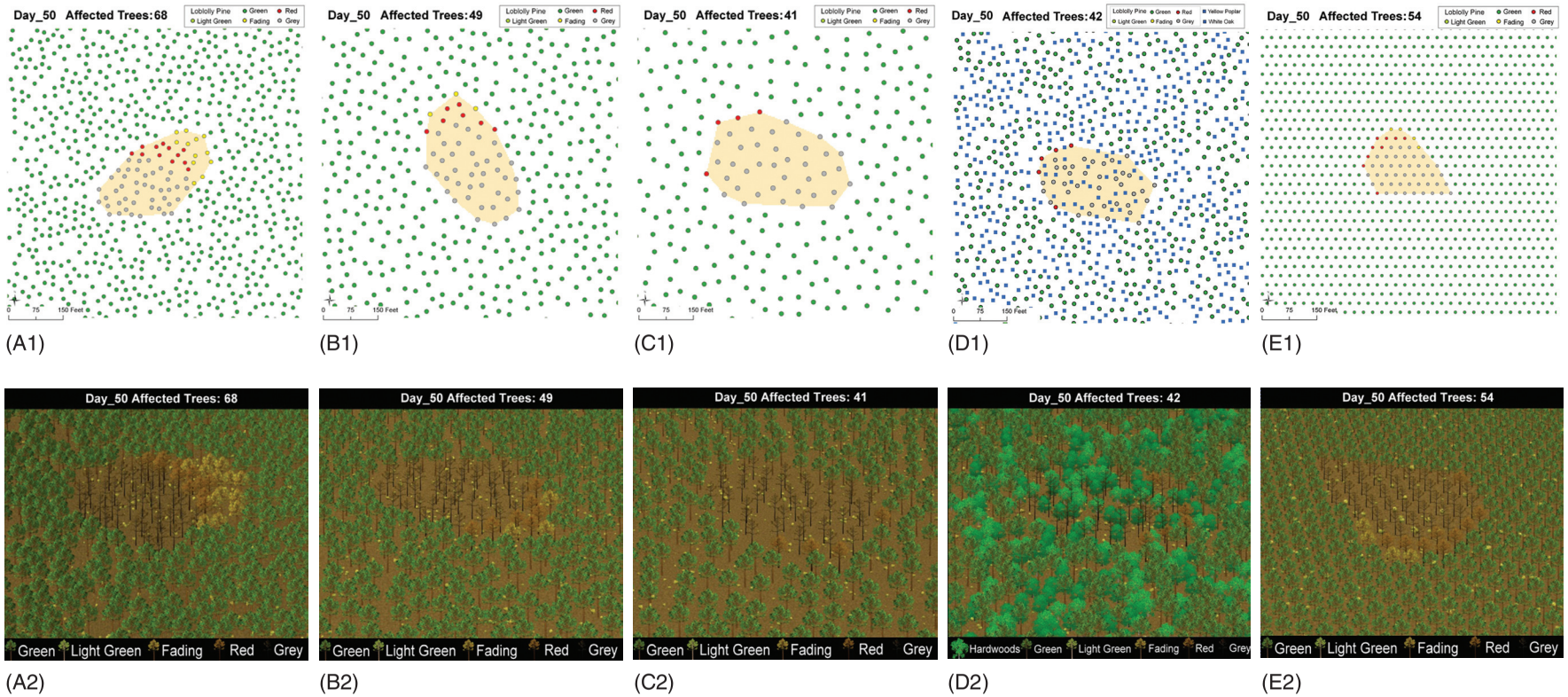

Figure 3—The comparisons of simulated GIS maps and three-dimensional visualizations of SPB spot growth in five different silvicultural treatments during 50 days. “a1” and “a2” were simulated for the high-density pine stand, “b1” and “b2” were simulated for the medium-density pine stand, “c1” and “c2” were simulated for the low-density pine stand, “d1” and “d2” were simulated for the mixed pine-hardwood forest stand, and “e1” and “e2” were simulated for the young loblolly pine plantation. “a1”, “b1”, “c1”, “d1”, and “e1” are GIS maps of SPB spot growth. “a2”, “b2”, “c2”, “d2”, and “e2” are three-dimensional visualizations of SPB spot growth.

RESULTS AND DISCUSSION

Through the simulation approach, we generated five different scenarios, including natural mature loblolly pine stands with high-, medium-, and low-stand density, a mixed forest stand with loblolly pine, white oak, and yellow-poplar, and a young loblolly pine plantation with medium-stand density. Simulation outputs for these five management scenarios were represented as GIS maps and three-dimensional visualizations.

The Comparison of Visualizations among Different Stand Densities

During the simulations, we generated three different kinds of stand densities in the mature loblolly pine stands (fig. 3). In the end of the spot growth simulation (day 50), the spot intensity (number of trees killed) in the high-density stand is the greatest (68, 49, and 41 affected trees for the high-, medium-, and low-density stand, respectively). However, the infested areas are larger in medium- and low-density stands (0.110, 0.137, and 0.177 acres for the high-, medium-, and low-density stands, respectively). The largest area occurred in the low-density stand.

Moreover, through the simulation of SPB spot growth in 50 days, we also generated animations for the SPB infestation dynamics with different stand densities. From these animations, we can see that the infested area in the low-density stand is the biggest, although its spreading speed of spot growth is always the lowest. Hence, not only can we figure out the trend of spot growth and compare their spreading speeds among different stand densities, but also, we can detect that the changing patterns of infested areas among them.

Therefore, the visualizations of GIS maps could show us the overall spatial pattern with abstract symbols (fig. 3). In addition, three-dimensional visualizations allow us to observe the same phenomena with more vivid foliage features, stereo viewshed, and specially designed tree images for different affected stages and tree species (fig. 3). Based on three-dimensional visualizations, we can see the trend of spot growth in the high-density stand is aggregated and extensive. In contrast, the spot spread trends are sparser and slower for medium- and low-density stands. Then, we can see that the number of trees killed has high positive correlation with stand density. However, the infested area has a negative relationship with it.

The Comparison of Visualizations between Loblolly Pine Stand and Mixture Forest Stand

When we compared the simulations of SPB spot growth in the pure loblolly pine stand and mixture forest stand (fig. 3), infested area in the pure pine stand (0.137 acres) is slightly smaller than the mixture species stand (0.129 acres); the spot intensity of the former is significantly greater than the latter (68 and 42 affected trees for pure pine and mixture forest stand, respectively).

Besides, if we are concerned more about the timber harvest in economy, the losses from SPB attacking in pure pine stand must be more serious than in the mixture forest stand since, in the pure pine stand, the speed of spot growth is faster and keeps growing. Although the infested area in the mixture forest stand is larger, its spot grows slowly. Therefore, if our concern is economic impact, we will be more interested in spot intensity than infested area.

Furthermore, from these GIS maps, it's difficult to represent different species by specified symbols. It is easier to identify different tree species in the three-dimensional visualizations. In VNS, we can use different foliage effects to represent diverse tree species, ages, and seasons by specified foliage colors, crown shapes, and form structures. Compared to the GIS maps, three-dimensional visualizations give a better and more realistic representation for the trend of spot growth in forest stands with different species compositions.

The Comparison of Visualizations between the Mature Loblolly Pine Stand and Young Pine Plantation

When comparing the simulations between the mature stand and young plantation, the number of trees killed in young pine plantation (54 affected trees) is a little bit larger than the mature stand (49 affected trees). The infested area in the latter (0.110 acres) is significantly greater than the former (0.066 acres). Here indicates that younger stand is not necessarily highly resistant to the SPB attacking, if the stand is still dense. Trends of spot growth between the mature and young loblolly pine stands are distinct in both the GIS maps and three-dimensional visualizations. One is more dense, and tall, and the other is more regular and small. Although, both of them have fast speed of spot spreading in dense stand. As a result, we can get a conclusion that the stand density, especially the distance between pines, is an important factor when we consider the spot intensity.

The Comparison of Spot Intensities and Infested Areas among Different Silvicultural Treatments

Furthermore, in this study, we emphasize the silvicultural treatments—the thinning, species mixture, and stand regeneration—could reduce the damage from SPB infestations especially when the duration of spot growth is longer, the effect of treatments would be more significant. From the figure of comparison of spot intensity (fig. 4), at the beginning, the differences among these treatments are not strong. Then, when the spots continue to grow, the differences among them become obvious.

Moreover, the reduction in stand density and the mixture in species composition can reduce the spot intensity. However, the reductions in the spot intensity do not necessarily result in a reduction in the area of the stand affected. From the figure of infested area comparison (fig. 5), the low-density stand always has the greatest infested area, and, next are medium-density and mixed-species stands. Therefore, when we aim to modify losses from SPB attacks by silvicultural treatments, we not only have to think about the effects on the spot intensity, but also the spatial impact of the infested area.

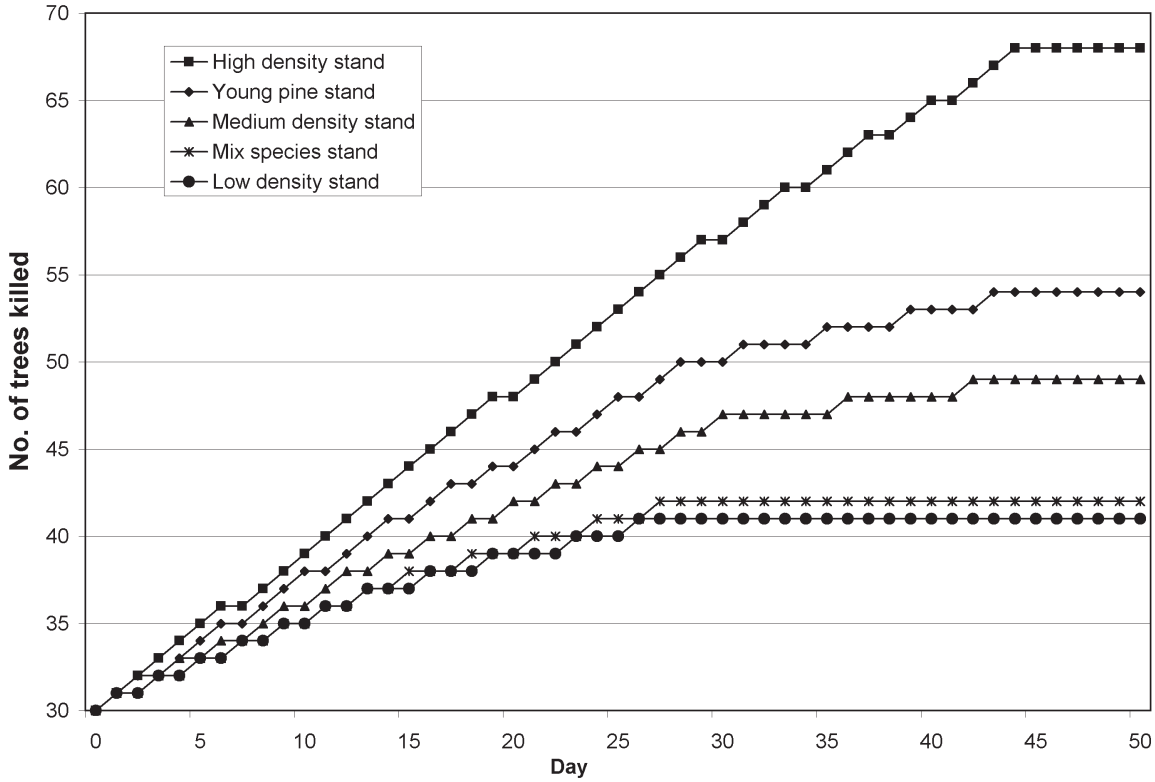

Figure 4—Comparisons of the number of trees killed (spot intensity) among five different silvicultural treatments during 50-day period of SPB infestation. The number of trees killed on the initial day (day 0) is 30 trees.

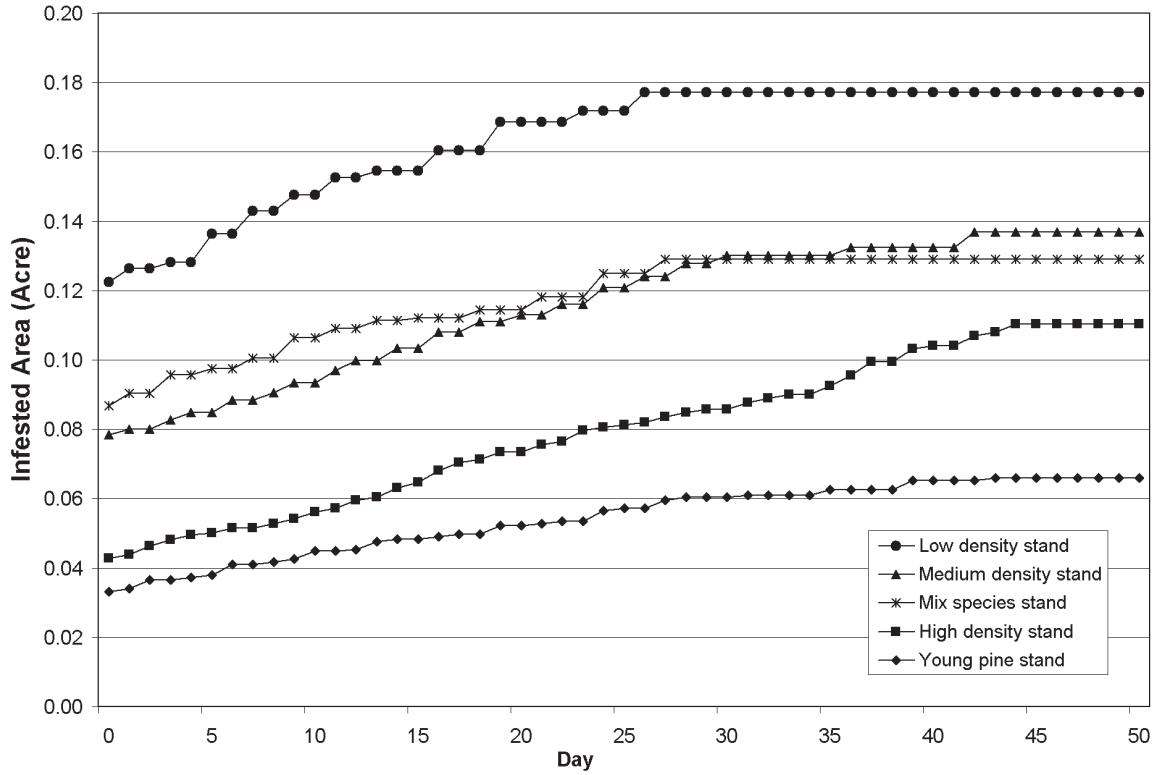

Figure 5—Comparison of the infested area among five different silvicultural treatments during 50-day period of SPB infestation.

CONCLUSION

According to the flexible and various representative styles of three-dimensional landscape visualizations, we can simulate time-series visualizations to compare influences of disturbance on different forest stands, monitor the infestation spot growth as three-dimensional visual animation for an instant short-term outbreak, or evaluate the response and efficiency from different management strategies on three-dimensional visual landscape panoramas.

In a summary, silvicultural treatments undoubtedly can modify the impacts from SPB infestations, especially the thinning and species mixture strategies. In this study, we emphasize the following conclusion—GIS-based visualizations indeed can be a comprehensive communication media to simplify the complicated information, and to provide multiscale visualizations with diverse spatial and temporal dimensions, and improve the representation and understandability for different decisionmakers with diverse backgrounds.

Finally, in the future, we would aim to apply the GIS-based spot growth model on the more practical scenarios to predict the trends of spot growth and improve the comparisons of simulated spot growth from different silvicultural treatments in real stand situations. Furthermore, in addition to simulating the instant outbreaks from SPB infestation among different silvicultural treatments, we also intend to link our model with other expert ecological prediction models and more available GIS database to predict the future patterns after SPB infestations in the long-term effects. Eventually, multispatial and temporal three-dimensional visualizations would be expected to improve the SPB decisionmaking process by combining the GIS-based spot growth model and visual simulator.

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LITERATURE CITED

3D Nature. 2002. Using VNS (manual). 3D Nature, LLC. Arvada, CO. http://3dnature.com/. [Date accessed: January 29, 2009].

Almo, F. 2006. Principles and methods in landscape ecology: toward a science of landscape. London, UK: Dordrecht: Springer. 412 p.

Chou, C.-Y.; Hedden, R.L.; Song, B.; Williams, T.M. 2008. The simulation of southern pine beetle spot growth in loblolly pine stands. In: Bettinger, P.; Merry, K.; Fei, S. [and others], eds. Proceedings of the 6th southern forestry and natural resources GIS conference. Athens, GA: University of Georgia, Warnell School of Forestry and Natural Resources: 178–195. http://soforgis.net/2008/presentations/178_Chou.pdf. [Date accessed: January 29, 2009].

Clarke, S. 2001. Review of the operational IPM program for the southern pine beetle. Integrated Pest Management Reviews. 6: 293–301.

Coster, J.E. 1981. Developing integrated management strategies. In: Thatcher, R.C.; Searcy, J.L.; Coster, J.E.; Hertel, G.D., eds. The southern pine beetle. Tech. Bull. 1631. Washington, DC: U.S. Department of Agriculture Forest Service, Expanded Southern Pine Beetle Research and Application Program: 195–203.

Edmonds, R.L.; Agee, J.K.; Gara, R.I. 2000. Forest health and protection. Boston: McGraw Hill Co., Inc. 630 p.

Hedden, R.L. 1978. The need for intensive forest management to reduce southern pine beetle activity in east Texas. Southern Journal of Applied Forestry. 2: 19–22.

Hedden, R.L. 1985. Simulation of southern pine beetle-associated timber loss using CLEMBEETLE. In: Branham, S.J.; Thatcher, R.C., eds. Integrated pest management research symposium: the Proceedings. Gen. Tech. Rep. SO-56. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 288–291.

Hedden, R.L.; Billings, R.F. 1979. Southern pine beetle: factors influencing the growth and decline of summer infestations in east Texas. Forest Science. 25: 547–566.

Martell, D.L. 2001. Forest fire management. In: Johnson, E.A.; Miyanishi, K., eds. Forest fires: behavior and ecological effects. San Diego: Academic Press: 527–583.

McGaughey, R.J. 1998. Techniques for visualizing the appearance of forestry operations. Journal of Forestry. 96(6): 9–14.

Moore, M.M.; Covington, W.W.; Fulé, P.Z. 1999. Reference conditions and ecological restoration: a southwestern ponderosa pine perspective. Ecological Applications. 9: 1266–1277.

Oliver, C.D.; Ferguson, D.E.; Harvey, A.E. [and others]. 1994. Managing ecosystems for forest health: an approach and the effects on uses and values. In: Sampson, R.N.; Adams, D.L., eds. Assessing forest ecosystem health in the inland West. Washington, DC: Forest Policy Center: 113–133.

Orland, B. 1994. Visualization techniques for incorporation in forest planning Geographic Information Systems. Landscape and Urban Planning. 30: 83–97.

Pollet, J.; Omi, P.N. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire. 11: 1–10.

Price, T.S.; Doggett, C.; Pye, J.M.; Smith, B. 1998. A history of southern pine beetle outbreaks in the Southeastern United States. Macon, GA: Georgia Forestry Commission. 72 p.

Pyne, S.J.; Andrews, P.L.; Laven, R.D. 1996. Introduction to wildland fire. 2nd ed. New York: John Wiley. 769 p.

Sheppard, S.R.J.; Salter, J. 2004. The role of visualization in forest planning. In: Burley, J.; Evans, J.; Youngquist, J.A., eds. Encyclopedia of forest sciences. Oxford, UK: Academic Press: 486–498.

Song, B.; Wang, X.; Williams, T.M. [and others]. 2006. Visualizing a landscape, its changes, and driving processes. In: Chen, J.; Saunders, S.C.; Brosofske, K.D.; Crow, T.R., eds. Ecology of hierarchical landscapes: from theory to application. New York: Nova Science Publishers, Inc.: 167–191.

Stark, R.W.; Waters, W.E.; Wood, D.L. 1985. Summary. In: Waters, W.E.; Stark, R.W.; Wood, D.L., eds. Integrated pest management in pine-bark beetle ecosystems. New York: John Wiley: 91–202.

Stephen, F.M.; Lih, M.P. 1985. A *Dendroctonus frontalis* infestation growth model: organization, refinement, and utilization. In: Branham, S.J.; Thatcher, R.C., eds. The proceedings of integrated pest management research symposium. Gen. Tech. Rep. SO-56. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 186–194.

Stephens, S.L. 1998. Evaluation of the effects of silvicultural and fuels treatments on potential fire behavior in Sierra Nevada mixed-conifer forests. Forest Ecology and Management. 105: 21–35.

Stephens, S.L.; Ruth, L.W. 2005. Federal forest-fire policy in the United States. Ecological Applications. 15(2): 532–542.

Vité, J.P. 1990. Present and future use of semiochemicals in pest management of bark beetles. Journal of Chemical Ecology. 16: 3031–3041.

Waters, W.E. 1985. Monitoring bark beetle population and beetle-caused damage. In: Waters, W.E.; Stark, R.W.; Wood, D.L., eds. Integrated pest management in pine-bark beetle ecosystems. New York: John Wiley: 141–176.

FLOODING EFFECTS ON STAND DEVELOPMENT IN CYPRESS-TUPELO

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Abstract—The effects of inundation on growth of cypress (*Taxodium* spp.) and tupelo (*Nyssa* spp.) trees have been extensively researched, but conclusions are often complicated by attendant effects on stand development. Flooding affects development of cypress-tupelo stands by limiting seedling germination and survival, truncating species richness, and reducing site quality. Persistence of the cypress-tupelo type therefore depends on flood stress sufficient to prevent establishment of other species, and sufficient stability of hydrologic regime to prevent mortality. This research investigated the role of flooding stress in controlling stand development in a pair of natural bald cypress (*T. distichum*)-water tupelo (*N. aquatica*) stands in Louisiana. Both stands have been at high enough density to experience self-thinning during the duration of the measurements, 1980 to 2005. Bald cypress is establishing dominance in both stands because of crown breakage in water tupelo, but flooding stress itself does not appear to be favoring one species over another. The most obvious effect of flood stress on stand development is to slow the rate of growth and self-thinning.

INTRODUCTION

There are presently about 342 000 ha of second-growth, even-aged bald cypress [*Taxodium distichum* (L.) Rich]-water tupelo (*Nyssa aquatica* L.) forests in coastal Louisiana (Chambers and others 2005). Most of these stands originated after clearcut logging from about 1880 to 1930; much of the area regenerated naturally then received little subsequent attention (Conner and Toliver 1990). Commercial timber management in some second-growth stands is attractive because many stands now consist of sawtimber-sized trees (Williston and others 1980). In addition to concerns about the role of forest management in these complex wetlands that have multiple ecosystem services subject to degradation from logging (Aust and others 2006, Shepard 1994), there is also a paucity of information on basic silvical processes such as regeneration, intraspecific competition, self-thinning, and responses to stand manipulations that are needed for development of appropriate management strategies.

Perhaps the most salient property of cypress-tupelo swamps is that they occupy sites with the highest degree of stress from flooding of any forest type in the region. Flooding effects on stand initiation and intraspecific competition have been extensively researched. Flooding delays or prevents germination of most species, and tolerance varies across species (Hosner 1957). Inundation of seedlings similarly reduces species diversity because many species cannot survive more than 2 weeks of inundation (Hosner 1960). Timing of flood events interacts with germination times to stochastically control seedling establishment (Jones and Sharitz 1998; Jones and others 1994, 1997). Flooding stress can also eliminate many species from the stand by causing mortality long after establishment (Broadfoot and Williston 1973, Conner and others 2002, Kozlowski 2002). All these processes reduce competition with *Taxodium* and *Nyssa* by other species less tolerant of flooding (Eggler and Moore 1961, Young and others 1995).

Less is known about the role that flood stress plays in the development of cypress-tupelo stands once they reach the self-thinning stage. Rapid changes in hydrological regime that cause extensive mortality (e.g., Eggler and Moore 1961, Harms and others 1980) are obvious drivers of stand development, but differences in stand development among sites with relatively stable hydrological regimes have been more difficult to explore. For example, although there has been extensive work to understand differences between cypress and tupelo in regeneration (e.g., Effler and Goyer 2006), the role of competition and flood stress on species composition and canopy structure is less certain. Dicke and Toliver (1990) compared 5-year growth rates and mortality in two sites with differing flood regimes and concluded that bald cypress outcompetes water tupelo in a seasonally flooded site, whereas continuous flooding favors neither species.

The archetypal cypress-tupelo stand has experienced sufficient flood stress to eliminate nearly all other overstory species. However, variation in hydrological regime and resulting flood stresses can threaten these stands. *Taxodium* (and to a lesser extent *Nyssa*) are very long-lived individuals that can persist even when the site hydrological regime has changed and is no longer suitable for regeneration (Devall 1998). Many cypress-tupelo swamps in the Delta of the Mississippi River now occupy sites where hydrological regimes are changing rapidly (Conner and Brody 1989). In addition to observed reduction in regeneration (Conner and Toliver 1990, Conner and others 1986), there are also indications that mortality and decreased growth of existing trees are widespread conditions across the region (Chambers and others 2005, Keim and others 2006).

The objective of this work is to identify how flood stress affects stand development in cypress-tupelo. We test the hypotheses that (1) flooding slows development of structure in established cypress-tupelo stands but does not affect the nature of the self-thinning process, and (2) flooding does not give a competitive advantage to either *Nyssa* or *Taxodium*. To make these tests, we use stand density as a measure of competition (Jack and Long 1996, Long 1985). We track development of two stands through stand density space,

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and employ established competition theory to address the hypotheses.

METHODS

Study Sites

The study area is a pair of stands in the Atchafalaya River Basin Floodway in southern Louisiana. The stands are in the same large area of swamp but differ in their hydrological condition. The Pigeon site, which was termed "seasonally flooded" by Dicke and Toliver (1990), is slightly drier than the Mallet site, which was termed "continuously flooded" by Dicke and Toliver (1990). Pigeon is slightly higher elevation and is on the lower elevation portion of the natural levee of Bayou Pigeon. Mallet is on the crest of the natural levee of Bayou Mallet. Thus, Pigeon receives only backwater flooding and experiences an average of 116 unflooded days during the growing season. In contrast, Mallet is directly hydrologically connected to the adjacent bayou and experiences an average of 35 unflooded days during the growing season (Keim and others 2006).

Increment cores and historical cruise data indicate trees at Pigeon were established about 1917 after a clearcut of the primary bald cypress-water tupelo forest, and we assume the origin of the stand at Mallet is similar. The stands were first measured in 1980 and selected for similar stand structure in terms of basal area and species composition (Dicke and Toliver 1988, 1990; Prenger 1985. The sites were selected for use in a thinning study, but data only from unthinned plots were used by Dicke and Toliver (1990) and by us to infer natural stand dynamics in contrasting hydrological regimes.

Data

Twelve 0.05-ha circular plots were originally established at each site. Nine of the plots at Pigeon were thinned and three remained as control, which we used for this study. The control plots were randomly assigned within a grid of plots, interspersed with thinned plots. Trees cut from the thinned plots were felled in place but not removed from the site. There was no thinning at Mallet, but only four of the plots could be located in 2005, so they were the only ones included in this study. Differences between our results and those presented by Dicke and Toliver (1990) are because our analyses were restricted to a subset of the plots they were able to use. All trees >3.8 cm diameter within the plots were permanently marked with aluminum tags and unique identification numbers affixed to nails driven at the point of diameter measurement in 1980, which was 50 cm above the maximum extent of stem buttressing, i.e., normal diameter. Tree diameters in both stands were measured in 1980, 1984, 1986, and 2005 using diameter tape (1980, 1984, 2005), calipers (1986 only), or Wheeler's Pentaprism Caliper (2005 only).

In 2005, we could not repeat the measurement of diameter at the marked points because the height of buttressing had increased in some trees between 1980 and 2005. Parresol and Hotvedt (1990) recommended standardizing diameter measurement of bald cypress at 3 m (d3) above the ground. Therefore, we measured diameter both at the location of normal diameter in 1980 and at 3 m above the ground in 2005. To compare diameter measurements across time periods, we used a stem profile equation in conjunction with d3 or normal diameter to estimate diameter at breast height (d.b.h.) as if stem buttressing were not present (d.b.h.0). We used a polynomial curve to interpolate between two local stem profile equations (Hotvedt and others 1985, Parresol and others 1987) and extrapolate stem profile from above the buttressing, thereby estimating expected stem form in the absence of buttressing:

$$d_h/D = 0.40(h/H)^3 - 1.51(h/H)^2 + 2.06(h/H) \qquad (1)$$

where
- H = total tree height
- h = height of diameter measurement as distance from top of the tree
- D = reference diameter
- d_h = diameter measured at h

To estimate d.b.h. in the absence of buttresses, d.b.h.0, we calculated D from field measurements of d3, h, and H using equation (1), then solved equation (2) for d.b.h. at $h = H - 1.37$ m. We used d.b.h.0 as the basis for all diameter analyses.

Tree heights at Pigeon were measured using a clinometer (1980, 1984, 1986) or Haglöf Vertex hypsometer and Criterion laser height finder (2005). Tree height data for Mallet before 2005 were lost, but Dicke and Toliver (1990) published stand-average data by species.

To estimate volume of each stem in 2005, we used the equation of Hotvedt and others (1985) with coefficients modified for S.I. units:

$$V = 0.0001063(\mathrm{d3})^{1.7876}(H)^{0.9522} \qquad (2)$$

where
- V (m^3) = total volume inside bark
- d3 (cm) = diameter measured outside bark at 3 m
- H = total tree height (m)

To estimate volume of Mallet trees prior to 2005, we applied a regression of volume on normal diameter (dn) parameterized using 2005 measurements ($R^2 = 0.83$):

$$V = 0.0010462(\mathrm{dn})^2 - 0.0089963(\mathrm{dn}) \qquad (3)$$

Equations 2 and 3 were developed locally for bald cypress, but we assumed they held for water tupelo as well.

Analyses: Stand Density

To quantify density and infer competition, we calculated stand density index (SDI) for each stand and each measurement period:

$$\mathrm{SDI} = \sum n_i(d_i/25)^{1.6} \qquad (4)$$

where

n = the number of trees (per ha) of species i

d = quadratic mean diameter (cm) of trees of species i

This form of SDI assumes the contributions of each species to stand density are additive and allows estimation of competition in mixed-species stands (Ducey and Larson 2003, Williams 2003). The SDI has been shown to be mostly independent of site quality (Jack and Long 1996), so it is an appropriate tool for evaluating effects of stress on stand dynamics. The ratio of SDI to the maximum SDI observed for that species (or mixture of species) is a measure of relative density (RD) of that stand. As rules of thumb, crown closure begins near RD = 0.15, full site occupancy near RD = 0.35 and self-thinning begins near RD = 0.55 (Drew and Flewelling 1979).

Woodall and others (2005) applied theoretical relationships between stand density and tree stem mechanical properties proposed by Dean and Baldwin (1996), by applying equation 4 to U.S. Forest Service, Forest Inventory and Analysis data across the United States for multiple forest types. Their result is a broadly applicable expected maximum stand density, $SDI_{99} = 2057.3 - 2098.6(SG_g)$, where SDI_{99} is the 99th percentile of observed densities (s.i. units), and SG_g is the mean of the green specific gravity of wood for each tree in the stand. We applied the Woodall equation to estimate the relative density of the two study stands for each measurement period and compared RD to expected thresholds to infer competition processes in the stands. The maximum stand density for bald cypress ($SG_g = 0.42$) is 1176, for water tupelo ($SG_g = 0.46$) is 1092, so SDI_{99} for a stand of equal proportions bald cypress and water tupelo is 1134.

RESULTS

In 1980 at Pigeon, bald cypress were 81 percent of the trees, 83 percent of the basal area, 85 percent of the volume, and 83 percent of the stand density (table 1). By 2005, bald cypress were 77 percent of the trees, 83 percent of the basal area, 87 percent of the volume, and 82 percent of the stand density.

In 1980 at Mallet, bald cypress were 65 percent of the trees, 75 percent of the basal area, 75 percent of the volume, and 73 percent of the stand density. By 2005, bald cypress were 70 percent of the trees, 83 percent of the basal area, 87 percent of the volume, and 81 percent of the stand density.

There was mortality at Pigeon of 39 percent of the trees present in 1980, 96 percent of which were smaller diameter trees in intermediate or suppressed crown classes in 1980 (fig. 1). Only 11 percent of the mortality was of tupelo (22 percent mortality of the original trees, compared to 43 percent for bald cypress), all of which were intermediate or suppressed. Only 5 percent of trees in dominant and codominant crown classes died, none of which were tupelo.

There was mortality at Mallet of 15 percent of the trees present in 1980. Crown classes were not recorded in 1980, but mortality was mostly of the small trees; the average 1980 diameter of trees that died was 23.2 cm, compared to 27.2 cm for trees that survived. The largest tree in the study—a tupelo 58 cm diameter in 1986—died between 1986 and 2005. Omitting this large tree, the average diameter of trees that died was 21.8 cm. Sixty percent of the mortality was of tupelo (27 percent mortality of the original trees, compared to 9 percent for bald cypress). The average diameter of bald cypress trees that died was 24.0 cm, compared to 22.7 cm for tupelo (20.2 cm omitting the single large tree).

Volume of tupelo decreased in both stands from 1980 to 2005. At Pigeon, volume lost to tupelo mortality (10 m^3/ha) was 9 percent of the total volume lost to mortality, which approximates the 11-percent mortality of tupelo by number of trees. The net gain in stand-level volume of bald cypress was by concentration of growth in large trees, whereas there are few large tupelo trees. Thus, although mortality of tupelo was less than bald cypress, the surviving tupelo are mostly in subordinate crown classes.

Table 1—Characteristics of two baldcypress-water tupelo stands in Louisiana

		Trees		Basal area		Volume		SDI		Average height	
Site	Species	1980	2005	1980	2005	1980	2005	1980	2005	1980	2005
		------ per ha ------		*------ m^2/ha ------*		*------ m^3/ha ------*		*----- s.i. units -----*		*-------- m --------*	
Pigeon	BC	1320	753	44.4	42.1	483	537	976	836	19.4	24.2
(less flood)	WT	300	233	8.8	8.5	86	81	199	183	17.6	17.9
	All	1620	986	53.2	50.6	569	618	1175	1019	19.1	22.7
Mallet	BC	707	480	35.6	38.1	430	423	681	705	21.4	21.9
(more flood)	WT	373	205	11.8	7.9	140	63	247	169	17.6	13.9
	All	1080	685	47.4	46.0	570	486	928	874	19.9	19.5

SDI = stand density index; BC = bald cypress; WT = water tupelo.

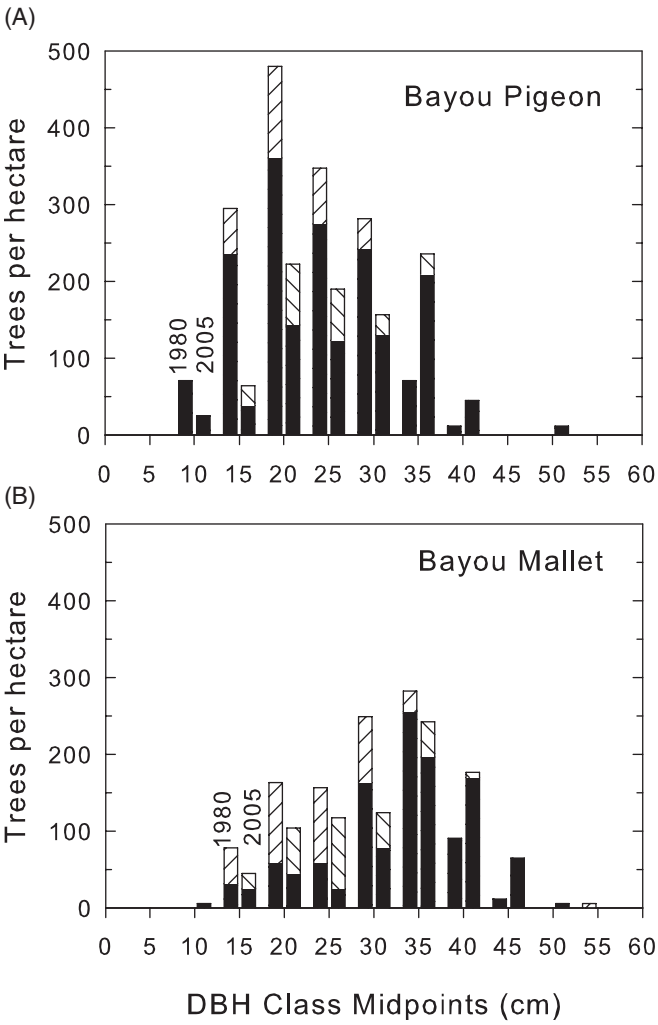

Figure 1—Diameter distributions of two cypress-tupelo stands in Louisiana. (A) Bayou Pigeon and (B) Bayou Mallet. Solid portions of bars are bald cypress and hatched portions of bars are water tupelo.

Volume of both species decreased from 1980 to 2005 at Mallet. Volume lost to tupelo mortality (43 m^3/ha) was 61 percent of the total volume lost to mortality, which approximates the 60-percent mortality of tupelo by number of trees. Excluding the single large tupelo that died, the volume lost to tupelo mortality (27 m^3/ha) was 49 percent of the total; this reflects the fact that bald cypress mortality was of trees that were slightly larger than tupelo mortality.

The average height of bald cypress has been more than tupelo throughout 1980 to 2005 because tupelo trees were more likely to be in intermediate or suppressed canopy positions. Average height of bald cypress was 1 m more at Mallet than at Pigeon in 1980, but by 2005 the bald cypress at Pigeon were 2.3 m taller than those at Mallet. Height of the tallest 15 percent of bald cypress trees, i.e., dominant and strong codominants only, at Pigeon was 24.9 m in 1980 and 30.0 m in 2005. Height of the tallest 15 percent of bald cypress at Mallet in 1980 is unknown, but in 2005 it was 26.1 m.

The differences in tree height between 1980 and 2005 were strongly affected by crown dieback and breakage, especially for tupelo. Sixty-six percent of tupelo trees at Pigeon were shorter in 2005 than in 1980, and 34 percent were more than 3 m shorter. In contrast, 20 percent of bald cypress trees at Pigeon were shorter in 2005 than in 1980, and 6 percent were more than 3 m shorter. Thus, dieback and breakage occurred throughout the study, but large breakage events (loss of 3 m or more) were concentrated in the 1986 to 2005 period. During the first period of the study, from 1980 to 1983, 71 percent of tupelo and 46 percent of cypress trees decreased in height, but only 10 percent of tupelo and <1 percent of cypress decreased in height by more than 3 m. From 1986 to 2005, 69 percent of tupelo and 31 percent of cypress trees decreased in height, and 29 percent of tupelo and 6 percent of cypress decreased in height by more than 3 m. Nearly all (95 percent) tupelo trees decreased in height during at least one measurement period, and most (84 percent) cypress trees did also. However, 40 percent of tupelo trees experienced loss of at least 3 m of height in at least one study period, but only 5 percent of cypress trees experienced such a large loss. By 2005, almost all water tupelo showed evidence of past crown damage; this was true for both Pigeon and Mallet.

Overall, Pigeon had a RD = 1.01 in 1980 and RD = 0.88 in 2005. Mallet had a RD = 0.80 in 1980 and RD = 0.75 in 2005 (fig. 2). Because the threshold density for self-thinning is generally about RD = 0.55, both stands have been dense enough to experience self-thinning for the entire duration of the study. Although Pigeon has been denser than Mallet since at least 1980, the temporal changes in RD between the two stands have been nearly identical.

DISCUSSION

Differences in the heights of dominant trees in the two stands is evidence that Pigeon (less flooding) is a more productive stand than is Mallet (more flooding). In 2005, the dominants at Pigeon were 3.9 m taller than dominants at Mallet. Although data are not available for the height of dominant trees in Mallet in 1980, we can estimate from diameter distributions (fig. 1) and mean heights of the stands at that time (table 1) that dominants there were likely taller than at Pigeon. Based on height growth history and the general negative correlation between flood stress and productivity, the most likely conclusion is that Pigeon is a younger stand and more productive site.

The stand densities at both sites were clearly high enough to cause mortality from self-thinning. The loss of density at both sites has been at approximately on the same trajectory, but at a slower rate at Mallet. Mortality causes episodic losses of density that can only be replaced by continued growth of the residual trees, so stand development typically follows a stochastic sawtooth pathway in density space (Long 1985), and predicting future stand development from recent history

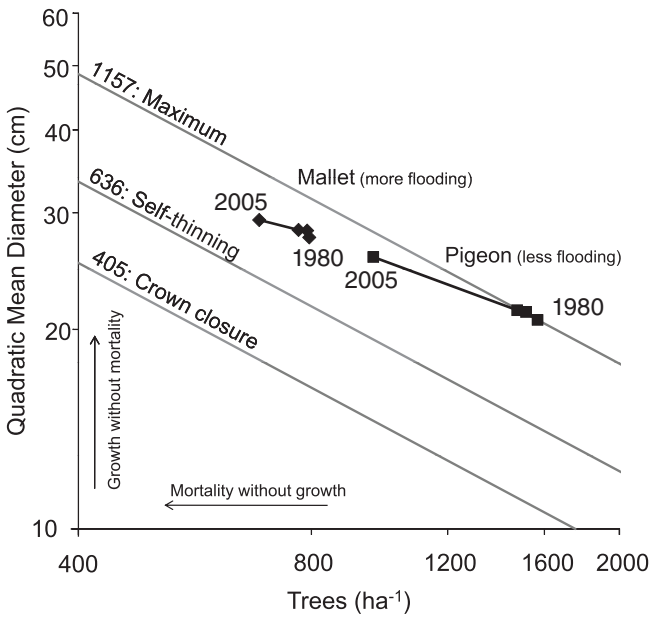

Figure 2—Density diagram for two cypress-tupelo stands in Louisiana. Gray lines indicate important threshold densities for a stand of evenly mixed bald cypress and water tupelo.

is therefore difficult. However, the slower tree growth at Mallet means that remaining trees will be slower to capture resources made available by mortality, and also slower to come into competition with other residual trees.

Although the stands were chosen for similar initial basal area, the trees at Mallet were larger. It is possible this stand is older, had a lower initial stand density, or that it was initially a more productive site than Pigeon. However, stand volume was similar in 1980, so it is also possible that stand structure was simply different. For example, the current canopy trees may have established dominance sooner at Mallet than at Pigeon, so that growth was concentrated on larger trees for a longer period of time compared to Pigeon.

The long-term prospect for tupelo at Pigeon appears to be continued loss of volume, with a few codominant trees remaining in the stand indefinitely. To date, most mortality in this stand has been of intermediate and suppressed trees of both species. If there were to be mortality of larger trees that created openings, it is not clear whether surviving tupelo trees in subordinate canopy positions would be able to occupy the new growing space, in part because of their generally poor condition and broken tops. The situation at Mallet is slightly different but the long-term prospects appear similar. There were originally more tupelo in that stand, and fewer suppressed trees than in Pigeon, but there has been mortality of large tupelo at Mallet. The loss of larger trees, in combination with broken tops and generally poor condition of many surviving trees, has resulted in loss of more than half the volume of tupelo. Whereas in 1986 it appeared that tupelo would continue to be near equal to bald cypress at Mallet (Dicke and Toliver 1990), it now appears that tupelo is losing dominance to bald cypress in much the same way as at Pigeon.

A recent complicating factor in the development of these stands may be disturbance by Hurricane Andrew in 1992. We have no observations of effects on our study stands, but tree damage was ubiquitous throughout the Atchafalaya Basin (Doyle and others 1995). Cypress-tupelo stands were the least damaged stand type in that storm (Doyle and others 1995), as is regionally typical (Loope and Duever 1994), but most tupelo trees in our study stands and in the surrounding area have broken tops and show signs of general decline. There are few signs of significant damage to bald cypress and only isolated breakage in the study stands, so it is possible the hurricane favored dominance by bald cypress. Rates of tupelo mortality were the same, e.g., 4 percent of eventual total mortality per year at Mallet, for the prehurricane 1980 to 1986 period and for the 1986 to 2005 period which included the hurricane; however, major crown breakage in tupelo was concentrated in the 1986 to 2000 period. Hurricanes Danny (1985), Juan (1985), Lili (2002), and Tropical Storm Beryl (1988) also passed near the research stands, but we have no information on damage that may have occurred during these storms. The degree to which tropical cyclones may affect stand development and species composition in general is unknown but likely varies between coastal and inland swamps. If tupelo is more likely to experience crown damage from windstorms, as it appears was the case in this study, tupelo would be at a competitive disadvantage in coastal stands.

CONCLUSIONS

The differences in flood stress between the two cypress-tupelo stands did not apparently fundamentally alter competition within the stands, and both stands are developing approximately according to expectations based on stand-density relationships for other species. However, the slower rate of growth appears to be slowing the rate of development at the site experiencing more flooding stress. We conclude that flooding stress does not fundamentally alter density-dependent stand development in cypress-tupelo. There is also little evidence that flooding stress itself is responsible for the apparent competitive advantage that bald cypress has in the study stands, but that water tupelo is decreasing in importance because of greater susceptibility to crown breakage in tropical storms.

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LITERATURE CITED

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

Broadfoot, W.M.; Williston, H.L. 1973. Flooding effects on southern forests. Journal of Forestry. 71: 584–587.

Chambers, J.L.; Conner, W.H.; Day, J.W., Jr. [and others]. 2005. Conservation, protection and utilization of Louisiana's coastal wetland forests. Final report to the Governor of Louisiana from the Science Working Group on Coastal Wetland Forest Conservation and Use. 102 p. http://www.coastalforestswg.lsu.edu/TheFinalReport.pdf. [Date accessed: July 26, 2011].

Conner, W.H.; Brody, M. 1989. Rising water levels and the future of southeastern Louisiana swamp forests. Estuaries. 12: 318–323.

Conner, W.H.; Day, J.W. Jr. 1988. Rising water levels in coastal Louisiana: implicationa for two coastal forested wetland areas in Louisiana. Journal of Coastal Research. 4: 589–596.

Conner, W.H.; Mihalia, I.; Wolfe, F. 2002. Tree community structure and changes from 1987 to 1999 in three Louisiana and three South Carolina forested wetlands. Wetlands. 22: 58–70.

Conner, W.H.; Toliver, J.R. 1990. Long-term trends in the bald-cypress (*Taxodium distichum*) resource in Louisiana (U.S.A.). Forest Ecology and Management. 33/34: 543–557.

Conner, W.H.; Toliver, J.R.; Sklar, F.H. 1986. Natural regeneration of baldcypress (*Taxodium distichum* (L.) Rich.) in a Louisiana swamp. Forest Ecology and Management. 14: 305–317.

Dean, T.J.; Baldwin, V.C. 1996. The relationship between Reineke's stand-density index and physical stem mechanics. Forest Ecology and Management. 81: 25–34.

Devall, M.S. 1998. An interim old-growth definition for cypress-tupelo communities in the Southeast. Gen. Tech. Rep. SRS-19. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 13 p.

Dicke, S.D.; Toliver, J.R. 1988. Effects of crown thinning on baldcypress height, diameter, and volume growth. Southern Journal of Applied Forestry. 12: 252–256.

Dicke, S.D.; Toliver, J.R. 1990. Growth and development of bald-cypress/water-tupelo stands under continuous versus seasonal flooding. Forest Ecology and Management. 33/34: 523–530.

Doyle, T.W.; Keeland, B.D.; Gorham, L.E.; Johnson, D.J. 1995. Structural impact of Hurricane Andrew on the forested wetlands of the Atchafalaya Basin in south Louisiana. Journal of Coastal Research. 21: 354–364.

Drew, T.J.; Flewelling, J.W. 1979. Stand density management: an alternative approach and its applications to Douglas-fir plantations. Forest Science. 25: 518–532.

Ducey, M.J.; Larson, B.C. 2003. Is there a correct stand density index? An alternate interpretation. Western Journal of Applied Forestry. 18:179–184.

Effler, R.S.; Goyer, R.A. 2006. Baldcypress and water tupelo sapling response to multiple stress agents and reforestation implications for Louisiana swamps. Forest Ecology and Management. 226: 330–340.

Eggler, W.A.; Moore, W.G. 1961. The vegetation of Lake Chicot, Louisiana, after eighteen years of impoundment. Southwestern Naturalist. 6: 75–183.

Harms, W.R.; Schreuder, H.T.; Hook, D.D.; Brown, C.L. 1980. The effects of flooding on the swamp forest in Lake Ocklawaha, Florida. Ecology. 61: 1412–1421.

Hosner, J.F. 1957. Effects of water upon the seed germination of bottomland trees. Forest Science. 3: 67–70.

Hosner, J.F. 1960. Relative tolerance to complete inundation of fourteen bottomland tree species. Forest Science. 6: 246–251.

Hotvedt, J.E.; Cao, Q.V.; Parresol, B.R. 1985. Tree-volume and stem-profile functions for baldcypress. Southern Journal of Applied Forestry. 9: 227–232.

Jack, S.B.; Long, J.N. 1996. Linkages between silviculture and ecology: an analysis of density management diagrams. Forest Ecology and Management. 86: 205–220.

Jones, R.H.; Allen, B.P.; Sharitz, R.R. 1997. Why do early-emerging tree seedlings have survival advantages? A test using *Acer rubrum* (Aceraceae). American Journal of Botany. 84: 1714–1718.

Jones, R.H.; Sharitz, R.R. 1998. Survival and growth of woody plant seedlings in the understorey of floodplain forests in South Carolina. Journal of Ecology. 86: 574–587.

Jones, R.H.; Sharitz, R.R.; Dixon, P.M. [and others]. 1994. Woody plant regeneration in four floodplain forests. Ecological Monographs. 64: 345–367.

Keim, R.F.; Chambers, J.L.; Hughes, M.S. [and others]. 2006. Ecological consequences of changing hydrological conditions in wetland forests of coastal Louisiana. In: Xu, Y.J.; Singh, V.P., eds. Coastal environment and water quality. Highlands Ranch, CO: Water Resource Publications: 383–395.

Kozlowski, T.T. 2002. Physiological-ecological impacts of flooding on riparian forest ecosystems. Wetlands. 22: 550–561.

Long, J.N. 1985. A practical approach to density management. Forestry Chronicle. 61: 23–37.

Loope, L.; Duever, M. 1994. Hurricane impact on uplands and freshwater swamp forest. Bioscience. 44: 238–246.

Parresol, B.R.; Hotvedt, J.E. 1990. Diameter measurement in baldcypress. Forest Ecology and Management. 33/34: 509–515.

Parresol, B.R.; Hotvedt, J.E.; Cao, Q.V. 1987. A volume and taper prediction system for baldcypress. Canadian Journal of Forest Research. 17: 250–259.

Prenger, R.S. 1985. Response of a second-growth natural stand of baldcypress trees (*Taxodium distichum* (L.) Rich.) to various intensities of thinning. Baton Rouge, LA: Louisiana State University. 52 p. M.S. thesis.

Shepard, J.P. 1994. Effects of forest management on surface water quality in wetland forests. Wetlands. 14: 18–26.

Thomson, D.A.; Schaffer, G.P.; McCorquodale, J.A. 2002. A potential interaction between sea-level rise and global warming: implications for coastal stability on the Mississippi River Deltaic Plain. Global Planetary Change. 32: 49–59.

Toliver, J.R.; Dicke, S.G.; Prenger, R.S. 1987. Response of a second-growth natural stand of baldcypress to various intensities of thinning. In: Phillips, D.R., comp. Proceedings of the fouth biennial southern silvicultural research conference; 1986 Nov. 4-6, Atlanta, GA. Gen. Tech. Rep. SO-54. Asheville, N.C.: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station.: 462–465.

Williams, R.A. 2003. Use of stand density index as an alternative to stocking percent in upland hardwoods. Northern Journal of Applied Forestry. 20: 137–142.

Williston, H.L.; Shropshire, F.W.; Balmer, W.E. 1980. Cypress management: a forgotten opportunity. For. Rep. SA-FR 8. Atlanta: U.S. Department of Agriculture Forest Service, Southeastern Area State and Private Forestry. 8 p.

Woodall, C.W.; Miles, P.D.; Vissage, J.S. 2005.Determining maximum stand density index in mixed species stands for strategic-scale stocking assessments. Forest Ecology and Management. 216:367–377.

Young, P.J.; Keeland, B.D.; Sharitz, R.R. 1995. Growth response of baldcypress [*Taxodium distichum* (L.) Rich.] to an altered hydrological regime. American Midland Naturalist. 133: 206–212.

INCREASING POPULATIONS OF KENTUCKY LADY'S SLIPPER ORCHID ON THE KISATCHIE NATIONAL FOREST: SEEDLING PRODUCTION AND OUTPLANTING TRIALS

James Barnett, Shannan Sharp, Kevin Allen, and Andy Scott[1]

Abstract—The Kentucky lady's slipper orchid (*Cypripedium kentuckiense* C.F. Reed) is a tall, stately perennial plant with the largest flowers of any *Cypripedium* known. Its range includes much of the Southeastern United States, though it is rare throughout its range due to specific edaphic and climatic habitat requirements. In Louisiana, a few plants are located on four sites within the 600,000-acre Kisatchie National Forest (KNF). This effort is to increase populations of one of the most spectacular orchids native to the region. A high school student located a flowering orchid in the KNF, caused it to be pollinated, and later collected a fertile seed pod. A collaborative effort began between KNF, Southern Research Station, and Central Louisiana Orchid Society (CLOS) to restore the orchid on appropriate sites. Grants in 2006 from the Southwest Regional Orchid Growers Association and in 2007 from the U.S. Forest Service allowed CLOS to purchase plantlets grown from the collected seed pod. A research study is now underway to develop propagation protocols and compare effects of seedling age, fungal inoculation, and depth and season of planting on establishment success.

INTRODUCTION

A project to restore one of the rarest and most spectacular orchids native to the west Gulf Coast region began with the curiosity of a high school student. Inspired by the beauty of lady's slippers in a book on orchids, Kevin Allen, then a high school student, was determined to locate the Kentucky lady's slipper (*Cypripedium kentuckiense* C.F. Reed) orchid that was known to exist in the Kisatchie National Forest (KNF). With help from KNF botanists, he located two small populations and self-pollinated plants when in flower (fig. 1). After 3 years, he obtained a seed capsule. This seed pod was sent to Spangle Creek Labs in Bovey, MN, who are specialists in *Cypripedium* seed germination. Seeds from the collection were viable and Spangle Creek Labs produced small plantlets which could be used in the effort. Kevin Allen approached the KNF to determine their interest in a restoration project. Lacking the technical expertise to grow the plantlets to plantable size, Southern Research Station (SRS) and Central Louisiana Orchid Society (CLOS) personnel were approached for assistance in the effort. Thus began a collaborative effort to increase Kentucky lady's slipper orchids on KNF (Barnett 2008).

Literature specific to *C. kentuckiense* largely accesses terminology of the species because there has been debate and confusion among plant taxonomists about its proper classification (Atwood 1984). For years it had been classified as *C. calceolus* var. *pubescens* or a natural hybrid. An isozyme study has confirmed that *C. kentuckiense* is a distinct species (Case and others 1998). Based on herbaria collections, it ranges from eastern Texas and Oklahoma to Kentucky and Virginia (Atwood 1985). However, most States where the species now occurs have designated *C. kentuckiense* as endangered, threatened, special concern, or otherwise imperiled and vulnerable to extinction (Allen and others 2004, Case and others 1998). The population continues to decline from loss of habitat and human predation (Cash 1991).

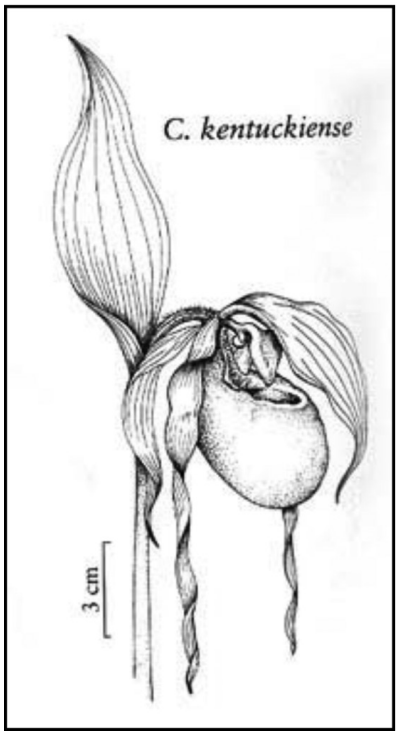

Figure 1—*Cypripedium kentuckiense* from "The Flora of North America" Volume 26. Copyright © 2000 Flora of North America Association.

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Information on the propagation and reestablishment of this terrestrial species is sparse. Orchid enthusiasts generally grow orchids of the epiphytic type in greenhouses and grow relatively few terrestrial species even in garden situations. Some research trials have been conducted with other species of the *Cypripedium* genus. Huber (2002) has successfully established *C. montanum* by mixing seeds with a carrier such as forest soil, sugar, or cracked corn prior to sowing in forest openings. The most significant reestablishment effort of a *Cypripedium* species has been the effort by Ramsey and Stewart (1998) in the United Kingdom with *C. calceolus*. They have had some limited success with direct seeding, but have studies underway to pot seedlings for planting in the wild when grown to an appropriate size.

SEED CAPSULE COLLECTION

One of the orchids self-pollinated by Kevin Allen produced a seed capsule. Seeds from either green or dry capsules can be used to produce seedlings via in vitro methods (Cullina 2004); Allen collected the capsule in its dry condition. He sent the dried fruit to Spangle Creek Labs (www.spanglecreeklabs.com) within a few weeks after collection.

SEED GERMINATION AND PLANTLET DEVELOPLMENT

Orchid seeds have essentially no stored food resources and require external sources of nutrition for germination. In nature, they obtain nutrients and energy by absorption from a symbiotic fungus which invades the seeds (Ramsey and Stewart 1998). The term symbiotic is usually used for the relationship between a fungus and orchid; however, no benefit to the fungus has been found. Germination proceeds in *in vitro* culture where nutrients are supplied in a liquid solution or gel medium. Different species of orchids, even within the same genus, require media of different composition (Steele 2007).

Both media and seeds require sterilization; otherwise, the cultures would quickly be overrun by competing organisms such as fungi and bacteria. Steele (1995, 2007) outlines procedures for sterilizing orchid seed; methods used by Spangle Creek Labs are not proprietary and can be reviewed by contacting the lab (www.spanglecreeklabs.com). The sterilized seeds are then moved to a specialized agar medium in a flask for germination and development under aseptic conditions. These flasks must remain under dark conditions for the entire germination period.

After 3 to 4 months on specialized media in flasks, the embryos begin to swell and rupture the seed coat, and a white body known as the protocorm is formed. Protocorms are transferred from the sowing flasks to fresh media when sufficient size to be handled by a forceps without damage. To minimize browning of the plants that may result from the handling, protocorms are placed at a density that allows good root development to take place with no requirement for additional transfers that might damage them (Ramsey and Stewart 1998). Flasks are kept in the dark throughout this period to encourage root rather than shoot growth. It will take up to 6 months for the plantlets to develop to a size for deflasking.

Many species of *Cypripediums* require chilling to hasten germination and shoot development (Curtis 1943). Since *C. kentuckiense* has a more southern range than many of these orchids; extent of dormancy in this species is not understood. Upon removal from the flasks, the plantlets are thoroughly rinsed in water and placed in sealed plastic storage conditions with enough water to barely cover the bottom of the container. The containers are then placed in a refrigerator for about 4 months to ensure the seedlings are properly vernalized (Steele 2007).

TRANSFER OF PLANTLETS TO TRAYS FOR GROWTH

Once problems of collecting and sowing seed and maintaining plant growth and development *in vitro* have been resolved, the grower now must transplant these seedlings successfully to a nursery environment and ultimately establish them in forest conditions. Many plants grown *in vitro* are difficult to transfer into compost-type medium as they are acclimatized to high levels of humidity.

In our study, plantlets received from Spangle Creek Labs had buds near 1 cm long and roots 4 cm or more in length (fig. 2). These were planted in trays filled with media consisting of compost, commercial potting soil (PRO-MIX®) and sand. The 200 plantlets received in late May 2006 were distributed to several growers and the potting medium, seedling culture, and environmental exposures varied by grower in an effort to determine guidelines for *C. kentuckiense* seedling production. Although initial shoot growth was excellent, survival of seedlings during the next several months was poor.

Most growers initially started plantlets in greenhouse environments, but it became apparent that summer greenhouse temperatures were excessive and mortality

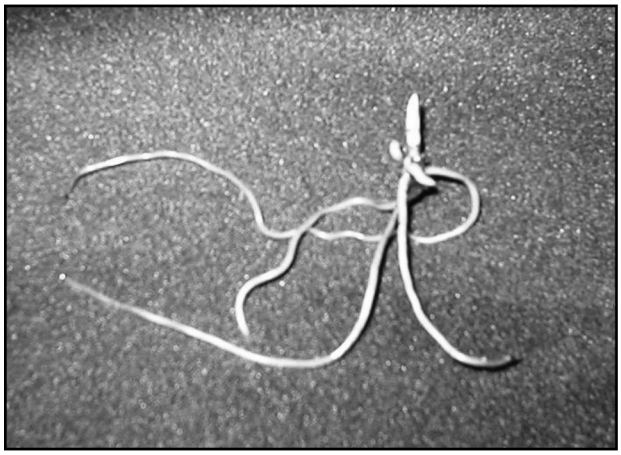

Figure 2—Plantlet of *Cypripedium kentuckiense* after 6 weeks in potting medium showing initial shoot development and root system.

or dormancy occurred. Growers then moved the plants to outside, shaded sites.

SEEDLING PRODUCTION

Based on observations through 2006, we concluded that (1) greenhouse temperatures during the summer are too high for these terrestrial orchids, (2) seedlings develop better in a dappled shade environment, and (3) a peat-sand mix worked well for plant development. Seedling shoots died early in the summer if the plants were under stress, either by high temperatures or droughty conditions. If the stress was not severe, many of the seedlings became dormant with the root system still alive. These dormant seedlings began growth again early in the next spring.

In 2007, 700 additional seedlings from the same seed source and lot were purchased. These, again, were distributed among growers. A commercial potting mix (PRO-MIX®) with sand (2 to 1 ratio) was used by all growers, and seedlings were grown in shaded outdoor environments. Percentage of seedlings alive at the end of the first-growing season improved over the previous year.

STUDY TREATMENTS IN FIELD PLANTINGS

On December 18, 2007, and March 6, 2008, seedlings grown from the 2006 and 2007 shipments of plantlets were outplanted in field trials in the Catahoula Ranger District of the KNF. Four replications of five seedlings per plot were planted along a stream near a small group of native Kentucky lady's slipper orchids. Plants that serve as indicator plants for orchid sites are American beech (*Fagus grandifolia*), eastern hophornbeam (*Ostrya virginiana*), horsesugar (*Symplocos tinctoria*), and witch hazel (*Hamamelis virginiana*) in the overstory and an abundance of poison ivy (*Rhus radicans*) and broad beechfern (*Thelypteris hexagonoptera*) in the understory (Allen and others 2004). Before planting, number of roots and shoots of each seedling was recorded (table 1).

For outplanting seedlings, we followed recommendations Cullina (2008) developed for transplanting divisions of *Cypripedium* plants. The procedure requires first working compost or duff into the planting site to a depth of 6 to 8 inches, then developing a cavity in the soil several inches deep, placing the root system so the seeding bud is near the groundline surface, and replacing soil around the seedling with the bud at the surface. The planted seedlings were covered with mulch to protect the plant and conserve moisture.

Results from the December planting

The December planting compared seedling age (seedlings from 2006 and 2007 crops). Seedlings from the 2006 crop were received from Spangle Creek Labs in late May and transplanted into trays of potting media in early June. Lack of knowledge of growing culture resulted in considerable mortality, but enough from the 200 purchased remained to plant after a second-growing season in the trays of potting mix. Seedlings from a 2007 crop were obtained in April and were transplanted into trays with a PRO-MIX®-sand mixture.

It is interesting to note that seedlings from the 2007 crop had higher numbers of roots and shoots than those from the 2006 crop (fig. 3). This suggests that holding the 2006 seedlings through 2007 in the same trays without additional nutrients reduced seedling vitality. Steele (2007) reports lack of success in growing *C. arietinum* seedlings in any sort of peat-based mix for longer than one season.

Seedling survival was measured April 1 and July 7, 2008 (table 1). Survival in April was 70 and 50 percent for seedlings

Table 1—Seedling characteristics and survival of various treatments when planted on December 18, 2007, and March 6, 2008

	Characteristics at planting		Survival	
Planting variable	Roots	Shoots	4/1/08	7/7/08
	---------- *number* ----------		---------- *percent* ----------	
	December 18, 2007, planting			
2006 crop	6.5[a]	1.0	70	25
2007 crop	8.3	1.3	50	15
	March 6, 2008, planting			
2007 unfertilized	8.4	1.3	85	15
2007 fertilized[b]	8.1	1.2	65	10

[a] The numbers represent an average of four replications of five seedlings each.

[b] Slow-release nutrients (1 teaspoon of Osmocote 19-6-12) were applied at each planting spot after each seedling was planted.

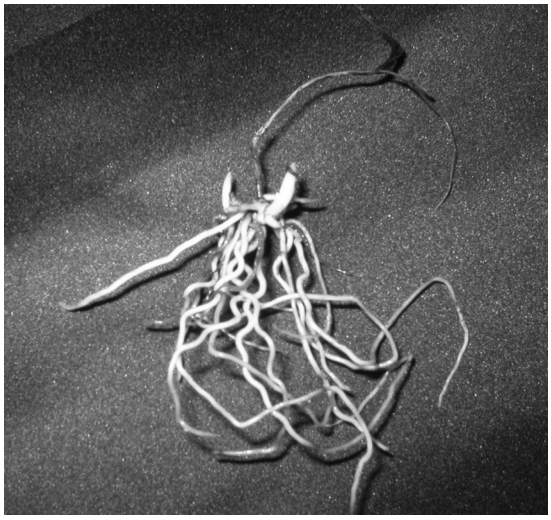

Figure 3—*Cypripedium kentuckience* seedling lifted after 8 months in potting medium and before being outplanted into the Kisatchie National Forest. Note the dead shoot retained after the seedling became dormant and the two developing shoots.

from the 2006 and 2007 crops, respectively. However, 3 months later, survival measured was only 25 and 15 percent for the same treatments. Rapid increases in seedling mortality are likely related to a combination of droughty conditions at the planting site and the shallow planting of seedling root systems. At planting, the seedlings were small with poorly developed root systems.

Results from the March planting

Variables evaluated in March were fertilization applied as the seedlings were planted—fertilized and nonfertilized. Fertilization was accomplished by scattering 1 teaspoon of Osmocote® 19-6-12 to the planting site after the seedling was planted. No data were available to guide the rate of application; but, it was believed that some level of fertilization would help seedling growth and potential flowering. Results indicated that the treatment rate, as applied, was too high and adversely affected survival (table 1).

CONCLUSIONS

This restoration effort resulted from a high school student's curiosity about one of the South's rarest and most spectacular orchids—*C. kentuckiense*. After several years' efforts in locating and pollinating flowering plants and with assistance of a *Cypripedium* specialist, plantlets became available for potential seedling development and reintroduction studies. Thus began a collaborative effort among Kevin Allen, Spangle Creek Labs, U.S. Forest Service's KNF and SRS, and the CLOS.

Quickly, it became apparent that little information exists to propagate, culture, and outplant this terrestrial orchid species. In fact, information about the entire *Cypripedium* genus is sparse. Largely by trial and error, information is being developed to provide guidelines to those interested in orchid restoration. Current trials indicate that plantlet development into appropriate seedling size and condition requires peat-sand medium that receives a low level of nutrients. Data also indicate that seedlings that perform well in the field may take 1 or more years to develop under nursery conditions.

Gains are being made in developing cultural practices appropriate for the species; however, additional studies are needed to provide meaningful guidelines for restoration. Studies are now underway to evaluate fertilization needs, seedling size, fungal inoculation, and planting depth. Another aspect of restoration is to quantify sites appropriate for reintroduction of the species. Indicator species provide some direction in site selection, but additional efforts are needed to determine soil types and pH and light conditions most suited for Kentucky lady's slipper establishment.

In an effort to provide future planting stock, Kevin Allen, now a science teacher at Captain Shreve High School is beginning an effort to refine seed germination and plantlet development technology so that seedlings can be produced in sufficient quantities and at reasonable costs so that restoration efforts may continue. This effort will be accomplished as honors chemistry projects and will be supported by grant funding from the U.S. Forest Service and orchid organizations.

LITERATURE CITED

Allen, Charles M.; Thames, Sara; Trichell, Spencer; White, Jeremy. 2004. A quantitative study of the vegetation surrounding yellow lady-slipper orchid (*Cypripedium kentuckiense*, Orchidaceae) populations at Fort Polk in West Central Louisiana. Sida Contributions to Botany. 21(1): 409–417.

Atwood, John T. 1985. The range of *Cypripedium kentuckiense*. American Orchid Society Bulletin. 54(10): 1197–1199.

Atwood, John T., Jr. 1984. In defense of *Cypripedium kentuckiense* C.F. Reed. American Orchid Society Bulletin. 53(8): 835–841.

Barnett, James. 2008. Helping the wild orchid bloom again. Forests & People. 58(2): 4–5.

Case, Martha A.; Mlodozeniec, Henry T.; Wallace, Lisa E.; Weldy, Troy W. 1998. Conservation genetics and taxonomic status of the rare Kentucky lady's slipper: *Cypripedium kentuckiense* (Orchidaceae). American Journal of Botany. 85(12): 1779–1786.

Cash, Catherine. 1991. The slipper orchids. Portland, OR: Timber Press Inc.: 184–187.

Cullina, William. 2004. Understanding orchids: an uncomplicated guide to growing the world's most exotic plants. New York: Houghton Mifflin Co. 260 p.

Cullina, William. 2008. Planting large and small yellow lady-slippers, Kentucky lady-slippers, and their hybrids. http://williamcullina.com/files/download/planting.pdf. [Date accessed: August 9].

Curtis, John T. 1943. Germination and seedling development in five species of *Cypripedium* L. American Journal of Botany. 30(3): 199–206.

Huber, Andrew G. 2002. Mountain lady's slipper (*Cypripedium montanum*): establishment from seeds in forest openings. Native Plants Journal. 3(2): 151–154.

Ramsey, Margaret M.; Stewart, Joyce. 1998. Re-establishment of the lady's slipper orchid (*Cypripedium calceolus* L.) in Britain. Botanical Journal of the Linnean Society. 126: 173–181.

Steele, William K. 1995. Growing *Cypripedium reginae* from seed. American Orchid Society Bulletin. 64: 382–391.

Steele, William K. 2007. Propagation protocol for ram's head lady's slipper (*Cypripedium arietinum*). Native Plants Journal. 8(1): 58–64.

THINNING TO IMPROVE GROWTH, BOLE QUALITY, AND FOREST HEALTH IN AN *INONOTUS HISPIDUS*-INFECTED, RED OAK-SWEETGUM STAND IN THE MISSISSIPPI DELTA: 10-YEAR RESULTS

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Abstract—A 55-year-old red oak-sweetgum (*Quercus* spp.-*Liquidambar styraciflua*) stand on the Delta National Forest in western Mississippi was subjected to a combination of low thinning and improvement cutting in 1997. Special emphasis was placed on removing all red oaks infected with *Inonotus hispidus*, a canker decay fungus that causes severe degradation and cull. Stand-level growth during the 10 years since thinning has been minimal. Thinning significantly increased diameter growth of residual trees, especially red oaks, but has not yet produced a significant increase in stand-level quadratic mean diameter. Thinning had little influence on the production of new epicormic branches on residual red oaks, but it greatly increased the number of epicormic branches on residual sweetgum trees. Because it removed all red oaks infected with *Inonotus hispidus*, thinning improved overall forest health. During the 10 years since the thinning operation, thinning has had no adverse effects on the incidence of new infections by a variety of pathogens.

INTRODUCTION

Thinnings and improvement cuttings often are used in mixed-species forests to enhance growth of residual trees and to improve both species composition and quality of the residual stand (Meadows 1996). These three goals—increased growth, improved species composition, and enhanced quality—are critically important for profitable management of southern bottomland hardwood stands for the production of high-quality sawtimber.

Thinning regulates stand density and increases diameter growth of residual trees. In general, diameter growth of residual trees increases as thinning intensity increases. However, very heavy thinning may reduce stand density to such an extent that stand growth declines to an unsatisfactory level even though growth of individual trees may be greatly enhanced. Residual stocking of very heavily thinned stands simply becomes so low that the stand is unable to utilize fully the potential productivity of the site. For example, thinning to a residual stocking level of 33 percent in a relatively young water oak (*Quercus nigra*) plantation created a severely understocked condition that likely will depress stand growth for many years (Meadows and Goelz 2001). Based on data and recommendations from Putnam and others (1960), Goelz (1995) estimated that desirable residual stocking after thinning in even-aged, sawtimber stands of southern hardwoods ranges from 65 to 80 percent. Goelz (1997) further estimated that the minimum stocking level necessary to maintain satisfactory stand-level growth in these same stands ranges from 40 to 60 percent, depending on tree size. Similar ranges for minimum acceptable stocking have been reported in upland oak forests (Hilt 1979) and Allegheny hardwood forests (Lamson and Smith 1988).

The combination of thinning and improvement cutting used in mixed-species hardwood stands also improves both species composition and quality of the residual stand (Meadows 1996). Marking rules and prescriptions that emphasize both quality and value of individual trees, rather than uniform spacing and residual stand density, tend to increase the proportion of high-quality, high-value trees and to decrease the proportion of low-quality, low-value trees in the residual stand. Trees that are damaged or diseased, have low-quality boles, or are undesirable species are removed from the stand under these marking rules; trees that are healthy, have high-quality boles, and are desirable species are retained. Improvement cuttings also may reduce the populations of disease-causing fungi in stands with a high proportion of diseased trees.

Thinnings may have adverse effects on bole quality, specifically in the form of new epicormic branches that may develop along the boles of residual hardwood trees. Epicormic branches are adventitious twigs that develop from dormant buds along the bole. If present in sufficient numbers, epicormic branches may reduce log grade and both lumber grade and value. Species and tree health appear to control the release of these dormant buds when the tree is exposed to some type of disturbance, such as thinning (Meadows 1995). Well-designed hardwood thinnings tend to retain healthy, sawtimber trees and to remove most poletimber trees and low-quality sawtimber trees. As a result, the proportion of dominant and codominant trees typically increases after thinning. These healthy, upper crown-class trees are much less likely to produce epicormic branches than are unhealthy, lower crown-class trees (Meadows 1995). Consequently, the production of epicormic branches across the residual stand actually may decrease after a well-designed thinning (Sonderman and Rast 1988). In contrast, poorly designed thinnings, in which marking guidelines fail to focus on retention of healthy, high-quality trees, typically result in the production of numerous epicormic branches along the boles of residual trees.

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Bottomland hardwood stands in the Delta region of western Mississippi often are infected with *Inonotus hispidus*, a canker decay fungus that causes the disease commonly known as hispidus canker. The fungus occurs most frequently on willow oak (*Q. phellos*), water oak, and Nuttall oak (*Q. texana*) in the Delta region, but also may be found on other red oaks, white oak (*Q. alba*), hickory (*Carya* spp.), and other hardwoods. Hispidus canker causes severe degradation and cull in infected trees. The fungus results in formation of a large, spindle-shaped canker usually at the site of an old branch stub 12 to 15 feet or more up the bole of the infected tree (McCracken 1978). The central part of the canker is concave. Damage occurs in the form of heartwood decay, in which the wood behind the canker becomes soft and delignified. Presence of hispidus canker greatly increases the likelihood of stem breakage at the site of the canker. Improvement cuttings to remove trees with hispidus canker may reduce spore production and dissemination within infested stands and thus may minimize spread of the disease to adjacent trees (McCracken and Toole 1974).

Our study is part of a larger research project investigating relationships between silvicultural practices and insect and disease populations in southern hardwood forests. The goals of this larger project are (1) to understand and quantify the effects of stand modification on insect and disease populations and (2) to use this knowledge to develop pest management recommendations for use in silvicultural prescriptions.

This paper considers only one study site and addresses only the silvicultural component of the larger project. Our objectives were (1) to determine the effects of thinning on stand growth, development, and yield; (2) to determine the effects of thinning on tree growth and bole quality; and (3) to determine the effects of thinning on insect and disease populations, with emphasis on those pests that lead to degradation and/or mortality.

METHODS

Study Area

The study is located on the Delta National Forest in the Delta region of western Mississippi. Specifically, the study area is adjacent to Ten Mile Bayou, within the flood plain of the Big Sunflower River, in southeastern Sharkey County. The site is nearly flat and is subject to frequent periodic flooding during the winter and spring months. Floodwaters may remain on the site for several weeks.

Soils across most of the study area are classified in the Sharkey series (very-fine, smectitic, thermic Chromic Epiaquerts), but small portions of the area are interspersed with Alligator soils (very-fine, smectitic, thermic Chromic Dystraquerts). Dowling soils (very-fine, smectitic, nonacid, thermic Vertic Endoaquepts) also occur in small depressions. All three soils are poorly to very poorly drained, very slowly permeable clays that shrink and form wide, deep cracks when dry and expand when wet. They formed in fine-textured, Mississippi River alluvium deposited in slackwater areas of the flood plain. Average site indices of the Sharkey soils are 92 feet at 50 years for willow oak and 91 feet at 50 years for Nuttall oak, whereas average site index of the Alligator soils is 88 feet at 50 years for both species (Broadfoot 1976).

The study site supports an even-aged, red oak-sweetgum (*Quercus* spp.-*Liquidambar styraciflua*) stand, in which the primary red oak species are willow and Nuttall oaks. In addition to sweetgum, other common species in the overstory include sugarberry (*Celtis laevigata*), American elm (*Ulmus americana*), common persimmon (*Diospyros virginiana*), green ash (*Fraxinus pennsylvanica*), and honeylocust (*Gleditsia triacanthos*). The stand was 55 years old when we installed the study.

Plot Design

Plot design was modified from the standard format for silvicultural research plots, as described by Marquis and others (1990). Each treatment was applied uniformly across a 4.8-acre rectangular treatment plot that measured 6 by 8 chains (396 by 528 feet). We established four 0.6-acre rectangular measurement plots in the center of each treatment plot. Each measurement plot was 2 by 3 chains (132 by 198 feet), which allowed a buffer strip 1 chain (66 feet) wide around each group of four measurement plots. The entire study area is 9.6 acres.

Treatments

Two levels of treatment were applied to the study area: (1) an unthinned control, and (2) an operational thinning marked by personnel from Delta National Forest. The harvest operation combined low thinning and improvement cutting to remove most of the poletimber trees and those sawtimber trees that were damaged, diseased, had poor bole quality, or were undesirable species. Special emphasis was placed on removal of all red oaks infected with *Inonotus hispidus*.

The operational thinning was applied in August 1997. A mechanized feller buncher with a continuously running cutting head was used to directionally fell all marked trees. Felled trees were topped and delimbed in the woods. Rubber-tired skidders removed merchantable products in the form of longwood.

Measurements and Statistical Analysis

We conducted a preharvest survey to determine initial stand density and species composition on each 0.6-acre measurement plot. Species, diameter at breast height (d.b.h.), crown class, and tree class, as defined by Meadows (1996), were recorded on all trees ≥5.5 inches d.b.h. The number of epicormic branches on the 16-foot-long butt log of all “leave” trees was also tallied. Log grade, as defined by Rast and others (1973), of the 16-foot-long butt log and sawtimber merchantable height were recorded on all “leave” trees ≥13.5 inches d.b.h. We measured d.b.h., crown class, and the number of epicormic branches on the butt log at the end of each of the first 3 years after thinning. We measured these variables, as well as tree class, log grade, and sawtimber merchantable height, again at the end of the sixth year after

thinning. Meadows and others (2002) reported 3-year results, and Meadows and others (2006) reported 6-year results. At the end of the 10th year after thinning, we measured d.b.h., crown class, and the number of epicormic branches, and surveyed all plots for damage and infection by insects and diseases.

Data were subjected to a one-way analysis of variance for a randomized complete block design with four replications of two treatments, for a total of eight experimental units. All effects were considered fixed. Alpha was set at 0.05. Plot-level variables represented the mean for all residual trees on each measurement plot. Means were separated through the use of Duncan's multiple range test.

RESULTS AND DISCUSSION

Stand Conditions Prior to Thinning

Prior to thinning, the study area as a whole averaged 98 trees and 125 square feet of basal area per acre, with a quadratic mean diameter of 15.4 inches. These means represent data averaged across all eight plots. Quadratic mean diameter is a stand-level variable calculated from stand basal area per acre and the number of trees per acre. It is defined as the diameter of a tree whose basal area equals the average basal area per tree within the stand. Average stocking across the entire study area was 102 percent, which exceeded the level (100 percent) at which thinning is recommended in even-aged stands of southern bottomland hardwoods (Goelz 1995). We found no significant differences between treatments in any preharvest characteristics (table 1). Although the stand was overstocked, most dominant and codominant trees appeared vigorous and exhibited few signs of poor health. Hispidus canker was found on about 24 percent of red oaks in the study area, but most infected red oaks were in the intermediate and overtopped crown classes.

Red oaks and sweetgum clearly dominated the stand. Prior to thinning, these species together accounted for 91 percent of the basal area of the stand. Red oaks (primarily willow and Nuttall oaks) comprised 43 percent of the basal area and dominated the upper canopy of the stand. Quadratic mean diameter of red oaks before thinning was 16.7 inches. Nearly all of the largest trees in the stand were red oaks. Sweetgum accounted for 48 percent of the basal area and was found in both the upper and middle canopies. Quadratic mean diameter of sweetgum before thinning was 15.1 inches. Other species, such as sugarberry and American elm, made up the remaining 9 percent of the basal area. These species were found almost exclusively in the lower canopy.

Stand Development After Thinning

Stand conditions immediately after thinning—The thinning operation reduced stand density to 32 trees and 59 square feet of basal area per acre, produced a quadratic mean diameter of 18.4 inches, and reduced stocking to 47 percent (table 2). It removed 66 percent of the trees and 52 percent of

Table 1—Treatment means (±SE) for stand conditions prior to application of two thinning treatments

Treatment	Trees	Basal area	Quadratic mean diameter	Stocking
	number per acre	*square feet per acre*	*inches*	*percent*
Unthinned	100±4 a	127±6 a	15.2±0.4 a	104±5 a
Thinned	95±9 a	123±6 a	15.5±0.4 a	101±6 a

Means followed by the same letter are not significantly different at the 0.05 level of probability (n = 4 per treatment; P = 0.70 for number of trees, P = 0.74 for basal area, P = 0.74 for quadratic mean diameter, P = 0.72 for stocking).

Table 2—Treatment means (±SE) for stand conditions immediately after application of two thinning treatments

Treatment	Trees	Basal area	Quadratic mean diameter	Stocking
	number per acre	*square feet per acre*	*inches*	*percent*
Unthinned	100±4 a	129±6 a	15.4±0.4 a	105±5 a
Thinned	32±1 b	59±6 b	18.4±0.8 a	47±4 b

Means followed by the same letter are not significantly different at the 0.05 level of probability (n = 4 per treatment; P < 0.01 for number of trees, P = 0.01 for basal area, P = 0.09 for quadratic mean diameter, P = 0.01 for stocking).

the basal area. Average d.b.h. of trees removed was 13.5 inches. Across the study site, thinning removed about 3,500 board feet (Doyle scale) of sawtimber and about 11 cords of pulpwood, per acre. Thinning produced stand density characteristics significantly different from the unthinned control plots (table 2), but quadratic mean diameter of the thinned plots, 18.4 inches, was not significantly different ($P = 0.09$) from quadratic mean diameter of the unthinned control plots, 15.4 inches.

The thinning operation reduced stocking to 47 percent, a level approaching the minimum residual stocking level necessary to maintain satisfactory stand-level growth, as recommended for southern hardwoods (Goelz 1997) and for other hardwood forest types (Hilt 1979, Lamson and Smith 1988). Removal of all red oaks infected with hispidus canker resulted in an unusually heavy thinning. However, even with the additional removal of diseased red oaks, thinning improved species composition of the residual stand. It increased the red oak component from 43 to 56 percent of stand basal area, and reduced the sweetgum component from 48 to 41 percent of stand basal area.

Stand conditions 10 years after thinning—There has been little stand-level growth in the unthinned control plots over the past 10 years (table 3). Stand basal area in the unthinned control plots increased from 129 to 135 square feet per acre, an average of only 0.6 square feet per acre per year during the 10-year period, with no net basal area growth over the past 4 years. The number of trees per acre decreased from 100 to 83, an average mortality of 1.7 percent per year, a somewhat higher than normal rate for unmanaged stands of southern bottomland hardwoods. The losses from mortality in the unthinned control plots negated most of the gross growth in basal area, such that there has been very little net gain in stand basal area over the past 10 years. Stocking across the unthinned control plots 10 years after study inception averaged 108 percent (table 3), a level that exceeds maximum full stocking (100 percent). The unthinned control plots are clearly overstocked and stagnant, a condition that has led to very slow stand-level growth and moderately high mortality.

Ten-year stand basal area growth was not significantly greater ($P = 0.18$) in the thinned plots than in the unthinned control plots (table 3). Stand basal area in the thinned plots increased from 59 to 69 square feet per acre, an average of 1.0 square feet per acre per year during the 10-year period, well below the rate that might be expected in a fully stocked stand of southern bottomland hardwoods. According to the stocking chart published by Goelz (1997), average stocking across the thinned plots 10 years after thinning (54 percent) falls on the C-10 line of stocking, indicating that it will take 10 more years of growth for the thinned plots to reach minimum full stocking (B-line). The thinning operation in our study reduced stocking to a level well below the B-line, creating an understocked residual stand that is still understocked 10 years after thinning. The thinned plots have not been able to utilize fully the potential productivity of the site over the past 10 years and are not expected to do so during the next 10 years. Consequently, stand-level growth in the thinned plots may be depressed for as much as 10 more years before full site occupancy is recovered.

We were unable to detect significant differences ($P = 0.06$) between treatments in quadratic mean diameter 10 years after thinning (table 3). Quadratic mean diameter of the unthinned control plots increased 1.8 inches over the past 10 years, from 15.4 to 17.2 inches, whereas quadratic mean diameter of the thinned plots increased 2.3 inches, from 18.4 to 20.7 inches (table 3). However, the similarity between the two treatments in the magnitude of the 10-year increase in quadratic mean diameter is misleading. Much of the 10-year increase in quadratic mean diameter of the unthinned control plots is the direct result of the deaths of numerous small trees rather than the result of actual diameter growth by surviving trees. Because quadratic mean diameter is calculated directly from stand basal area per acre and the number of trees per acre, deaths of trees smaller than the current quadratic mean diameter produce an immediate increase in the quadratic mean diameter of the surviving trees in the stand. In contrast, because mortality in the thinned plots was relatively low, most of the 10-year increase in quadratic mean diameter of the thinned plots is due to actual diameter growth of residual trees. Therefore, even though the magnitude of the 10-year increase in quadratic mean diameter is similar between the two treatments, the basis for the increase is clearly different.

Table 3—Treatment means (±SE) for stand conditions 10 years after application of two thinning treatments

Treatment	Trees	Cumulative mortality	Basal area	Cumulative basal area growth	Quadratic mean diameter	Stocking
	number per acre	*percent*	*-------- square feet per acre --------*		*inches*	*percent*
Unthinned	83±3 a	16±2 a	135±8 a	6±3 a	17.2±0.5 a	108±6 a
Thinned	30±2 b	7±4 a	69±6 b	10±2 a	20.7±0.7 a	54±5 b

Means followed by the same letter are not significantly different at the 0.05 level of probability ($n = 4$ per treatment; $P < 0.01$ for number of trees, $P = 0.0502$ for cumulative mortality, $P = 0.02$ for basal area, $P = 0.18$ for cumulative basal area growth, $P = 0.06$ for quadratic mean diameter, $P = 0.01$ for stocking).

Diameter Growth

Diameter growth is a tree-level variable defined, in this study, as the average diameter growth of all trees across a plot or across some specified group within a plot. Diameter growth, a tree-level variable, and the increase in quadratic mean diameter, a stand-level variable, are not synonymous terms. Diameter growth is an indication of the average rate of growth of individual trees within a stand, whereas the increase in quadratic mean diameter is an indication of the change in size of the average tree in a stand.

Thinning significantly increased cumulative diameter growth of residual trees ($P = 0.01$ for year 1, $P < 0.01$ for years 3, 6, and 10), averaged across all species, throughout the 10 years since thinning (fig. 1). A significant difference in average diameter growth between the thinned plots and the unthinned control plots was detected even after the first year, which is somewhat unusual. The difference between treatments widened over time. By the end of the 10th year after thinning, cumulative diameter growth of residual trees in the thinned plots was about 2.3 times greater than cumulative diameter growth of surviving trees in the unthinned control plots—2.5 inches as compared to only 1.1 inches, respectively.

When we separated the data by species groups, we found that red oaks and sweetgum in the thinned plots had similar cumulative diameter growth responses 10 years after the operational thinning (fig. 2). Thinning roughly doubled diameter growth of both species groups, relative to the unthinned control. Residual red oaks in the thinned plots grew 2.8 inches in diameter; residual sweetgum in the thinned plots grew 2.2 inches. Both values were significantly greater ($P = 0.01$ for red oak, $P < 0.01$ for sweetgum) than the corresponding values in the unthinned control. Ten-year cumulative diameter growth of red oaks and sweetgum in the unthinned control plots averaged 1.5 and 0.9 inches, respectively.

Of special importance in this study is that thinning significantly increased ($P = 0.01$) 10-year cumulative diameter growth of codominant trees by 73 percent over the unthinned control, when averaged across all species (fig. 3). Residual codominant trees in the thinned plots grew 2.6 inches in diameter, whereas codominant trees in the unthinned control plots grew 1.5 inches. Codominant trees comprise the bulk of sawtimber crop trees in most hardwood stands and are generally the most valuable trees in the stand. However, we were unable to detect statistically significant differences ($P = 0.34$) between treatments in 10-year cumulative diameter growth of dominant trees. Thinning also nearly tripled 10-year cumulative diameter growth of residual trees in the intermediate crown class, relative to the unthinned control. This significant increase ($P = 0.03$) in diameter growth by residual trees in the intermediate crown class is important because many of these poletimber trees exhibited good potential to develop into valuable sawtimber trees in the near future. We made no comparisons for trees in the overtopped crown class because thinning removed all overtopped trees.

The operational thinning successfully increased diameter growth of residual trees during the first 10 years after treatment. We observed excellent diameter growth responses by both red oak and sweetgum trees in the codominant crown

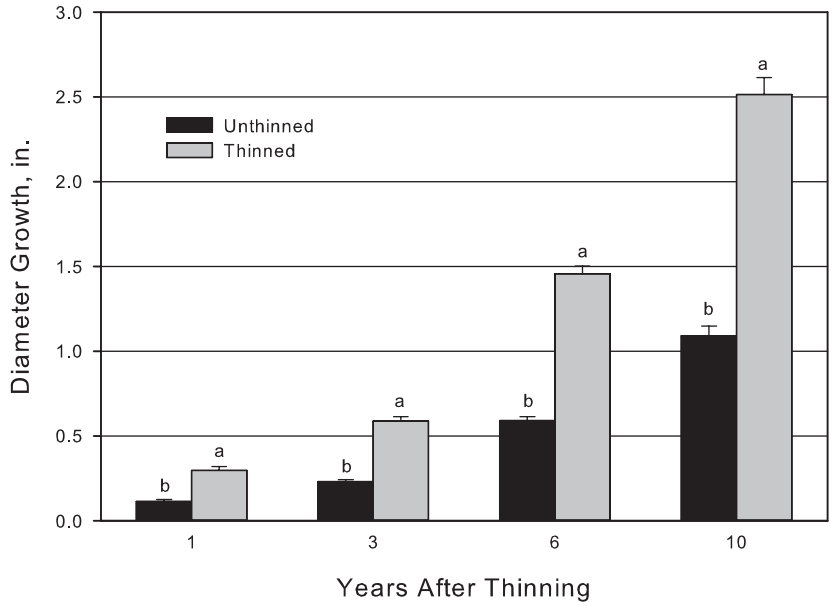

Figure 1—Cumulative diameter growth (±SE) of residual trees 1, 3, 6, and 10 years after application of two thinning treatments. Means within each year followed by the same letter are not significantly different at the 0.05 level of probability ($n = 4$ per treatment; $P = 0.01$ for year 1, $P < 0.01$ for years 3, 6, and 10).

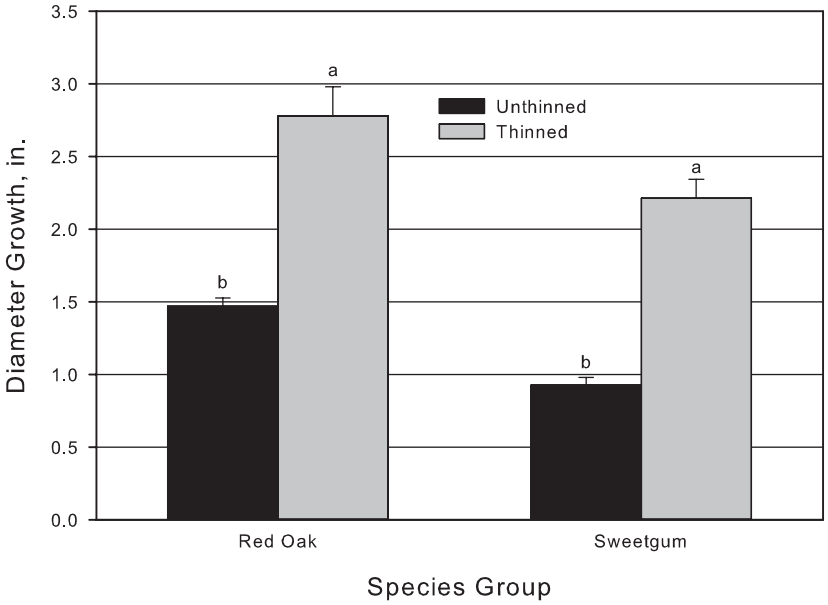

Figure 2—Cumulative diameter growth (±SE) of residual trees, by species group, 10 years after application of two thinning treatments. Means within each species group followed by the same letter are not significantly different at the 0.05 level of probability (n = 4 per treatment; P = 0.01 for red oak, P < 0.01 for sweetgum).

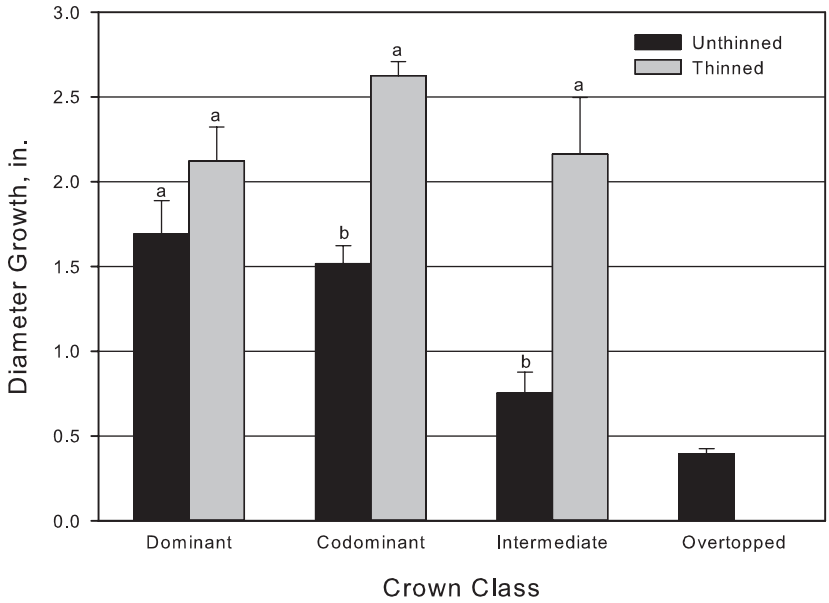

Figure 3—Cumulative diameter growth (±SE) of residual trees, by crown class, 10 years after application of two thinning treatments. Means within each crown class followed by the same letter are not significantly different at the 0.05 level of probability (n = 4 per treatment; P = 0.34 for dominant, P = 0.01 for codominant, P = 0.03 for intermediate).

class. Most of those trees, but particularly the red oaks, are classified as crop trees and are the most desirable and most valuable trees in the stand for production of high-quality sawtimber. From a timber production perspective, thinning greatly enhanced diameter growth of the most valuable trees in the stand.

Production of Epicormic Branches

Thinning operations in hardwood stands, while producing positive impacts on diameter growth of residual trees, also may have negative effects on bole quality. Thinning may stimulate production of new epicormic branches along the merchantable boles of residual trees. Because they cause

defects in the underlying wood and can reduce both log grade and subsequent lumber value, the possible production of epicormic branches on residual trees can be a serious problem when thinning hardwood stands. However, well-designed thinning prescriptions with marking rules that emphasize retention of healthy, high-quality trees can minimize production of new epicormic branches in most hardwood stands.

When averaged across all trees and all species, the mean number of epicormic branches on the butt logs of residual trees in the thinned plots increased steadily through the end of the third year after thinning, but thereafter remained relatively stable through the end of the 10th year (fig. 4). In contrast, the mean number of epicormic branches on the butt logs of trees in the unthinned control plots remained fairly constant through the first 10 years of the study. Through the first 6 years, means for the two treatments did not differ significantly from each other ($P = 0.38$ for year 1, $P = 0.10$ for year 3, $P = 0.07$ for year 6). However, the relatively large standard errors associated with the thinned plots at the end of the third and sixth years (SE = 1.0 for the thinned plots in both years, SE = 0.6 for the unthinned plots in both years) may indicate that there was too much variation within the thinned plots to allow detection of statistically significant differences in both years. We did detect a statistical difference between treatments ($P = 0.04$) at the end of the 10th year after thinning. At that time, residual trees in the thinned plots averaged 6.1±0.8 epicormic branches on the butt log, whereas surviving trees in the unthinned control plots averaged 3.1±0.5 branches.

Based on the 10-year means presented in figure 4, thinning had a detrimental effect on bole quality, as evidenced by a significant increase in the number of epicormic branches on the butt logs of residual trees in the thinned plots. However, treatment means in figure 4 were calculated across all trees and all species and reflect only a broad analysis of the data. In fact, we observed that the number of epicormic branches on the butt log varied widely among individual trees. Most healthy trees, with large, well-shaped crowns and dense foliage, had either no epicormic branches or only a few. Conversely, most unhealthy trees, with small crowns and sparse foliage, generally had many epicormic branches.

To diagnose the source of the broad variation in epicormic branch production across individual trees, we partitioned the data by species groups. Hardwood species vary widely in their susceptibility to the production of epicormic branches (Meadows 1995). In our study, thinning had no effect ($P = 0.56$) on production of epicormic branches on residual red oaks, but caused a large, significant increase ($P = 0.01$) in the number of epicormic branches on the butt logs of residual sweetgum trees 10 years after thinning (fig. 5). In fact, residual red oaks in the thinned plots averaged only 3.2 epicormic branches on the butt log, whereas residual sweetgum trees averaged 9.7 branches, more than three times as many. Surviving red oaks in the unthinned control plots averaged 2.5 branches on the butt log, whereas surviving sweetgum trees averaged 3.8 branches.

Because most red oaks and sweetgum are classified as highly susceptible to the production of epicormic branches (Meadows 1995), further diagnosis was required to explain

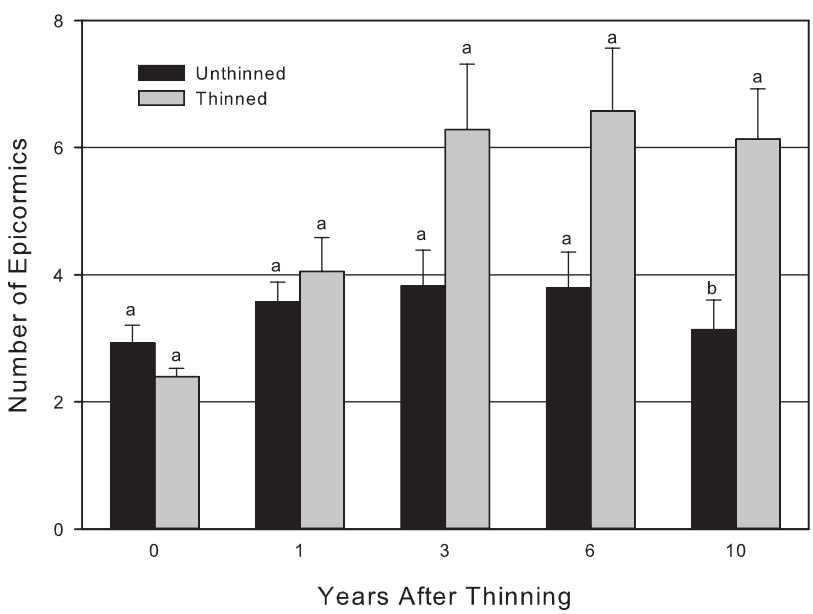

Figure 4—Mean number (±SE) of epicormic branches found on the butt logs of residual trees initially and 1, 3, 6, and 10 years after application of two thinning treatments. Means within each year followed by the same letter are not significantly different at the 0.05 level of probability ($n = 4$ per treatment; $P = 0.15$ for year 0, $P = 0.38$ for year 1, $P = 0.10$ for year 3, $P = 0.07$ for year 6, $P = 0.04$ for year 10).

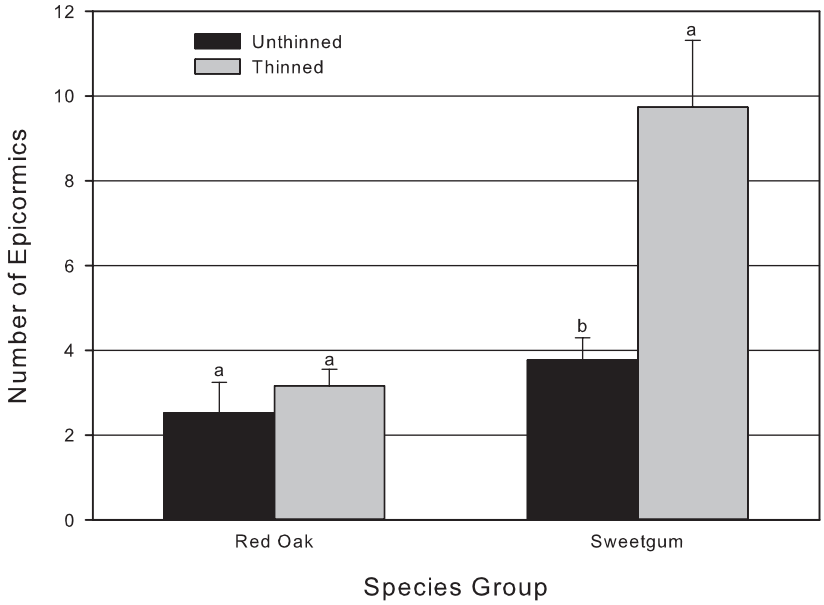

Figure 5—Mean number (±SE) of epicormic branches found on the butt logs of residual trees, by species group, 10 years after application of two thinning treatments. Means within each species group followed by the same letter are not significantly different at the 0.05 level of probability (n = 4 per treatment; P = 0.56 for red oak, P = 0.01 for sweetgum).

the large difference between these species groups in the number of epicormic branches found on the butt logs of residual trees in the thinned plots 10 years after thinning. We observed that most residual red oaks in the thinned plots were healthy, dominant, or strongly codominant trees that produced very few new epicormic branches after thinning. On the other hand, most residual sweetgum trees in the thinned plots exhibited poor-to-moderate health and were classified as intermediate or weakly codominant trees that produced many new epicormic branches after thinning. These observations strongly support the hypothesis proposed by Meadows (1995) that healthy, vigorous trees, even of highly susceptible species like most bottomland red oaks, are much less likely to produce epicormic branches than are trees in poor health.

When evaluating the effects of thinning on the production of epicormic branches in hardwoods, the most important consideration is the number of epicormic branches on the butt logs of crop trees, particularly red oak sawtimber trees and, to a much lesser extent, sweetgum sawtimber trees. Crop trees are favored during the thinning operation and are most likely to produce high-quality, high-value sawtimber. Because they are the most valuable trees in the stand, any significant increase in the production of epicormic branches along the boles of crop trees will reduce the overall value of the stand and will prove to be costly to the landowner. In our study, thinning had no effect (P = 0.96) on the number of epicormic branches on the butt logs of red oak sawtimber trees 10 years after thinning (fig. 6). In fact, red oak sawtimber trees in both the thinned plots and the unthinned control plots averaged fewer than three epicormic branches on the butt log, generally not sufficient to cause a negative impact on log grade. Conversely, thinning significantly increased (P = 0.01) the number of epicormic branches on the butt logs of sweetgum sawtimber trees. These trees averaged 9.5 epicormic branches on the butt log alone. Based on a general rule of thumb that as few as five epicormic branches may be sufficient to cause a reduction in log grade (Meadows and Burkhardt 2001), it is likely that log grade of many sweetgum sawtimber trees in the thinned plots was affected adversely by the increased production of epicormic branches following thinning. The boles of red oak poletimber trees in the thinned plots supported 10.2±5.2 epicormic branches 10 years after thinning. Due to this large standard error, we were unable to detect significant differences (P = 0.13) between treatments, even though red oak poletimber in the unthinned control plots averaged only 3.2±1.8 epicormic branches. Many of the poletimber trees in the thinned plots were relatively unhealthy, lower crown-class trees that produced many new epicormic branches after thinning. We made no comparisons among sweetgum poletimber trees because they have little or no potential to develop into high-value crop trees.

Pathogen Infections Since Thinning

Our third objective in this study was to determine the effects of thinning on insect and disease incidence. The thinning operation removed all red oaks infected with *Inonotus hispidus* in an effort to minimize the likelihood of new hispidus infections and thereby improve forest health.

We surveyed all treatment plots to determine the incidence of new infections of various pathogens by the end of the 10th

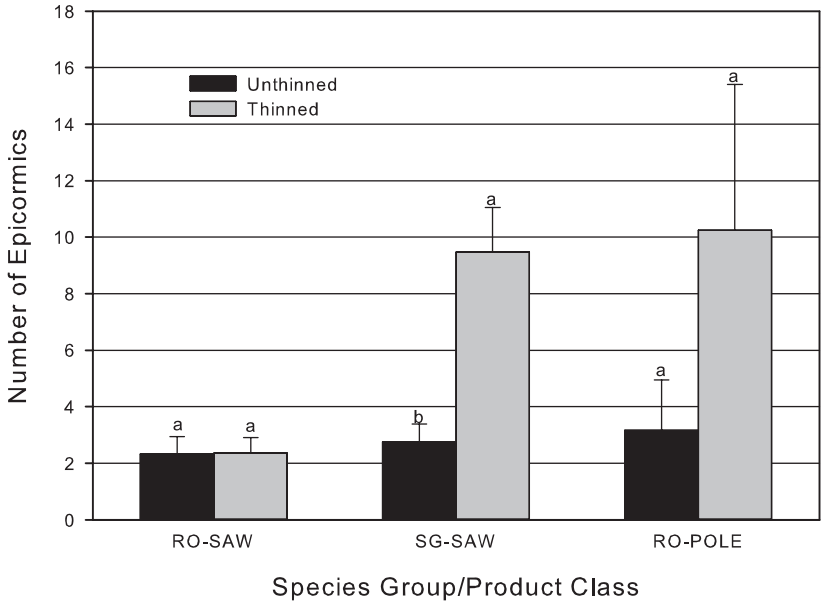

Figure 6—Mean number (±SE) of epicormic branches found on the butt logs of residual trees, by species group and product class, 10 years after application of two thinning treatments (RO-SAW = red oak sawtimber, SG-SAW = sweetgum sawtimber, RO-POLE = red oak poletimber). Means within each species group/product class combination followed by the same letter are not significantly different at the 0.05 level of probability (n = 4 per treatment; P = 0.96 for red oak sawtimber, P = 0.01 for sweetgum sawtimber, P = 0.13 for red oak poletimber).

Table 4—Percentage (±SE) of trees with new infections of various pathogens 10 years after application of two thinning treatments

	Pathogens				
Treatment	Deadwood insects	Hispidus canker	Spiculosa canker	Hypoxylon canker	Bacterial wetwood
Unthinned	6.6±1.6 a	1.0±0.6 a	0.0±0.0 a	3.5±0.5 a	2.0±0.1 a
Thinned	10.7±3.8 a	1.4±1.4 a	2.9±1.7 a	1.4±1.4 a	5.7±2.3 a

Means followed by the same letter are not significantly different at the 0.05 level of probability (n = 4 per treatment; P = 0.46 for deadwood insects, P = 0.85 for hispidus canker, P = 0.19 for spiculosa canker, P = 0.34 for hypoxylon canker, P = 0.21 for bacterial wetwood).

year after thinning (table 4). Incidence of deadwood insects (termites, carpenter ants, ambrosia beetles, and powderpost beetles) did not differ significantly between treatments (P = 0.46). In addition to hispidus canker, caused by *Inonotus hispidus*, we surveyed for new infections of spiculosa canker, a white rot disease caused by *Phellinus spiculosus*, and hypoxylon canker, a decay disease caused by *Hypoxylon atropunctatum* common on drought-stressed and suppressed trees. Incidence of new disease infections was uniformly low across both treatments and did not differ significantly between treatments (P = 0.85 for hispidus, P = 0.19 for spiculosa, P = 0.34 for hypoxylon). Incidence of bacterial wetwood, a common disease condition that causes "honeycombing" and "shake" in oak lumber, was not significantly different between treatments (P = 0.21). Our 10-year survey of new pathogen infections indicates that thinning had no detrimental effects on overall forest health.

CONCLUSIONS

1. Thinning severely reduced stocking and created an understocked residual stand that has not been able to utilize fully the potential productivity of the site during the 10 years since thinning and is not expected to do so during the next 10 years.
2. Thinning significantly increased diameter growth of residual trees, especially red oak sawtimber trees.

3. Thinning had no effect on production of epicormic branches on the butt logs of red oak sawtimber trees, but greatly increased production of epicormic branches on the butt logs of sweetgum sawtimber trees.
4. Thinning improved overall forest health by creating tree and stand conditions not conducive to new infections by a variety of pathogens. Through the targeted removal of all red oaks infected with hispidus canker, thinning reduced *Inonotus hispidus* inoculum, but did not reduce the number of new hispidus infections relative to the unthinned control plots.

ACKNOWLEDGMENTS

We express appreciation to Delta National Forest for providing the study site and for its cooperation in all phases of study installation and measurement. We specifically thank Larry Moore and Ralph Pearce of Delta National Forest for their continuing assistance in this study. We also thank Luben Dimov and Tracy Hawkins for providing helpful suggestions on earlier drafts of this manuscript.

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.

Goelz, J.C.G. 1995. A stocking guide for southern bottomland hardwoods. Southern Journal of Applied Forestry. 19(3): 103–104.

Goelz, J.C.G. 1997. C-lines of stocking for southern bottomland hardwoods: a guide to identifying insufficient stocking. Res. Note SO-385. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 3 p.

Hilt, D.E. 1979. Diameter growth of upland oaks after thinning. Res. Pap. NE-437. Broomall, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 12 p.

Lamson, N.I.; Smith, H.C. 1988. Thinning cherry-maple stands in West Virginia: 5-year results. Res. Pap. NE-615. Broomall, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 7 p.

Marquis, D.; Smith, C.; Lamson, N. [and others]. 1990. Standard plot layout and data collection procedures for the Stand Establishment and Stand Culture Working Groups, Northeastern Forest Experiment Station. Warren, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 55 p.

McCracken, F.I. 1978. Canker-rots in southern hardwoods. For. Insect & Dis. Leaf. 33. Washington, DC: U.S. Department of Agriculture Forest Service. 4 p.

McCracken, F.I.; Toole, E.R. 1974. Felling infected oaks in natural stands reduces dissemination of *Polyporus hispidus* spores. Phytopathology. 64(2): 265–266.

Meadows, J.S. 1995. Epicormic branches and lumber grade of bottomland oak. In: Lowery, G.; Meyer, D., eds. Advances in hardwood utilization: following profitability from the woods through rough dimension: Proceedings of the twenty-third annual hardwood symposium. [Memphis, TN]: National Hardwood Lumber Association: 19–25.

Meadows, J.S. 1996. Thinning guidelines for southern bottomland hardwood forests. In: Flynn, K.M., ed. Proceedings of the southern forested wetlands ecology and management conference. Clemson, SC: Clemson University, Consortium for Research on Southern Forested Wetlands: 98–101.

Meadows, J.S.; Burkhardt, E.C. 2001. Epicormic branches affect lumber grade and value in willow oak. Southern Journal of Applied Forestry. 25(3): 136–141.

Meadows, J.S.; Goelz, J.C.G. 2001. Fifth-year response to thinning in a water oak plantation in north Louisiana. Southern Journal of Applied Forestry. 25(1): 31–39.

Meadows, J.S.; Leininger, T.D.; Nebeker, T.E. 2002. Thinning to improve growth and control the canker decay fungus *Inonotus hispidus* in a red oak-sweetgum stand in the Mississippi Delta. 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: 183–188.

Meadows, J.S.; Leininger, T.D.; Nebeker, T.E. 2006. Thinning to improve growth and bole quality in an *Inonotus hispidus*-infected, red oak-sweetgum stand in the Mississippi Delta: sixth-year 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: 623–629.

Putnam, J.A.; Furnival, G.M.; McKnight, J.S. 1960. Management and inventory of southern hardwoods. Agric. Handb. 181. Washington, DC: U.S. Department of Agriculture. 102 p.

Rast, E.D.; Sonderman, D.L.; Gammon, G.L. 1973. A guide to hardwood log grading. Revised. Gen. Tech. Rep. NE-1. Upper Darby, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 31 p.

Sonderman, D.L.; Rast, E.D. 1988. Effect of thinning on mixed-oak stem quality. Res. Pap. NE-618. Broomall, PA: U.S. Department of Agriculture Forest Service, Northeastern Forest Experiment Station. 6 p.

POSTERS

Dr. Mike Shelton of the Southern Research Station describes long-term stand development during a field tour stop in the 2005 Crossett Forestry Field Day, Crossett Experimental Forest, Ashley County, Arkansas. (Photo by James M. Guldin)

STREAM CARBON DYNAMICS IN LOW-GRADIENT HEADWATERS OF A FORESTED WATERSHED

April Bryant-Mason, Y. Jun Xu, and Johnny M. Grace III[1]

Abstract—Headwater streams drain more than 70 percent of the total watershed area in the United States. Understanding of carbon dynamics in the headwater systems is of particular relevance for developing best silvicultural practices to reduce carbon export. This study was conducted in a low-gradient, predominantly forested watershed located in the Gulf Coastal Plain region, to (1) investigate spatiotemporal dynamics of carbon concentrations in the headwaters, (2) assess the relationships among stream carbon and nutrient conditions, and (3) quantify carbon export from the entire watershed. Fifteen sites were selected along four first-to-third order streams. Monthly and storm event water samples were collected from December 2005 to September 2007. These samples were analyzed for total and dissolved organic and inorganic carbon, nitrate/nitrite nitrogen, and total and dissolved phosphorus. In addition, instream water quality parameters (dissolved oxygen, temperature, pH) were measured monthly at each site to gather information on stream environmental conditions. The study found a seasonal variation of stream carbon concentration ranging from 9.6 to 30.0 mg/L with the lowest concentrations in January and February. There was a clear increasing trend of dissolved inorganic carbon from the winter to summer months, indicating a critical metabolic role of carbon supply and transport. Over the entire 22 months, the watershed exported a total of 1054.35 t carbon, in a decreasing trend of fluxes from 5.2 to 1.3 kg/ha per month with the increasing drainage area. This information can be useful for designing silvicultural practices that will conserve and maintain ecosystem carbon.

INTRODUCTION

Land use activities by humans have enormously altered the timing, magnitude, and nature of inputs of materials such as sediments, nutrients, and organic matter to aquatic ecosystems. One of the dominant themes in streamwater quality research is the effect of organic materials on eutrophication of coastal waters. Organic carbon interacts with the biogeochemical nitrogen cycle (Campbell and others 2000, Cooper and others 2006, Qualls and others 1991), aids in pollutant transport (Kalbitz and others 2000), and may be a major energy source for microorganisms (del Giorgio and Cole 1998, Marschner and Kalbitz 2003, Tranvik 1992). In forested watersheds, the upper horizons of the soil can contain large amounts of organic matter such as plant litter and soil organic matter degraded by microorganisms (Cory and others 2004). Seventy-five percent of carbon present on land is found as soil organic carbon (Sparks 2003). Consequently, surface runoff and erosion can contribute a large input of carbon to streams.

In aquatic systems, organic carbon is either consumed by the biological community, deposited in the benthic zone, or transformed into atmospheric carbon, all of which can affect streamwater quality. Organic matter is an important part of the aquatic food web, especially in headwater streams where primary production is limited as a result of the canopy cover. Most nitrogen transported by rivers to oceans is associated with organic matter. Therefore, understanding the carbon dynamics in streams and rivers can give a better picture of nitrogen present and the potential for eutrophication.

Headwater streams are particularly important for water quality of an entire watershed because they often drain over 70 percent of the total watershed area. Streams are lotic systems; therefore, upstream effects are ultimately felt downstream. This study was conducted in the headwater streams of a low-gradient, subtropical watershed located in central Louisiana, USA. The study aimed to (1) investigate spatiotemporal dynamics of organic and inorganic carbon concentrations, (2) assess the relationships among stream carbon and nitrate, and (3) quantify carbon export from the headwater catchment.

METHODS

Study Area

The Flat Creek watershed is located in the western part of the Ouachita River Basin in central Louisiana (fig. 1). The basin drains a total land area of 41 439 km^2, characterized by a flat to slightly rolling topography. Flat Creek's drainage area is approximately 369 km^2. Forestry is the dominant land use in the watershed, occupying 61 percent of land and followed by rangeland with 21 percent (Louisiana Department of Environmental Quality 2001). Climate in this region is subtropical with hot, humid summers and mild winters. Long-term average temperatures range from 2.3 °C to 34.1 °C and long-term average rainfall is about 1500 mm/year. Soils in the area are dominated by poorly drained Guyton (silt loam) series along the Flat Creek and Turkey Creek flood plains, with moderately well-drained Sacul-Savannah (fine sandy loam) soils in the upland areas.

Streamwater Sampling and Laboratory Analyses

Four streams in the Flat Creek watershed were sampled: Spring Creek, Turkey Creek, Flat Creek, and Big Creek. Fifteen sites were visited monthly from January 2006 to September 2007 (fig. 1). *In-situ* water quality measurements, including dissolved oxygen, temperature, conductivity, and pH were taken at each site using an YSI 556 (YSI, Inc., Yellow Springs, OH). During each visit grab-water samples

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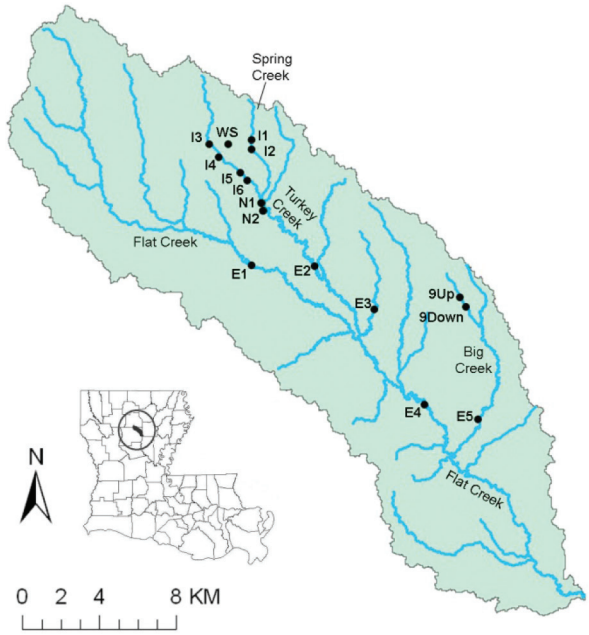

Figure 1—Geographical location of the Flat Creek watershed and water-quality monitoring sites. A weather station (WS) is established between Spring Creek and Turkey Creek.

were collected at each site. In addition, stormwater samples were collected at 6 of the 15 locations with automated Isco samplers (model 6712, Teledyne Isco, Inc., Lincoln, NE). Storm events were defined as enough rain to cause the stream to rise 15 cm in 24 hours. Depending on the rainfall intensity, time since last rainfall, and stream and riparian characteristics, the amount of precipitation for a 15-cm increase of stream level varied.

The laboratory measurements on water samples were conducted in the Wetland Biogeochemistry Institute, Louisiana State University. Water samples were analyzed for total and dissolved organic and inorganic carbon with a Shimadzu Total Organic Carbon Analyzer (model TOC-VCSN, Shimadzu Corporation, Kyoto, Japan) using the combustion/nondispersive infrared gas analysis method. Inorganic carbon and total carbon were measured by the analyzer, and the organic partition was calculated as the difference between total and inorganic carbon. Water for dissolved organic and inorganic carbon analysis was first filtered through a 47-µm glass fiber filter (GF/F Whatman International Ltd., Maidstone, United Kingdom).

Streamflow Measurements and Climatic Observations

Streamflow measurements were collected with a flow meter (SonTek®/YSI Inc., Yellow Springs, OH) and top setting wading rod (Rickly Hydrological Co., Columbus, OH) monthly during baseflow as well as whenever possible during higher flow conditions. Because the streams in the Flat Creek watershed are relatively narrow, most measurements consisted of 5 to 10 cross sections. The autosamplers at the intensive sites record stream level every 15 minutes. Stage-discharge curves developed for sites I1, I3, and I4 were used in conjunction with the stream level to calculate daily discharge. Detailed information about development of the stage-discharge rating curves can be found in Saksa (2007).

Data Analysis

Summary statistics such as mean and standard error were calculated for each month for all stations as well as each site for each sampling month. The number of samples varied with each month (table 1). The number of samples for total carbon

Table 1—Number of samples used in calculating mean and standard error

Month	Total carbon samples	Dissolved carbon samples	Month	Total carbon samples	Dissolved carbon samples
Jan-06	11	11	Jan-07	15	10
Feb-06	8	8	Feb-07	14	15
Mar-06	12	12	Mar-07	15	15
Apr-06	12	12	Apr-07	13	13
May-06	14	14	May-07	15	15
Jun-06	13	5	Jun-07	13	13
Jul-06	13	13	Jul-07	11	12
Aug-06	0	0	Aug-07	11	12
Sep-06	7	5	Sep-07	13	13
Oct-06	9	4			
Nov-06	15	3			
Dec-06	14	11			

is the number of samples for all total carbon concentrations including total inorganic and organic carbon. Similarly, the number of samples for dissolved carbon concentrations refers to dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC).

Carbon mass loading was calculated as:

$$L = e^{(a * \ln Q + b + \varepsilon)} \quad (1)$$

where
L = loading
Q = discharge
a and b = constants (table 2)
ε = an error term assumed to be evenly distributed

The a and b terms were adjusted for each site based on the loading to discharge curve (table 2). E4 calculated classically as:

$$L = Q * C \quad (2)$$

where
C = concentration
L = loading
Q = discharge

RESULTS AND DISCUSSION

Seasonal and Spatial Fluctuation of Stream Carbon Concentrations

For the period from January 2006 to September 2007 total carbon (TC) concentration appeared to be lower during two winter months, January and February, than during other months of the year (>22 mg/L). TC was marginally higher during the summer ($P = 0.052$; $t = -1.95$) while total inorganic carbon (TIC), dissolved carbon (DC), and DIC were higher in the summer (May to October) than the remaining of the year (November to April) ($P < 0.001$). Total organic carbon (TOC) was lower in the summer months than the remaining months ($P < 0.001$). Average TC ranged from 9.6 to 30.0 mg/L with the lowest average concentration present in February 2007 and the highest in December 2006. When separating the total carbon into organic and inorganic forms, a much clearer trend of increased inorganic carbon in the summer and increased organic carbon in the spring is apparent (fig. 2). Organic carbon ranged from 8.4 mg/L in February 2007 to 25.3 mg/L in November 2006. Average inorganic carbon ranged from 1.0 mg/L in March 2007 to 13.2 mg/L in June 2006.

Monthly average of dissolved carbon concentrations ranged from 9.9 mg/L in January 2007 to 29.6 mg/L in July 2007. DOC and DIC had a similar trend to TC. DOC ranged from 9.3 mg/L in July 2006 to 28.1 mg/L in July 2007. January 2007 had the lowest DIC (0.3 mg/L) with September 2006 having the highest (14.3 mg/L). A peak in the TIC to TOC ratio is apparent in the summer months from June through October in 2006 (fig. 3). Late October 2006 marked large rains (more than 8 inches within a single week) which indicated the beginning of the "rainy" season typically in the late fall and winter in central Louisiana.

The increased organic carbon observed during the spring may have resulted from an increasing primary production and/or high storm runoff during the season. For the subtropical headwaters of Flat Creek, DOC in quickflow is a more likely reason than primary production for the seasonal pattern present. Headwater streams act as net sinks for carbon and

Table 2—Slope (a) and intercept (b) for equations to calculate nutrient loading at I1 and I4

Site ID	Nutrient	Intercept	Slope	R-squared
I1	TC	0.2992	1.1762	0.96
I4	TC	3.6900	0.9705	0.95
I1	TOC	–1.7486	1.3051	0.95
I4	TOC	1.8050	1.0825	0.95

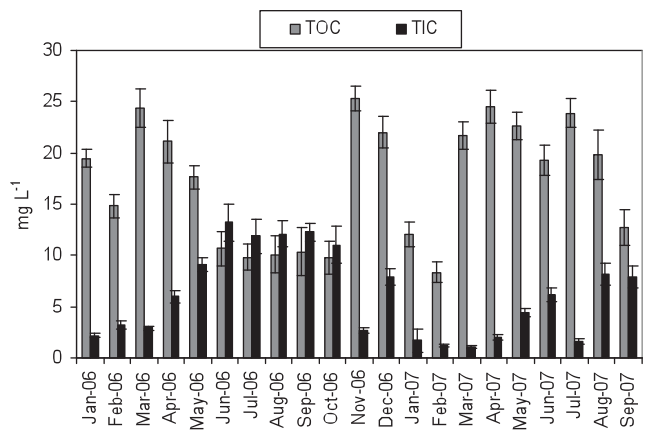

Figure 2—Seasonal fluctuation of total organic carbon (TOC) and total inorganic carbon (TIC) concentrations in the Flat Creek watershed.

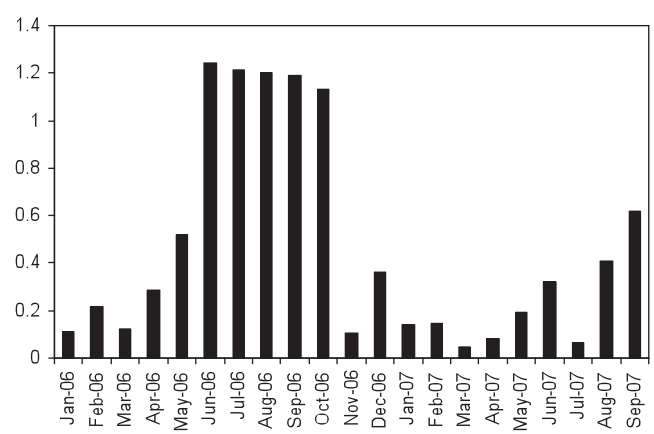

Figure 3—Seasonal trend of the ratio of TIC to TOC in headwater streams in the Flat Creek watershed.

nitrogen since the input is higher than what is processed within the stream (Cooper and others 2006). Also, because of the dense canopy cover in forested headwater streams, primary production has a lesser organic carbon contribution than the contribution from the organic layers of soil that is mobilized in storm events. DOC decreases with soil depth as sorption of dissolved organic matter (DOM) to mineral surfaces occurs in the deeper soil depths (Cory and others 2004); also DOM found in streams is more similar to shallow soil water DOM than the deep soil water DOM (Cory and others 2004). Johnson and others (2006) found that DIC is higher in deeper flow paths in which a 40 to 1 ratio of DIC to DOC existed for emergent ground water. During low flow conditions, which is found in the summer months in Flat Creek, streams receive water from ground water sources and water that has percolated through deeper soil layers enabling most organic carbon to be used by biological sources or abiotically adsorbed to mineral layers (Cory and others 2004) restricting the amount of carbon that is mobile to reach streams. Alternatively, during storm events which occur often in Louisiana during the winter and early spring, quickflow from throughfall, rainfall, and runoff carries rich organic water since it passes through the litter layer and surface soils. Additionally, the rise of streamwater within the banks allows organic materials to enter the water column. The decline in organic carbon in April 2006 to June 2006 shows that TOC is being consumed. DOC decomposition is slower in headwaters, but this process consumes oxygen and converts organic carbon (OC) to inorganic carbon (IC) (del Giorgio and Cole 1998). This fits nicely with the data in which there is a decrease in dissolved oxygen; OC and an increase in IC occurs from spring to summer. Considering spring tends to be a biologically active time, this is expected.

Average TC was lowest at I1 (13.5 mg/L) and highest at I5 (28.6 mg/L^{-}) (fig. 4). Most of the TC was in the dissolved form. Spatially, there was not a clear trend. The local variations, especially local soil characteristics appear to have a larger impact on carbon in the stream than location in the watershed. One site, 9Up, had a large variation due to limited samples collected at this intermittent site. E2 is located downstream of the confluence of Spring Creek (sites I1 and I2) and upper Turkey Creek (sites I3 to I6) and reflects the mixing of lower carbon at Spring Creek and higher carbon at the upper Turkey Creek sites.

Stream Carbon Concentrations during Storm Events

There was not a large difference in organic or inorganic carbon at different stages of the storm hydrograph, but there was a small increase in organic carbon (27.50+3.04 mg/L) in the falling limb (27.50+3.04 mg/L) vs. the rising limb (20.87+1.37 mg/L) and full peak (22.99+1.41 mg/L) (fig. 5). There were no changes in carbon concentrations (20.45 to 20.73 mg/L) during a storm event on January 16, 2007 (fig. 6). However, Spring Creek experienced higher carbon concentrations than Turkey Creek (21.34 mg/L at I1 vs. 18.65 mg/L at I4) during a storm event on October 17, 2006 (fig. 7).

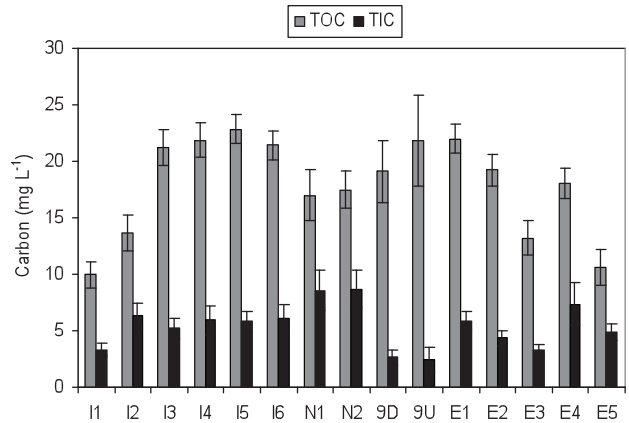

Figure 4—Average total organic carbon and total inorganic carbon concentrations at 15 locations in the Flat Creek watershed. Error bars represent standard error.

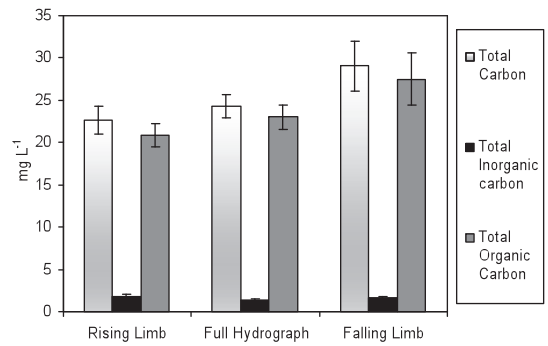

Figure 5—Average total carbon, total inorganic carbon, and total organic carbon during storm events in January 2006 to September 2007 during varying parts of the hydrograph in the Flat Creek watershed. Error bars represent standard error (n = 7 for rising limb, n = 24 for full hydrograph, and n = 5 for falling limb).

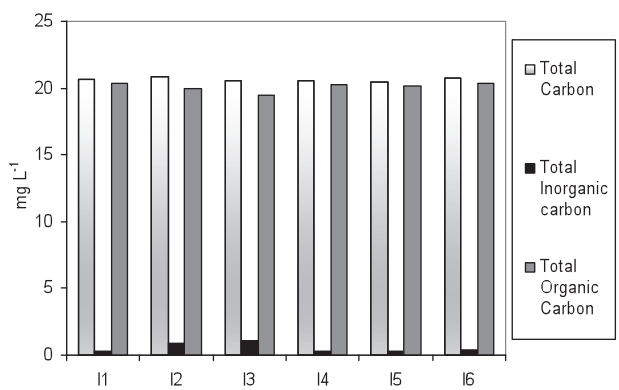

Figure 6—Total carbon, total inorganic carbon, and total organic carbon for all six sites during one storm event on January 15, 2007, in the Flat Creek watershed.

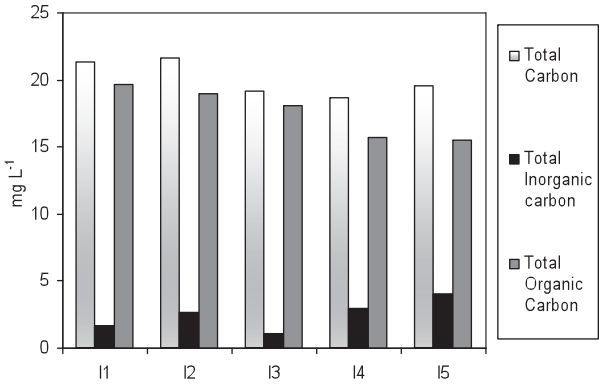

Figure 7—Total carbon, total inorganic carbon, total organic carbon for five sites during one storm event on October 16, 2006, in the Flat Creek watershed.

Previous research suggests that the highest DOC concentration should be during storm events (Cooper and others 2006); however, the DOC concentrations during storm events were only slightly elevated from max DOC measured during monthly water sampling. The literature related to the peak of DOC in the storm hydrograph is contradictory based on our review. Buffam and others (2001) state that the max should occur in the rising limb while Cooper and others (2006) cite various studies that found the max DOC on the falling limb. As stated above, the streams sampled in the study by Buffam and his colleagues had bedrock bottoms, so the streams themselves were not organic matter sources. This greatly contrasts the streams in the Flat Creek watershed. For this reason, it makes sense that Flat Creek's storm data follow more closely to Cooper and others (2006) and not Buffam and others (2001). During a storm event, carbon concentrations did not change among the six sites. This follows what was seen in monthly sampling. This specific storm event on January 16, 2007, followed multiple storm events in December and early January. A storm event on October 17, 2006, broke a long period of dry weather with 16.34 cm of rain. Spring Creek experienced higher carbon concentrations (21.34 mg/L at I1 and 21.65 mg/L at I2) than was seen in the January 16, 2007, storm, and Turkey Creek had lower carbon concentrations (18.65 to 19.57 mg/L). These are small variations and probably are due to differences in runoff and rainfall patterns.

Mass Loading and Transport of Carbon

Carbon loading was calculated using streamflow and concentration for two first-order streams (I1 and I4) and their downstream third-order watershed outlet (E4). Due to the flow conditions of this watershed, it was difficult to develop

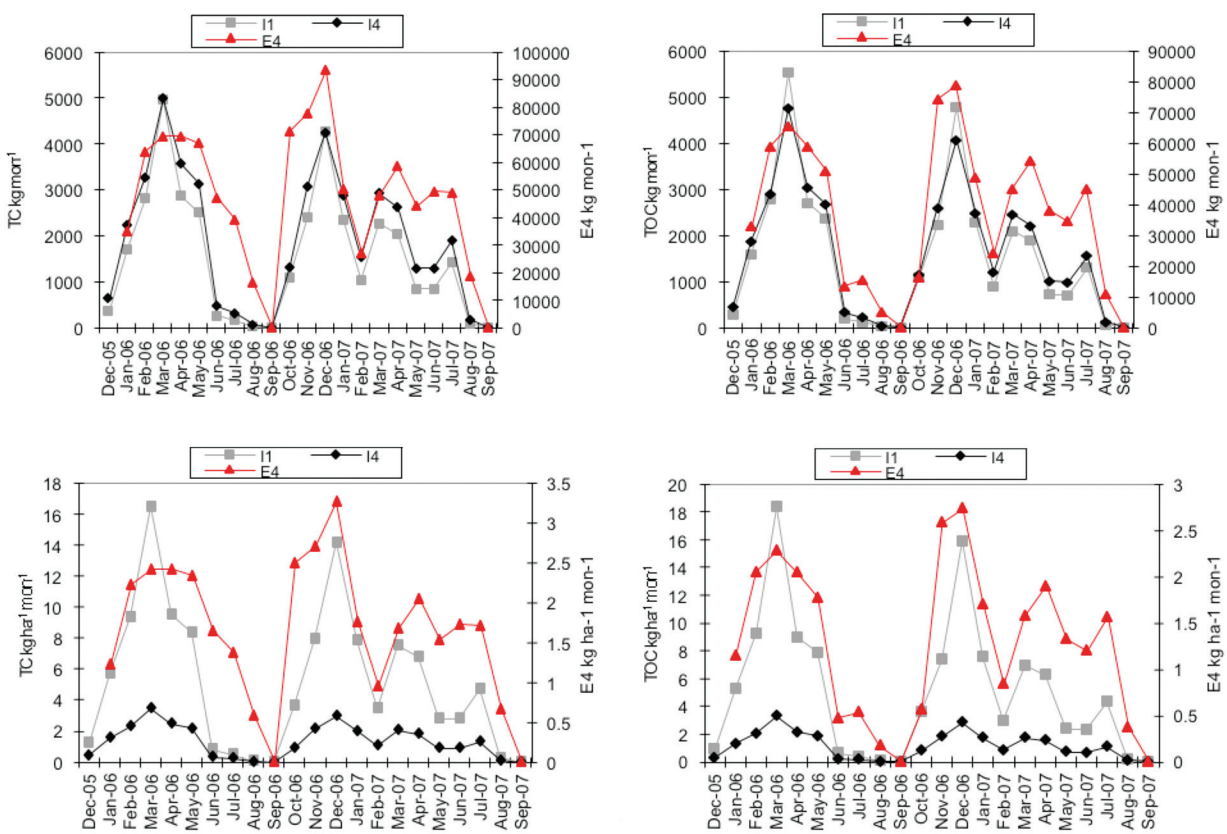

Figure 8—Comparison of mass loading and flux of total carbon and total organic carbon between two first-order (I1 and I4) streams and a third-order stream (E4) in the Flat Creek watershed.

accurate stage-discharge curves (see Saksa 2007). Sites I1 and I4 had the best relationships. The results show that over the 22-month study period total carbon loading at all three sites followed a similar seasonal trend (fig. 8). TC loading at E4 was higher at some points of the study, while I1 and I4 mirrored each other closely. TC loading was highest at E4 (47 925 kg/month) when compared with those at I4 (1905 kg/month) and I1 (1560 kg/month). The loading corresponded to rainfall where the majority of high loads occurred in spring and late fall/winter. I1 is a small stream and responds quickly to little rain. The summer months had low loading which corresponded to a period with little rainfall and low discharge. TOC loading had a similar pattern as that of TC. Loading at I1 had higher peaks than I4 February 2006 and December 2006 (fig. 8). Headwater TOC loading was 1524 kg/month at I1 and 1633 kg/month at I4 (fig. 8). TOC loading at E4 was 36 627 kg/month.

The headwater site, I1, showed higher carbon fluxes because of its smaller drainage size. Total carbon flux from the outlet of the watershed (E4) was 1.7 kg/ha/month, whereas the headwater sites I1 and I4 showed an average carbon flux of 5.2 and 1.33 kg/ha/month, respectively. Similar trends for total organic carbon fluxes were observed, with I1 having average monthly flux of 5.08 kg/ha, I4 having an average monthly flux of 1.14 kg/ha, and E4 having an average monthly flux of 1.28 kg/ha.

Both TC and total inorganic carbon fluxes were about average for a forested watershed in the streams of the Flat Creek watershed. Royer and David (2005) studied DOC loading in an agricultural watershed and found average flux of 3 to 25 kg/ha/year. Using average flux to determine the approximate yearly value, Flat Creek has a range of 16.0 to 62.4 kg/ha/year. This overlaps with the higher end of the range found by Royer and David (2005). An agricultural watershed in the Midwestern United States had DOC loads of 14.1 to 19.5 kg/ha/year (Dalzell and others 2007). It is expected that forests would have higher carbon due to inputs from trees and the organic layer of the soils. Also, agricultural watersheds input nutrients such as nitrate, so carbon would be used by organisms to process the nutrient input. In forested watersheds Dosskey and Bertsch (1994) found a carbon flux of 91.5 kg/ha/year. This is higher than what was calculated for our watershed. Loading in Flat Creek was lower than the Amite, Tangipahoa, and Tickfaw Rivers in Louisiana where these rivers had average annual loading 2404 to 15 780 Mg (Saksa and Xu 2006). Peatlands tend to have the highest organic carbons, and streams in Dee Valley, Scotland, have much higher carbon loads than the Flat Creek watershed. DOC loads in Dee Valley ranged from 1700 to 10 500 kg/km/year (Aitkenhead-Peterson and others 2006).

Relationship between Stream Carbon and Nitrogen

TOC and nitrate/nitrite were compared to see what effects organic carbon has on nutrients, especially nitrate/nitrite. There appears to be two dominating forces in nitrate/nitrite concentrations. The first is storm events. There was a peak in December 2006 (1378 mg/L) that can be attributed to a

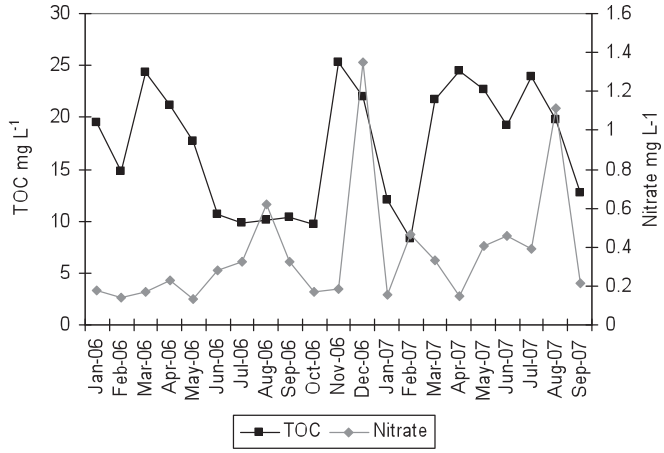

Figure 9—Average nitrate and total organic carbon for all 15 sites from January 2006 to September 2007.

rain event shortly prior to monthly sampling (fig. 9). Other peaks in nitrate/nitrite, such as in August 2006 or February 2007 correspond to decreased TOC. This is not a definitive relationship, however. There are a number of factors that could be impacting nitrate in addition to storm events and TOC concentrations.

Nitrate can be converted to gases such as N_2O and N_2 through the process of denitrification. The process demands the supplies of carbon and anaerobic conditions (Knowles 1982, Seitzinger 1988). When comparing monthly average nitrate/nitrite concentrations to organic carbon concentrations, there is an interesting pattern that arises (fig. 9). In the spring 2006, organic carbon is elevated; however, nitrate/nitrite is minimal. Straus and Lamberti (2000) found that organic carbon concentrations of 30 mg/L completely inhibited nitrification. TOC in March was 25 mg/L and corresponded to nitrate/nitrite of 0.2 mg/L, which is near the reported value for the detection limit. This inhibition of nitrification appears to be occurring in the spring, when biological activity is high. In the summer when TOC is low, there is a peak of nitrate/nitrite further supporting this theory. In the fall, however, there appears to be a different mechanism at work. TOC is high as is nitrate/nitrite. The peak in TOC corresponds with the start of the rainy season. Nitrate/nitrite peaks in December which also may be a result of increased runoff and organic input from leaf fall. In the early part of 2007 that is reported here, there is a repeat of the relationship seen in the spring of 2006 suggesting that this increased in TOC and decreased nitrate/nitrite is a result of biological activity.

Currently carbon analysis, organic or inorganic, is typically not used in regular water-quality monitoring programs. It has been found that carbon can affect nitrification in streams (Strauss and others 2002) indicating the potential importance of measuring carbon in streams. Carbon in streams, especially headwater streams, tends to reflect neighboring land use through surface runoff, making it a valuable parameter to understand. The general trend in figure 9 may indicate

that the carbon concentrations present in the stream may be influencing nitrate/nitrite levels. DOM in streamwater is strongly related to landscape level predictors including loading, transportation, removal, and dilution of DOM (Frost and others 2006). Carbon monitoring may be a beneficial indicator for water quality considering its relationship with nitrogen, a popular indicator for eutrophication and general water quality.

CONCLUSIONS

This study investigated the spatiotemporal dynamics of organic and inorganic carbon concentrations and carbon export in the headwater streams of a low-gradient, subtropical watershed in central Louisiana. Spatial variations did not play a key role in carbon dynamics, but seasonality was a large factor in organic and inorganic carbon levels. TC concentrations in the studied watershed are strongly influenced by storm events and the resulting input from riparian areas. The higher inorganic carbon level in the summer indicates increased metabolism which consumes oxygen. Although carbon is not classified as a classic nutrient like nitrogen or phosphorus, it does play a key role in nitrogen dynamics. High organic carbon is necessary in denitrification, which is becoming an important step in removing excess nitrate from nitrogen saturated forest ecosystems. Making carbon measurements a part of regular water-quality monitoring can give important insights into nitrogen dynamics as well as dissolved oxygen levels. This information can be useful for designing silvicultural practices that will conserve and maintain ecosystem carbon.

ACKNOWLEDGMENTS

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LITERATURE CITED

Aitkenhead-Peterson, J.A.; Smart, R.P.; Aitkenhead, M.J. [and others]. 2007. Spatial and temporal variation of dissolved organic carbon export from gauged and ungauged watersheds of Dee Valley, Scotland: effect of land cover and C:N. Water Resources Research. 43.

Buffam, I.; Galloway, J.N.; Blum, L.K.; McGlathery, K.J. 2001. A stormflow/baseflow comparison of dissolved organic matter concentrations and bioavailability in an Appalachian stream. Biogeochemistry. 53(3): 269–306.

Campbell, J.L.; Hornbeck, J.W.; McDowell, W.H. [and others]. 2000. Dissolved organic nitrogen budgets for upland, forested ecosystems in New England. Biogeochemistry. 49: 123–142.

Cooper, R.; Thoss, V.; Watson, H. 2006. Factors influencing the release of dissolved organic carbon and dissolved forms of nitrogen from a small upland headwater during autumn runoff events. Hydrological Processes. 21(5): 622–633.

Cory, R.M.; Green, S.A.; Pregitzer, K.S. 2004. Dissolved organic matter concentration and composition in the forests and streams of Olympic National Park, WA. Biogeochemistry. 67(3): 269–288.

Dalzell, B.J.; Filley, T.R.; Harbor, J.M. 2007. The role of hydrology in annual organic carbon loads and terrestrial organic matter export from a midwestern agricultural watershed. Geochimica Et Cosmochimica Acta. 71: 1448–1462.

del Giorgio, P.A.; Cole, J.J. 1998. Bacterial growth efficiency in natural aquatic systems. Annual Review of Ecology and Systematics. 29: 503–541.

Dosskey, M.G.; Bertsch, P.M. 1994. Forest sources and pathways of organic-matter transport to a blackwater stream—a hydrologic approach. Biogeochemistry. 24: 1–19.

Frost, P.C.; Larson, J.H.; Johnston, C.A. [and others]. 2006. Landscape predictors of stream dissolved organic matter concentration and physicochemistry in a Lake Superior river watershed. Aquatic Sciences. 68(1): 40–51.

Johnson, M.S.; Lehmann, J.; Couto, E.G. [and others]. 2006. DOC and DIC in flowpaths of Amazonian headwater catchments with hydrologically contrasting soils. Biogeochemistry. 81(1): 45–57.

Kalbitz, K.; Solinger, S.; Park, J.H. [and others]. 2000. Controls on the dynamics of 20 dissolved organic matter in soils: a review. Soil Science. 164(4): 277–304.

Knowles, R. 1982. Denitrification. Microbiological Reviews. 46: 43–70.

Louisiana Department of Environmental Quality. 2001 Flat Creek watershed TMDL for oxygen demand. Baton Rouge, LA: Department of Environmental Quality, Office of Environmental Assessment, Environmental Technology Division. http://www.deq.state.la.us/portal/portals/0/technology/tmdl/pdf/Flat-Creek-OD.pdf. [Date accessed: July 29, 2011].

Marschner, B.; Kalbitz, K. 2003. Controls of bioavailability and biodegradability of dissolved organic matter in soils. Geoderma. 113: 211–235.

Qualls, R.G.; Haines, B.L.; Swank, W.T. 1991. Fluxes of dissolved organic nutrients and humic substances in a deciduous forest. Ecology. 72(1): 254–266.

Royer, T.V.; David, M.B. 2005. Export of dissolved organic carbon from agricultural streams in Illinois, USA. Aquatic Sciences. 67: 465–471.

Saksa, P. 2007. The hydrology and sediment transport of low-gradient, forested headwater streams. Baton Rouge, LA: Louisiana State University. [Number of pages unknown]. M.S. thesis.

Saksa, P.; Xu, Y.J. 2006. Riverine nutrient inputs to Lake Ponchartrain, Louisiana, USA in coastal environment and water quality. In: Xu, Y.J.; Singh, V.J., eds. Coastal environment and water quality. Highlands Ranch, CO: Water Resources Publications: 195–204.

Seitzinger, S.P. 1988, Denitrification in freshwater and coastal marine ecosystems: ecological and geochemical significance. Limnology and Oceanography. 33: 702–724.

Sparks, D.L. 2003. Environmental soil chemistry. Amsterdam: Academic Press. 352 p.

Strauss, E.A.; Mitchell, N.L.; Lamberti, G.A. 2002. Factors regulating nitrification in aquatic sediments: effects of organic carbon, nitrogen availability, and pH. Canadian Journal of Fisheries & Aquatic Sciences. 59: 554–563.

Tranvik, L.R. 1992. Allochtonous dissolved organic matter as an energy source for pelagic bacteria and the concept of the microbial loop. Hydrobiologia. 229: 107–114.

AGROFORESTRY PLANTING DESIGN AFFECTS LOBLOLLY PINE GROWTH

D.M. Burner[1]

Abstract—The effect of plantation design on resource utilization has not been adequately investigated in agroforestry plantations. An experiment was conducted near Booneville, AR, on a silt loam soil with a fragipan. Loblolly pine (*Pinus taeda* L.) trees were planted in 1994 in three designs: two rows (1.2 by 2.4 m) with a 7.3-m alley, four rows (1.2 by 2.4 m) with a 12.2-m alley, and a rectangular 1.2- by 2.4-m configuration. Each 0.4-ha design was replicated three times. Height and d.b.h. were measured for 6 consecutive years (2002 to 2007) in 0.047-ha plots. Tree height increased annually from 7.30 m (2002) to 13.27 m (2007). For any given year, d.b.h. was greatest in the two-row design, and the four-row design had greater d.b.h. than the rectangular design in 2004 to 2007. Exterior rows in the four-row design had greater d.b.h. than interior rows. Depending on design, plantations might be useful for alley cropping, silvopasture, or pine straw.

INTRODUCTION

Agroforestry systems can help decrease financial risk and increase farm receipts through commodity diversification and the simultaneous production of food and fiber (Clason and Sharrow 2000, Pearson and others 1995). Pines, pastures, and cattle can be intentionally coproduced in an agroforestry system known as silvopasture. About 9 million ha in the South, much of it marginal crop and pastureland, could yield a greater economic return if planted to pine silvopastures (Haynes 1990).

Relatively few landowners in the United States employ silvopastoral practices, perhaps because there is poor understanding of the economics, marketing, and cost efficiencies involved with the production and sale of agroforestry products such as wood and pine straw (Pearson and others 1995). Further, the complex design and management of silvopastoral systems, compared to row crop, pine, or livestock monoculture, might constrain adoption of this technology. Pine straw production could be a financial incentive to landowners, especially if it occurred relatively early in the tree rotation (Moore and others 1996).

Information is needed on the appropriate design of conifer tree stands for agroforestry applications. Recommended densities for pine silvopastures are only broadly defined, ranging from 250 to 980 tree seedlings/ha, and are determined by tree crop and companion crop requirements, management objectives, and equipment constraints (Robinson and Clason 2000). The rapidity with which trees shade and impact herbage yield depends on tree species, initial row width, row orientation, site productivity, and subsequent thinning.

Loblolly pine (*Pinus taeda* L.) grows well when soil pH is between 4.5 and 6.0 (Schultz 1997), and silvopastures often are established on unfertilized sites with low herbage productivity (Pearson and others 1995). Fertilization usually enhances herbage and wood production in silvopastures (Clason 1999, Schultz 1997), even though many producers do not routinely fertilize their silvopastures (Morris and Clason 1997).

Pine spacing and silvopasture management are objective driven and site specific. The knowledge database needs to be expanded to enable growers to match silvopasture design and management to specific growing conditions, objectives, and budget. The objective of this study was to determine if plantation design affected loblolly pine height and d.b.h. growth.

METHODS

The experiment was conducted near Booneville, AR, on a Leadvale silt loam soil (fine-silty, siliceous, semiactive, thermic Typic Fragiudults). The site has a fragipan at 40 to 60 cm depth (Burner and MacKown 2005). Loblolly pine trees were planted in 1994 in an east-west row orientation in three designs: two rows (1.2 by 2.4 m) with a 7.3-m alley, four rows (1.2 by 2.4 m) with a 12.2-m alley, and a rectangular 1.2- by 2.4-m configuration. These will be subsequently referred to as two-row, four-row, and rectangular designs, respectively. Each 0.4-ha design was replicated three times.

The alley understory in the two- and four-row designs contained mainly tall fescue (*Lolium arundinaceum*) and bermudagrass (*Cynodon dactylon*), but due to a closed canopy there was essentially no understory vegetation in the rectangular design. The 7.3- and 12.2-m alleys in the two- and four-row treatments, respectively, were mowed and/or surface cultivated to 15-cm depth once or twice annually during the study period (2002 to 2007) to emulate a silvopastoral practice. Surface (15-cm depth) tillage was used to establish an annual ryegrass (*L. multiflorum*) cover crop without fertilization. The 2.4-m wide alleys in two- and four-row designs were vegetated with tall fescue and bermudagrass, and this understory was undisturbed throughout the study period. The rectangular design received no tillage or fertilization treatments, but yield of “red” pine straw was

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estimated at 7500 kg/ha from nonreplicated, hand-raked samples collected in 2005 and 2006.

Four contiguous tree rows were randomly selected within each plot (avoiding exterior rows in the rectangular design). Selected rows were randomly partitioned into two subplots (east and west one-half of the plot). The entire subplot received one of two pruning treatments in 2002: pruning to a height of 2 m (about 25 percent of the total tree height), or not pruned. Pruning debris was left onsite. Twenty dominant or codominant trees (about every other tree) within a row were marked with a numbered tag. Plots represented a 0.047-ha sample size. Height of every second tagged tree was measured with a clinometer, and d.b.h. of every tagged tree was measured with a d-tape. Data were collected for 6 years (2002 to 2007). Climatic data (minimum, maximum, and mean air temperatures, and rainfall) for 2002 to 2007 were obtained from National Oceanic and Atmospheric Administration (NOAA) (2002b, 2003) and from a nonofficial weather station (Model 900, Spectrum Technologies Inc., Plainfield, IL) located 1 km west of the study location, and data were compared to long-term (1971 to 2000) means (National Oceanic and Atmospheric Administration 2002a).

Analysis of variance of d.b.h. and height data used a mixed linear model, PROC MIXED (SAS Institute 2002). Fixed effects were year, design, pruning, row location (north, middle, or south, depending on the design), and interactions. Pruning was not a significant effect ($P > 0.07$) presumably because few live branches were removed, so this treatment was not included in the full model. Replication was the random effect. Tree within year, replication, and design was the repeated measure with a Toeplitz covariance structure and restricted maximum likelihood estimation method (SAS Institute 2002). Degrees of freedom were calculated by the Satterthwaite approximation method. Means were considered different at $P < 0.05$ using the Tukey honestly significant difference test.

RESULTS AND DISCUSSION

Tree counts at planting were estimated at 1,540 trees/ha (two- and four-row designs), and 3,340 trees/ha (rectangular design), at least twice the conservative rate (850 trees/ha) recommended for forestry plantations (South 2003). An estimated 75 to 90 percent of trees (40 to 46 trees per row) were alive in 2007 with no apparent difference in survival between designs. Survival was consistent with predicted estimates (Schultz 1997).

Mean annual air temperatures during the study period (table 1) tended to be cooler than the long-term mean (National Oceanic and Atmospheric Administration 2002a) but were within the suitable range for loblolly pine in the Southeastern United States. There were large annual fluctuations in total rainfall during the study period, and the mean (1063 mm) was 12 percent less than the long-term mean (National Oceanic and Atmospheric Administration 2002a). Rainfall during some years of the study was less than adequate (1020 mm) (Schultz 1997). An adjacent stand on the same soil with 995 trees/ha had more soil water during the growing season of 2002 (wetter year) than 2003 (drier year), and the rate of soil water depletion in 2003 was more rapid than that of a meadow (Burner and MacKown 2005). Tree growth of all plantations probably was constrained with respect to soil water availability, due to the fragipan, especially in years with below average rainfall. Further, annual cultivation of the 7.3- and 12.2-m wide alleys of agroforestry plantations also could have differentially impacted tree roots, water uptake, and growth compared to the rectangular configuration.

Table 1—Mean annual air temperature and total rainfall for Booneville, AR, for 2002 through 2007

Year[a]	Air temperature			Rainfall
	Minimum	Maximum	Mean	
	---------- °C ----------			*mm*
2002	n/a[b]	n/a	15.8	1393
2003	n/a	n/a	15.9	737[c]
2004	8.3	22.8	15.4	1279
2005	7.9	23.9	15.6	764
2006	8.0	24.1	15.9	1213
2007	8.4	23.2	15.5	994
Mean[d]	9.9	23.0	16.5	1214

[a] Data for 2002 and 2003 were from National Oceanic and Atmospheric Administration (2002b, 2003). Data for 2004 to 2007 were from a nonofficial weather station located 1 km west of the study location.

[b] Data not reported due to missing values.

[c] Some data were missing.

[d] Long-term (1971 to 2000) mean air temperature or total rainfall (National Oceanic and Atmospheric Administration 2002a).

Height

There was a year × design effect for height ($P < 0.001$) (table 2), but designs did not differ ($P > 0.28$) in height within any given year. Tree height increased annually ($P = 0.001$) from 7.30 m in 2002 to 13.27 m in 2007 (fig. 1). Tree height was not affected by the row location × design interaction ($P = 0.51$).

Diameter at Breast Height

There was a year × design effect for d.b.h. ($P < 0.001$). For any given year (fig. 2), tree d.b.h. was greater in the two-row than the four-row design ($P < 0.05$), and the four-row design had greater d,b,h, than the rectangular design in 2004 to 2007 ($P < 0.05$). Row location did not have an effect on d.b.h. in two-row and rectangular designs ($P > 0.98$), but exterior rows in the four-row design had larger d.b.h. than interior rows ($P < 0.01$, data not shown).

Table 2—Analysis of variance of height and d.b.h. for loblolly pine in three planting designs at Booneville, AR

	Height			D.b.h.	
Source of variation	*DF*	*F*-value	*Pr* > *F*	*F*-value	*Pr* > *F*
Year (Y)	5	3429.30	<0.001	8704.83	<0.001
Design (D)	2	0.29	0.761	32.13	0.001
Y × D	10	22.34	<0.001	47.46	<0.001
Row (R)	3	5.00	0.002	2.79	0.040
Y × R	15	2.73	<0.001	7.17	<0.001
D × R	6	0.88	0.510	5.05	<0.001
Y × D × R	30	1.53	0.033	9.89	<0.001

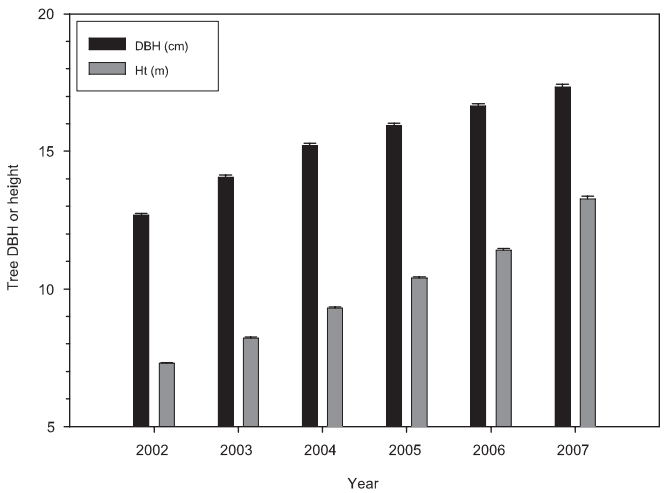

Figure 1—Mean height and d.b.h. of loblolly pine in three agroforestry designs at Booneville, AR. Small vertical bars which exceed the line width indicate standard errors (n = 360 and 720 for height and d.b.h., respectively). Years differ in d.b.h. and height (P < 0.05).

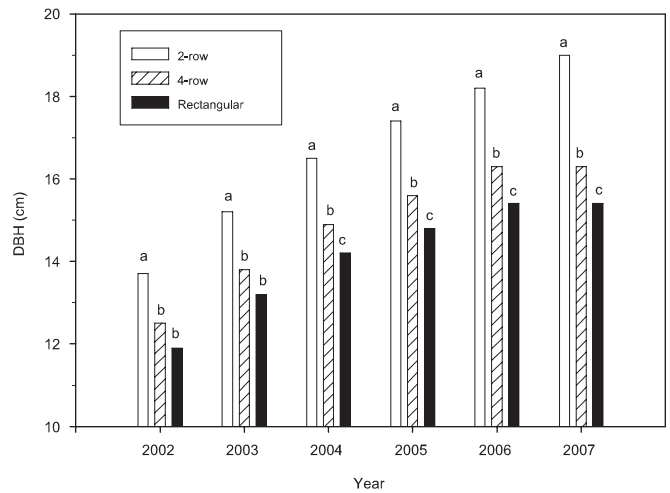

Figure 2—Effect of the year × design interaction on mean d.b.h. of loblolly pine grown in agroforestry plantations at Booneville, AR. Bars within a year having a common letter do not differ (P > 0.05).

CONCLUSIONS

Tree d.b.h. growth was greater in the two-row than four-row agroforestry design, and d.b.h. growth in either agroforestry design usually surpassed that of the rectangular design. This growth difference occurred even though tree roots in the 7.3- and 12.2-m wide alleys of agroforestry plantations might have been disturbed by cultivation. Each of these plantation designs has potential application for fiber, alley crop, or cattle production. While d.b.h. in the rectangular design was constrained by overstocking, this design could be used for pine straw production. The 7.3- and 12.2-m wide alleys in two- and four-row designs might be useful for alley cropping or silvopasture. Selection of the "best" design would depend on production objectives.

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LITERATURE CITED

Burner, D.M.; MacKown, C.T. 2005. Herbage nitrogen recovery in a meadow and loblolly pine alley. Crop Science. 45: 1817–1825.

Clason, T.R. 1999. Silvopastoral practices sustain timber and forage production in commercial loblolly pine plantations of northwest Louisiana, USA. Agroforestry Systems. 44: 293–303.

Clason, T.R.; Sharrow, S.H. 2000. Silvopastoral practices. In: Garrett, H.E. [and others], eds. North American agroforestry. Madison, WI: American Society of Agronomy: 119–147.

Haynes, R.W., coord. 1990. An analysis of the timber situation in the United States: 1989–2040—a technical document supporting the 1989 USDA Forest Service RPA assessment. Gen. Tech. Rep. RM-199. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Forest and Range Experiment Station. 268 p.

Moore, B.J.; Roth, F.A., II; Pearson, H.A.; Haywood, J.D. 1996. Pine straw harvesting: a new Arkansas agricultural enterprise. Publ. MP382-2M-9-96. Little Rock, AR: University of Arkansas, Cooperative Extension Service. 8 p.

Morris, D.R.; Clason, T.R. 1997. Summer forage production under loblolly pine stands. Communications in Soil Science and Plant Analysis. 28: 717–726.

National Oceanic and Atmospheric Administration. 2002a. Climatography of the United States no. 81, Arkansas. Asheville, NC: National Climatic Data Center. 26 p.

National Oceanic and Atmospheric Administration. 2002b. Climatological data. Annual summary for Arkansas 2002. Asheville, NC: National Climatic Data Center. 107(13). 25 p.

National Oceanic and Atmospheric Administration. 2003. Climatological data. Annual summary for Arkansas 2003. Asheville, NC: National Climatic Data Center. 108(13). 26 p.

Pearson, H.A.; Knowles, R.L.; Middlemiss, P.G. [and others]. 1995. United States agroforestry estate model. Compiler. 13: 27–37.

Robinson, J.L.; Clason, T. 2000. From a pasture to a silvopasture system. AF Note 22. [Place of publication unknown]: U.S. Department of Agriculture, National Agroforestry Center. 4 p.

SAS Institute Inc. 2002. SAS/STAT user's guide. Release 9.1. Windows version 5.1.2600. Cary, NC: SAS Institute Inc.

Schultz, R.P. 1997. Loblolly pine. Agric. Handb. 713. Washington, DC: U.S. Government Printing Office. 493 p.

South, D.B. 2003. "Correct" planting density for loblolly pine depends on your objectives and who you ask. Forest Landowner Manual. 34: 46–51.

CONVERSION OF AN OAK SEED ORCHARD TO OAK SILVOPASTURE

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Abstract—The potential of hardwood silvopasture has yet to be realized in the Southeastern United States. The decommissioning of the Stauffer Nursery, Opelika, AL, provided the opportunity to intensively research hardwood silvopasture using various oak species. Average crown diameter ranged from 5.9 feet in white oak (*Quercus alba*) to 10.7 feet in Nuttall oak (*Q. nuttallii* Palmer). Nuttall oak trees had significantly larger diameters, greater heights, and clear stem lengths than any of the other measured species, while white oak trees have the lowest values. Willow oak (*Q. phellos* L.) and cherrybark oak (*Q. pagoda* Raf.) averages are comparable.

INTRODUCTION

Land ownership patterns have shifted dramatically in the Southern United States. Large tracts of forested acreage have changed hands (Wear and Greis 2002), and long-term ownership may no longer be the norm (Clutter and others 2007, Wear 2007, Wear and others 2007). Changes in population growth and demographic patterns, i.e., urbanization, within the Southeast have resulted in subsequent changes in land valuations, with traditional forestry often a poor second to real estate development. Forest landowners are increasingly diverse in expectations of forest lands and forest experiences. These changes impact the ability of State and Federal Agencies to provide southern forest landowners and managers with viable land management alternatives and income from forests.

Many forested acres in the United States are stocked with an overabundance of small-diameter trees (Compass Magazine 2005). In areas in the Southeast where row cropping, grazing, or pulpwood markets are no longer viable or where there is a threat of tree insect and disease epidemics, silvopasture is an increasingly attractive land use alternative. Silvopasture combines intensively managed timber with pasturing and offers the opportunity to simultaneously accrue multiple benefits such as high-value timber products, livestock forage, wildlife habitat, other agricultural crops, and biofuel crops.

The potential of hardwood silvopasture has yet to be realized in the Southeastern United States. Whereas the concepts are not new, hardwood silvopasture is little used in this region. These systems require intensive management to maximize economic returns (Garrett and others 2004), and, in the Southern United States, such investments are usually focused on plantations of loblolly (*Pinus taeda* L.) and slash pine (*P. elliottii* Engelm.) which in the past have realized a respectable return on investment. However, the overabundance of high-density, low-quality loblolly pine stands may provide managers seeking alternate income sources an unprecedented opportunity to convert such stands to pine silvopasture through intensive thinning or to hardwood silvopasture on lands less suited to high-quality softwood production. Fike and others (2004) stress the importance of choosing species that are marketable, have high-quality wood and rapid growth, that are deep rooted and drought tolerant, and that produce additional products, such as nuts.

The decommissioning of one of Alabama's State nurseries, the Stauffer Nursery, in Opelika, AL, has provided a chance to intensively research hardwood silvopasture using various oak species. Many species of southern oaks have high-quality wood in addition to producing hard mast, a valuable food for wildlife. This makes the oaks a potentially good choice for establishing hardwood silvopasture. The objective of this study was to demonstrate silvopasture alternatives for managers and private landowners in hardwood systems, using various oak species.

MATERIALS AND METHODS

The study site at Stauffer Nursery was planted 10 to13 years ago with various oak species on a 30- by 30-foot spacing, now ideal for studying hardwood silvopasture systems. No one species was planted in a single year; rather, over the 3-year span, individuals selected from the wild were dug up and planted as mother trees. The planted species are valuable oaks, namely Nuttall oak (*Quercus nuttallii* Palmer) or Texas red oak (*Q. texana* Buckley), willow oak (*Q. phellos* L.), cherrybark oak (*Q. pagoda* Raf.), white oak (*Q. alba* L.), Shumard oak (*Q. shumardii* Buckley var. *shumardii*), and swamp chestnut oak (*Q. michauxii* Nutt.). All trees were planted in separate blocks and there are no replications, making the results applicable only to this particular site. We measured tree height, diameter at breast height (d.b.h.), number of epicormic branches over 3/8 inch in diameter, average clear stem, live crown ratio, and crown diameter in two directions (north and west). Live crown ratio is the live crown length divided by total tree height. Simple t-tests were performed to make between-species comparisons of the measured variable means. Future plans include determining log value of trees, influence of cattle on discoloration of sapwood, and use of the system by wildlife.

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Table 1—Average tree parameters for the studied species at Stauffer Nursery

Oak species	Trees	D.b.h.	Height	Clear stem	Crown diameter	Live crown ratio
	number	*inches*	---------- *feet* ----------			
Nuttall	132	8.9	34.6	8.6	10.7	0.7
Willow	163	7.1	26.8	6.8	9.3	0.7
Cherrybark	136	6.7	26.0	6.7	8.2	0.7
White	127	4.8	18.4	3.8	5.9	0.8
Shumard	104	6.1	24.4	4.4	7.7	0.8
Swamp chestnut	104	5.9	21.3	3.9	7.4	0.8

D.b.h. = diameter at breast height.

Table 2—T-test probability values for d.b.h. of six oak species at Stauffer Nursery

Oak species	Willow	Cherrybark	White	Shumard	Swamp chestnut
Nuttall	0.0001	0.0001	0.0001	0.0001	0.0001
Willow		0.1146	0.0001	0.0003	0.0001
Cherrybark			0.0001	0.0175	0.0001
White				0.0001	0.0001
Shumard					0.2767

Table 3—T-test probability values for height of six oak species at Stauffer Nursery

Oak species	Willow	Cherrybark	White	Shumard	Swamp chestnut
Nuttall	0.0001	0.0001	0.0001	0.0001	0.0001
Willow		0.1885	0.0001	0.0004	0.0001
Cherrybark			0.0001	0.0009	0.0001
White				0.0001	0.0001
Shumard					0.0001

Table 4—T-test probability values for clear stem length of six oak species at Stauffer Nursery

Oak species	Willow	Cherrybark	White	Shumard	Swamp chestnut
Nuttall	0.0001	0.0001	0.0001	0.0001	0.0001
Willow		0.6054	0.0001	0.0001	0.0001
Cherrybark			0.0001	0.0001	0.0001
White				0.0001	0.1026
Shumard					0.0001

RESULTS AND DISCUSSION

Six species of oaks were measured for baseline information at the Stauffer Nursery. Average crown diameter ranged from 5.9 feet in white oak to 10.7 feet in Nuttall oak (table 1). Epicormic branches were recorded for only three of the species, Nuttall oak (2.6 per tree), willow oak (0.6 per tree), and cherrybark oak (0.6 per tree), as they were the only species that had been pruned. Nuttall oak trees had significantly larger d.b.h., greater heights, and clear stem lengths than any of the other measured species (tables 2, 3, and 4; $P \leq 0.05$), while white oak trees have the lowest values. Willow and cherrybark oak averages are comparable.

A well-managed silvopasture system provides economic and environmental benefits to the landowner and to society (Shrestha and Alavalapati 2004). While the trees themselves are a long-term investment if utilized for timber, the livestock, understory crops, nuts, and wildlife can provide more constant, reliable, and immediate sources of income for the landowner. The potential for using a silvopasture system to produce biofuels and/or carbon credits has yet to be determined but will no doubt be thoroughly researched by those engaged in conserving open spaces and productive land resources for future generations. Future research at the Stauffer Nursery has the potential to provide information to landowners and natural resource managers on alternative forest management strategies to meet the changing land use objectives of today's forest owner.

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LITERATURE CITED

Clutter, M.; Mendell, B.; Newman, D. [and others]. 2007. Strategic factors driving timberland ownership changes in the US South. Working Pap. http://www.srs.fs.usda.gov/econ/pubs/southernmarkets/. [Date accessed: July 29, 2011].

Compass Magazine 2005. The biomass challenge. http://www.srs.fs.usda.gov/compass/winter2005. [Date accessed: July 29, 2011].

Fike, J.H.; Buergler, A.L.; Burger, J.A.; Kallenbach, R.L. 2004. Considerations for establishing and managing silvopastures. Plant Management Network. http://www.plantmanagementnetwork.org/pub/fg/review/2004/silvo/. [Date accessed: December 2008]. 12 p.

Garrett, H.E.; Kerley, M.S.; Ladyman, K.P. [and others]. 2004. Hardwood silvopasture management. In: Nair, P.K.R.; Rao, M.R.; Buck, L.E., eds. New vistas in agroforestry: a compendium for the 1st world congress of agroforestry. Dordrecht, Kluwer: 21–33.

Shrestha, R.K.; Alavalapati, J.R.R. 2004. Valuing environmental benefits of silvopasture practice: a case study of the Lake Okeechobee watershed in Florida. Ecological Economics. 49(3): 349–359.

Wear, D.N. 2007. Trends in southeastern forests. http://www.serdp-estcp.org/workshops/serps/doscs/Trends_in_Southeastern_Forests.pdf. [Date accessed: July 29, 2011].

Wear, D.N.; Carter, D.R.; Prestemon, J. 2007. The U.S. South's timber sector in 2005. A prospective analysis of recent change. GTR-SRS-99. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 29 p.

Wear, D.N.; Greis, J.G. 2002. Southern Forest Resource Assessment: summary report. Gen. Tech. Rep. SRS-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 103 p.

INTENSIVE STRAW HARVESTING, FERTILIZATION, AND FERTILIZER SOURCE AFFECT NITROGEN MINERALIZATION AND SOIL LABILE CARBON OF A LOBLOLLY PINE PLANTATION

K. Ellum, H.O. Liechty, and M.A. Blazier[1]

Abstract—Straw harvesting can supplement traditional revenues generated by loblolly pine (*Pinus taeda* L.) plantation management. However, repeated raking may alter soil properties and nutrition. In northcentral Louisiana, a study was conducted to evaluate the long-term effects of intensive straw raking and fertilizer source (inorganic or organic) on nitrogen (N) cycling and soil carbon (C) quality. Monthly *in situ* soil N mineralization, total N and C concentrations, and end-of-season soil labile C concentrations were measured in response to: (1) annual straw raking for 4 years, (2) annual straw raking and fertilization with inorganic fertilizer for 4 years, and (3) annual straw raking and fertilization with broiler poultry litter application for 4 years. Straw raking led to significant increases in N mineralization. Significant increases in N mineralization larger than those from straw raking alone occurred in response to both fertilization treatments, whereas N was immobilized in response to unfertilized treatments. Applying poultry litter annually to raked soil increased soil C, N, and labile C concentrations and thus reduced the soil C to N ratio. However, application of poultry litter did not raise N mineralization above that found in response to fertilization with inorganic fertilizer. Total soil N concentrations were highest in response to poultry litter application, suggesting that N applied with this fertilizer source was better retained within soil than with inorganic fertilizer. Application of inorganic fertilizer reduced the potential C turnover rate to levels below those of all other treatments. Poultry litter may be superior to inorganic fertilizer in maintaining nutrition of frequently raked loblolly pine plantations because it more readily increased soil N availability and labile C critical in soil nutrient turnover. Inorganic fertilizer, by contrast, increased the potential turnover of C in the soil.

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Intensive Straw Harvesting, Fertilization, and Fertilizer Source Affect Nitrogen Mineralization and Soil Labile Carbon of a Loblolly Pine Plantation

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Abstract

Straw harvesting can supplement traditional revenues generated by loblolly pine (*Pinus taeda*) plantation management. However, repeated raking may alter soil properties and nutrition. A previous study documented altered bulk densities and microbial populations related to straw harvesting and different fertilizer applications. Another study at this same site was initiated to evaluate the long-term effects of intensive straw raking and fertilizer source (inorganic or organic) on nitrogen cycling and soil carbon quality. Monthly in situ soil nitrogen mineralization and end-of-season soil labile C concentrations were measured in response to: (1) annual straw raking for four years, (2) annual straw raking and fertilization with inorganic fertilizer for four years, and (3) annual straw raking and fertilization with broiler poultry litter application for four years. Results of these analyses will be presented and related to the potential sustainability of these differing treatments.

Introduction

Pine straw (pine needles) harvested for mulch from pine plantations is a valuable commodity in the southeastern U.S. It is marketed to the landscaping industry and represents a multimillion dollar industry in this region. There is concern that excessive removal of pine needles and repeated trafficking with heavy equipment may have adverse affects on the sustainability of soils and the long term productivity of these stands. The nutrient content in pine needles is substantial and repetitive harvesting of pine straw removes significant amounts of organic nitrogen and carbon as well as other nutrients from the soil. This may also reduce nitrogen availability which would ultimately reduce tree productivity. Nutrient amendments are commonly utilized in southern pine plantations to replenish macro and micronutrients that are essential for tree growth and are limiting. Both inorganic and organic fertilizers (such as poultry litter) have been used to increase nutrient availability in and increase productivity of pine plantations harvested for pine straw. Poultry litter is an important by-product of broiler poultry production and consists of chicken manure, bedding materials (rice or peanut hulls and pine shavings) and feed waste. Poultry litter differs from inorganic fertilizer in that it adds organic matter along with nitrogen and phosphorus. This additional carbon source may be important for sustaining soil quality in stands where organic matter is removed by raking. The fertilizer source applied to pine plantations in order to achieve desired soil conditions will be crucial in maintaining the long term productivity and sustainability of pine plantations managed for pine straw raking.

Objectives

The purpose of this study is to examine the effects of pine straw raking and organic and inorganic fertilizer amendments on:

1. Nitrogen mineralization
2. Soil carbon and nitrogen

Figure 1. Study location in Ouachita Parish, Calhoun, LA

Treatment Regimes

- CONTROL - No pine straw raking and no fertilizer
- RAKE - Pine straw raking* with no fertilizer
- RAKE-IN - Pine straw raking* with 308 kg ha^{-1} urea‡, 280 kg ha^{-1} DAP‡
- RAKE-PL - Pine straw raking* with poultry litter‡, 7.7 Mg ha^{-1}

*Pine straw raking occurred three times annually in Feb., Aug., and Nov.
‡Annual fertilizer treatments were respectively applied at total N and P input rates of 193 kg ha^{-1} and 129 kg ha^{-1} beginning April 2003.

Methods

- Eight 0.08 ha plots located in each of two stands
- Four replications of each treatment regime
- Potential nitrogen mineralization determined by in situ soil incubations using the buried bag method during March 2007
- Labile carbon samples collected during seasonal peak of pine and understory growth, September 2007
- Labile carbon quantified using the sequential fumigation-incubation procedure
- Total carbon and total nitrogen quantified using the dry combustion method

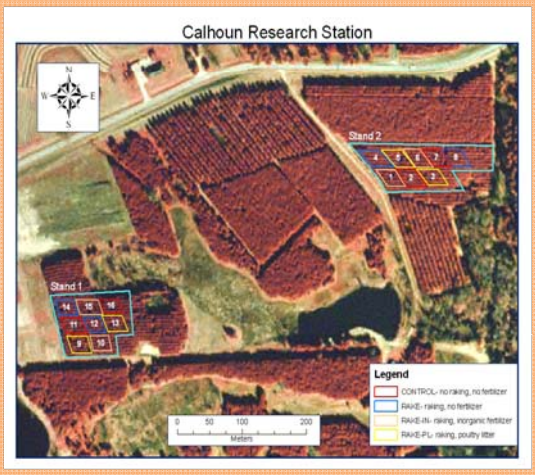
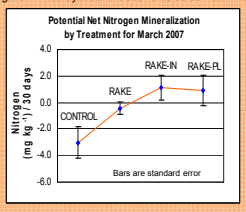
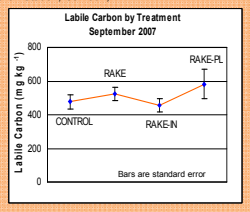
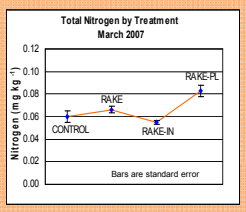
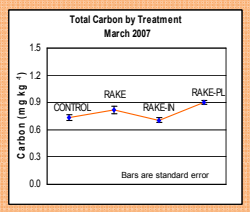


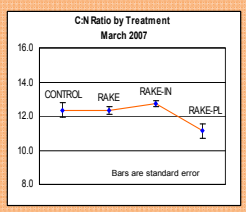
Figure 2. Plot layout and treatments at Calhoun Research Station, Calhoun, LA

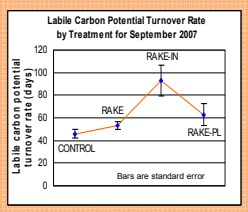












Preliminary Results

- Nitrogen mineralization occurred in fertilized treatments; mineral nitrogen became immobilized in unfertilized treatments.
- RAKE-IN treatments had concentrations of N, C, and labile carbon similar to those of the rake only and control treatments. The higher N mineralization in the RAKE-IN treatment appears to be related to N, C, and labile carbon.
- Poultry litter adds C, labile C, N and thus lowers C:N but does not appear to improve N mineralization over the RAKE-IN treatment.
- Reduced labile carbon in RAKE-IN increased the carbon turnover rate.
- Added N appears to be retained better with poultry litter
- Pine straw raking increases N mineralization

Conclusions

Fertilizer amendment is recommended in stands that are intensively raked for pine straw. Availability of mineral nitrogen that is important for productive tree growth can be improved with the addition of either organic or inorganic fertilizer. Nitrogen additions may be best retained with poultry litter resulting in greater nitrogen availability. Increased N mineralization following pine straw raking is perhaps due to increased soil temperature and may only be temporary. Soil labile carbon concentrations can be maintained in raked stands with organic or inorganic fertilizer sources but poultry litter increases soil labile carbon levels which may improve stand productivity. Soil productivity may be best sustained with poultry litter amendment due to the shorter labile carbon turnover rate. This short turnover rate may improve nitrogen mineralization rates, availability of nitrogen, and thus stand productivity. Poultry litter may be a preferred fertilizer source over inorganic fertilizer for replenishing nutrient resources and sustaining labile carbon levels in loblolly pine plantations intensively raked for pine straw production. This is important for the long-term sustainability and productivity of soil resources. The nutrient management applied to pine plantations in order to achieve desired soil conditions will be crucial in maintaining the long term productivity and sustainability of pine plantations managed for pine straw raking.

Acknowledgements

We thank the Arkansas Forest Resources Center and the LSU AgCenter for providing research funding.

FORESTED COMMUNITIES OF THE PINE MOUNTAIN REGION, GEORGIA, USA

Robert Floyd and Robert Carter[1]

Abstract—Seven landscape scale communities were identified in the Pine Mountain region having a mixture of Appalachian, Piedmont, and Coastal Plain species. The diagnostic environmental variables included elevation, B-horizon depth, A-horizon silt, topographic relative moisture index, and A-horizon potassium (K).

INTRODUCTION

The Pine Mountains of Georgia have been an area of considerable botanical interest since Harper (1903) first visited the region in 1901. To date, the only significant floristic survey performed in the region was by Jones (1974) in which he noted a unique assemblage of Appalachian and Coastal Plain species. However, no study of the plant communities has been performed. The objective of the study was to identify landscape scale plant communities based on the discriminating vegetation, soils, and landform variables.

METHODS

The study area was the Pine Mountain region in Upson, Meriwether, and Talbot Counties, GA. The elevation ranges from 200 to 347 m. The area is characterized by steep rocky slopes. Between 2003 and 2008, 45 plots were established in suitable forested sites. Vegetation was sampled following the Carolina Vegetation Survey protocol (Peet and others 1998). Soil samples were collected by horizon to determine soil horizon depths and chemical and textural properties. Landform variables sampled included slope, slope position, aspect, and landform index (LFI). The topographic moisture index (Parker 1982) was calculated based on percent slope, aspect, slope position, and site concavity/convexity. A topographic relative moisture index (TRMI) of zero indicates a xeric site, while 60 indicates a mesic site.

Communities were delineated through ordination (detrended correspondence analysis, canonical correspondence analysis) and cluster analysis (TWINSPAN) of importance value data (Hill 1979, McCune and Grace 2002). Environmental variables were related to the ecological units through stepwise discriminant analysis ($P = 0.10$).

RESULTS AND DISCUSSION

Seven communities were identified. All communities had some degree of homogenization of Coastal Plain, Piedmont, and Appalachian flora based on Duncan and Kartesz (1981). Communities with more Coastal Plain affinities included longleaf pine (*Pinus palustris*)-turkey oak (*Quercus laevis*)-Goat's Rue (*Tephrosia virginiana*) and sweetgum (*Liquidambar styraciflua*)-cinnamon fern (*Osmunda cinnamomea*)-laurel greenbrier (*Smilax laurifolia*). The rhododendron (*Rhododendron maximum* L.)-heartleaf (*Hexastylis arifolia*)-mountain laurel (*Kalmia latifolia*) and chestnut oak (*Q. prinus*)-longleaf pine-downy milkpea (*Galactia volubilis*) communities had stronger Appalachian affinity. Sand hickory (*Carya pallida*)-Alabama cherry (*Prunus alabamensis*)-broomsedge (*Andropogon virginicus*) community exhibited more Piedmont affinities. The hickory (*Carya*)-rusty blackhaw (*Viburnum rufidulum*)-ebony spleenwort (*Asplenium platyneuron*) and mountain laurel-longleaf pine-slender lespedeza (*Lespedeza virginica*) communities possessed mixed affinity for Appalachian, Coastal Plain, and Piedmont flora (table 1). The five most discriminating abiotic variables were elevation, B-horizon depth, TRMI, B-horizon silt, and A-horizon K.

ACKNOWLEDGMENTS

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[1] Graduate Student and Associate Professor, Jacksonville State University, Department of Biology, Jacksonville, AL, respectively.

Table 1—Community type, habitat, and diagnostic species for the Pine Mountain region, Georgia

	Community type			
	Rhododendron Heartleaf Mountain laurel	Mountain laurel Longleaf pine Slender lespedeza	Longleaf pine Turkey oak Goat's rue	Sand hickory Alabama cherry Broomsedge
	Habitat[a]			
Diagnostic species	Steep slopes bordering stream	Steep slopes bordering Flint River	Steep rocky upper slopes with low K	Mountain tops and slopes with low K
Magnolia pyramidata 1	X			
Rhododendron minus 2	X	X		
Hexastylis arifolia 4	X			
Lespedeza virginica 4	X	X	X	
Kalmia latifolia 2	X	X		
Dichanthelium boscii 4		X		
Erigeron pulchellus 4		X		
Eupatorium hyssopifolium 4		X		
Quercus laevis 1			X	
Pinus palustris 1	X	X	X	X
Pteridium aquilinum 4		X	X	X
Tephrosia virginiana 4		X	X	X
Galactia volubilis 4			X	X
Andropogon virginicus 4		X	X	X
Prunus alabamensis 1		X	X	X
Quercus prinus 1	X	X	X	X
Carya pallida 1			X	X
Danthonia sericea 4				X
Liquidambar styraciflua 1	X			
Liriodendron tulipifera 1	X			

	Community type		
	Chestnut oak Longleaf pine Downy milkpea	Hickory Rusty blackhaw Ebony spleenwort	Sweetgum Cinnamon fern Laurel greenbrier
	Habitat[a]		
Diagnostic species	Steep rocky midslopes	Mountain tops moist side slopes with high K	Moist sites near streams and springs
Dichanthelium boscii 4		X	X
Pinus palustris 1	X		
Pteridium aquilinum 4	X		X
Tephrosia virginiana 4	X		
Galactia volubilis 4	X	X	X
Andropogon virginicus 4	X		
Prunus alabamensis 1	X	X	
Quercus prinus 1	X	X	
Carya pallida 1	X	X	X
Danthonia sericea 4	X		
Carya glabra 1		X	
Asplenium platyneuron 4		X	
Aesculus pavia 2	X		
Viburnum rufidulum 1		X	
Osmunda cinnamomea 4			X
Arundinaria gigantea 4			X
Smilax laurifolia 4			X
Mitchella repens 4			X
Liquidambar styraciflua 1			X
Liriodendron tulipifera 1			X

[a] 1, 2, 3, 4, indicate tree, sapling, seedling, and herb, respectively.

LITERATURE CITED

Duncan, W.H.; Kartesz, J.T. 1981. Vascular flora of Georgia: an annotated checklist. Athens, GA: The University of Georgia Press. 143 p.

Harper, R.M. 1903. Botanical explorations in Georgia during the summer of 1901. Bulletin of the Torrey Botanical Club. 30: 282–295.

Hill, M.O. 1979. TWINSPAN: a FORTRAN program for arranging multivariate data in an ordered two-way table by classification of the individuals and attributes. Ithaca, NY: Cornell University, Section of Ecology and Systematics. 90 p.

Jones, S.B., Jr. 1974. The flora and phytogeography of the Pine Mountain region of Georgia. Castanea. 39: 117–149.

McCune, B.; Grace, J.B. 2002. Analysis of ecological communities. Gleneden Beach, OR: MjM Software Design. 300 p.

Parker, A.J. 1982. The topographic relative moisture index: an approach to soil-moisture assessment in mountain terrain. Physical Geography. 3(2): 160–168.

Peet, R.K.; Wentworth, T.R.; White, P.S. 1998. A flexible multipurpose method for recording vegetation composition and structure. Castanea. 63: 262–274.

THE LONGEST ACTIVE THINNED AND PRUNED LOBLOLLY PINE PERMANENT PLOTS: THE LAST MEASUREMENT

Denise R. General, Curtis L. VanderSchaaf, and B. Zeide[1]

Abstract—The longest active study of the effects of thinning and pruning on growth of loblolly pine (*Pinus taeda* L.) was established by Dr. James D. Burton in 1970 in a typical 12-year-old loblolly pine (plantation was 11 years old) stand planted by then the Georgia-Pacific Corporation in the southeastern corner of Arkansas. Basal area has been maintained at 30, 50, 70, and 90 square feet per acre by periodic thinnings. Within each level of basal area plots were pruned in two stages at 12 and 15 years, finally clearing the bole to heights of 33, 26, and 22 feet and reducing crown lengths to 25, 40, and 50 percent of the total tree height, respectively. Five control plots (without thinning or pruning) were installed at age 27. The 50-year remeasurement in the fall of 2007 will be the last since the study will be clearcut. Observed stand dynamics by thinning and pruning levels show that unthinned plots produced the maximum standing volume while moderate pruning did not substantially affect volume.

INTRODUCTION

Loblolly pine (*Pinus taeda* L.) has been one of the most widely planted tree species in the Southeastern United States, and its growth and yield has been extensively studied. Several long-term studies have been established across the Southeastern United States examining the yields of loblolly pine (e.g., Baldwin and others 2000, Sharma and others 2002). One of the longest continual studies was established in southeastern Arkansas and is referred to as the Monticello Thinning and Pruning Study. This study has been scheduled for termination and harvesting in the fall of 2008. Plot-level summary measures observed from this study beginning at seed age 12 (year 1970—plantation age 11 since 1-0 seedlings were planted) until the final inventory at seed age 50 (year 2007) are reported in this paper. All further references to age refer to seed age.

OBJECTIVES

This study was initialized to: (1) determine the optimal level of stand density to maximize stand productivity (intermediate and final harvest) and (2) determine the effect of pruning on long-term productivity and quality of wood.

METHODS

Study Site Description

The stand was established in the winter of 1958 to 1959 in a row-cropped old field at a spacing of 8 feet by 8 feet using 1-0 seedlings obtained from a State nursery located in Arkansas. Genetic stock was of a local seed source. Plots were originally established in 1970 when the trees were 12 years old. Four levels of thinning, three levels of pruning, and all their combinations were included in the study design. Each combination had three replications within a randomized complete block design. Four plots were also established for each of the 4 thinning treatments without pruning for a total of 40 plots. Each plot had a gross size of 132 by 132 feet and contained an inner plot 66 by 66 feet where all trees were individually numbered. Thus, the 0.1-acre measurement plot was surrounded by a similarly treated (including pruning) 0.3-acre buffer zone one-half chain wide. Site index (base age 25 years) was determined to be near 62 feet.

Thinning Treatments

Plots were initially thinned at age 12 to 40, 60, 80, and 100 square feet of basal area per acre. After the second inventory at age 15, basal areas were reduced to 30, 50, 70, and 90 square feet per acre. Plots were thinned again at ages 24, 27, 30, 35, and 40 to the same density levels (30, 50, 70, and 90 square feet). A natural reduction of growth rates observed after age 30 allowed for the use of a 5-year thinning period. Severely damaged plots 15 and 17 have not been thinned since the ice storm at age 16. Plot 4 recovered by age 35 and was thinned at age 40. After calculation of basal area for each measurement plot and its corresponding buffer area, trees were identified for removal to maintain the prescribed basal area. Trees were generally thinned from below. The following, somewhat overlapping criteria (in order of decreasing importance), was applied:

1. Inferior tree size (diameter and height)
2. Low increment
3. Poor stem form
4. Traces of insect infestation
5. Damaged stems (logging or lightning)
6. Damaged or lopsided crown
7. Uneven spatial distribution
8. Excessive cone production, an indicator of reduced increment

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Pruning Treatments

Only 12 numbered trees were pruned on each plot but 3 times as many trees were similarly treated on the surrounding buffer. These trees were pruned twice, at ages 12 and 15, to 25, 40, and 50 percent of total tree height.

Installation of Unthinned Control Plots

Originally, no unthinned plots were established. The need for such plots was later recognized, and at the age of 27 (in the summer of 1984) five control plots (without thinning or pruning) were established on the adjacent untreated part of the plantation. The size and arrangement of each plot was the same as that of the 40 original plots. To make growth comparable, hardwood competition was controlled on the plots by injecting Tordon® 101 R.

Ice Storm Modifications

A devastating ice storm hit the plots at a vulnerable age (16 years) and period (a year after thinning); see Bragg and others (2003). A salvage cut left three plots (4, 15, and 17) with basal areas below the intended densities. Plot 4 recovered in basal area at age 35; plots 15 and 17 recovered in basal area at age 43. Measurements included in the analyses for plot 4 were 12, 15, 16, 35, 36, 37, 40, 43, 45, 48, and 50; for plots 15 and 17 they were 12, 15, 16, 43, 45, 48, and 50. The two subsequent scheduled thinnings (at ages 18 and 21) were not conducted for any plot due to the reduction of density from the storm. Less severe ice storms occurred in 1979, 1994, and 2000.

Other Modifications

In 1986 the construction of a new road destroyed one of the control plots (after only one measurement—this observation was included in the analysis). This lost plot was replaced in 1986 and assigned the number 44. It was measured for the first time in 1987. Drought conditions during the spring and early summer of 1988 along with extremely high late summer temperatures placed many trees under severe stress. In August, three isolated areas of insect damage were located at the southeastern border of the study plots. An entomologist surveyed the area and found evidence suggesting the presence of southern pine beetles (*Dendroctonus frontalis* Zimmermann) and black turpentine beetles [*Dendroctonus terebrans* (Olivier)]. To control the infestation, 20 trees were cut just outside of the study area. Sixteen trees infested with turpentine beetles within the test plots were sprayed with Pestroy (9 ml/gallon of water). In 1997, six trees (including two damaged by lightning) had insect damage, possibly due to the southern pine beetle, and were salvaged. Prescribed burns were conducted in 1981, 1984, 1986, 1990, 1995, 2000, and the site was bush-hogged in 1972, 1986, 1987, 1997, 2002, and 2005 to reduce competition from hardwoods, shrubs, and herbaceous vegetation.

Remeasurement Procedures

Measurement methods and techniques have been maintained throughout the study to assure the compatibility of results from all inventories. The diameter of all trees located within the interior measurement plots was measured at a horizontal line originally placed at 4.5 feet aboveground level. Total tree height (to the top of the tree) and height to the base of the live crown were also measured for all surviving trees in the interior measurement plots.

A Zeiss teledendrometer was used to measure height to even-number upper stem diameters (2, 4, 6, etc., in inches) and the diameters themselves to calculate volume according to the Grosenbaugh height accumulation method (Grosenbaugh 1954). Lower even-number diameters were measured using a diameter tape or caliper. During the first 4 inventories, heights and upper stem diameters were measured for up to only 12 trees per plot. At subsequent inventories, these measurements were conducted for all living trees.

Plot-Level Summary Measures

Quadratic mean diameter [D (inches)], arithmetic mean height [H (feet)], basal area per acre [BA (square feet)], and cubic-foot volume per acre were calculated for each plot by measurement age. Summaries of the measurement plots for a variety of treatment combinations are presented.

RESULTS

During the course of the study, there was no single thinning or pruning treatment that appeared to vastly impact H (table 1). However, the lack of thinning reduced H growth, most likely due to excessive stand density. Thinning greatly impacted D (table 2) but pruning appeared to have little impact. Basal area per acre was greater as residual stand densities increased (table 3). Moderate rates of pruning did not appear to substantially impact basal area. As for volume per acre, the lack of thinning produced the greatest standing volume (table 4, fig. 1). Consistent with our expectations, more intensive thinning regimes reduced standing volume across a rotation. Moderate rates of pruning did not appear to largely impact volumes (table 4, fig. 2).

DISCUSSION

There is some discrepancy as to whether thinning increases height (Zhang and others 1997). Similar to the findings of others (Peterson and others 1997, Williston 1979), we found that H is not vastly affected by thinning intensity (table 1). However, extreme stand densities (unthinned) reduced H growth similar to the findings of Zhang and others (1997). During common regional economic and biological rotation ages (25 to 50 years), H in unthinned control plots ranged from 3 to 21 percent less than other treatments, percent reductions in H generally decreased with age. Height was not impacted by pruning treatments consistent with the findings of Sparks and others (1980).

More intensive thinning produced greater D similar to the findings of others (e.g., Baldwin and others 2000, Feduccia and Mann 1976, Sparks and others 1980, Williston 1979, Zhang and others 1997). Plots that received no pruning but thinned to a basal area of 30 square feet had the greatest D at all ages (table 2). Thinning resulted in much greater growing space for individual trees and the lack of pruning

Table 1—Arithmetic mean height across four pruning treatments and four residual basal areas (30, 50, 70, and 90 square feet of basal area per acre) and a control where no thinning or pruning treatments were conducted

	No pruning				Pruning 25 percent of total tree height			
Age	30	50	70	90	30	50	70	90
	feet				*feet*			
12	34.4	35.2	32.4	34.4	36.1	36.9	36.7	36.7
15	43.1	41.7	38.3	40.1	42.5	43.3	42.6	42.6
16	46.8	45.4	40.7	43.4	45.5	46.3	44.8	45.0
19	51.7	50.8	47.7	50.3	50.3	54.2	50.4	51.3
24	63.7	61.4	55.7	59.1	59.5	62.0	59.7	61.5
27	66.5	68.0	67.6	66.4	66.3	68.6	65.6	66.6
30	74.7	79.0	72.5	72.7	72.2	73.8	72.1	75.4
35	77.7	80.6	79.7	79.4	78.2	79.8	78.6	78.8
36	84.8	81.7	82.0	82.1	81.4	83.0	80.9	80.9
37	84.8	83.0	82.5	83.0	82.3	83.8	81.5	81.8
40	89.7	87.4	84.6	86.4	85.7	87.4	84.7	85.2
43	92.0	89.5	86.3	88.1	86.0	90.0	87.4	86.0
45	93.8	93.0	88.7	89.7	88.2	92.1	89.3	87.5
48	99.4	95.2	89.4	91.7	90.7	94.3	91.7	90.1
50	99.5	95.7	90.3	92.3	92.5	95.7	93.2	92.2
	Pruning 40 percent of total tree height				Pruning 50 percent of total tree height			
Age	30	50	70	90	30	50	70	90
	feet				*feet*			
12	36.3	35.3	35.6	34.5	35.5	36.9	36.7	35.0
15	42.8	41.6	41.5	40.0	42.0	43.4	42.8	40.4
16	45.8	45.4	44.7	42.4	46.1	46.7	46.0	43.8
19	50.3	51.1	51.5	48.8	51.4	51.5	51.3	50.1
24	61.1	64.2	60.6	58.5	60.8	62.9	61.5	58.3
27	66.6	69.6	67.5	65.4	66.8	69.1	68.6	66.1
30	72.3	74.1	74.2	71.6	72.5	74.6	74.3	72.0
35	75.8	80.5	79.6	79.0	77.8	80.1	79.9	78.6
36	76.6	81.3	81.6	81.4	78.6	81.7	81.6	79.8
37	77.7	82.1	82.3	82.4	80.1	82.8	82.4	80.8
40	82.2	85.2	85.3	85.9	84.4	86.9	85.5	84.1
43	82.2	88.3	86.7	87.9	85.2	89.0	88.0	86.5
45	83.7	90.6	87.9	90.1	87.9	91.9	89.3	88.1
48	85.8	93.0	89.8	93.0	90.8	92.7	91.8	90.9
50	87.2	94.8	91.1	94.2	91.5	93.3	92.9	92.5

Age	Control
	feet
27	55.4
30	62.4
35	70.1
36	71.2
37	72.3
40	75.9
43	78.7
45	80.3
48	82.6
50	84.7

Table 2—Quadratic mean diameter across four pruning treatments and four residual basal areas (30, 50, 70, and 90 square feet of basal area per acre) and a control where no thinning or pruning treatments were conducted

	No pruning				Pruning 25 percent of total tree height			
Age	30	50	70	90	30	50	70	90
	inches				*inches*			
12	7.1	7.0	6.0	6.2	6.7	7.0	6.4	6.7
15	9.6	8.6	7.1	7.0	8.5	8.5	7.4	7.6
16	10.8	9.5	7.7	7.9	8.9	9.0	7.8	8.1
19	13.4	11.1	9.1	9.2	10.5	11.1	9.2	9.8
24	15.7	12.3	10.3	10.1	12.5	13.0	10.8	11.7
27	17.7	14.8	13.3	11.7	14.4	14.7	12.2	12.9
30	19.4	16.4	14.8	12.7	16.7	16.3	13.9	14.6
35	21.8	18.6	16.8	14.8	20.6	18.9	15.8	15.3
36	22.9	18.9	17.2	15.4	21.0	19.8	16.1	15.9
37	23.2	19.2	17.5	15.7	21.3	20.1	16.3	16.2
40	24.2	20.3	18.3	16.2	22.2	20.9	17.1	17.0
43	25.7	21.5	19.3	17.3	23.5	22.4	18.1	17.4
45	26.4	22.3	19.6	17.7	24.0	23.0	18.4	17.8
48	27.3	23.0	20.4	18.4	25.0	24.0	19.1	18.5
50	28.1	23.8	21.1	19.1	25.9	24.8	19.7	19.0

	Pruning 40 percent of total tree height				Pruning 50 percent of total tree height			
Age	30	50	70	90	30	50	70	90
	inches				*inches*			
12	6.9	6.6	6.4	6.7	6.3	6.7	6.9	6.3
15	8.8	8.0	7.6	7.5	8.5	8.1	8.1	7.2
16	9.8	9.1	8.4	8.1	9.8	9.1	8.9	8.1
19	11.9	11.1	10.0	9.4	12.1	11.1	10.7	9.3
24	14.3	12.9	11.4	10.8	14.3	12.9	12.1	10.5
27	16.7	14.8	13.3	12.2	16.5	15.1	13.8	12.0
30	18.4	16.1	14.6	13.3	18.4	16.6	14.9	13.2
35	21.0	18.5	16.7	15.4	20.8	18.9	17.1	14.8
36	21.4	18.9	17.3	16.0	21.6	19.6	17.8	15.4
37	21.8	19.1	17.7	16.2	21.9	19.9	18.1	15.7
40	22.8	19.9	18.4	16.9	22.6	20.6	18.9	16.4
43	24.3	20.8	19.9	17.8	23.7	22.0	19.8	17.3
45	25.0	21.3	20.5	18.1	24.3	22.4	20.3	17.7
48	25.9	22.1	21.3	18.7	25.1	23.1	20.9	18.4
50	26.7	22.8	22.0	19.3	25.8	23.8	21.6	19.0

Age	Control
	inches
27	9.5
30	10.5
35	11.6
36	11.7
37	11.8
40	12.8
43	13.5
45	13.7
48	14.1
50	14.7

Table 3—Basal area per acre across four pruning treatments and four residual basal areas (30, 50, 70, and 90 square feet of basal area per acre) and a control where no thinning or pruning treatments were conducted

	No pruning				Pruning 25 percent of total tree height			
Age	30	50	70	90	30	50	70	90
	square feet				*square feet*			
12	55	76	86	101	56	76	90	105
15	71	85	105	124	61	83	101	123
16	38	59	77	99	32	53	73	92
19	59	73	96	110	30	42	60	68
24	81	91	122	132	43	58	76	90
27	34	59	87	104	37	59	85	110
30	41	59	84	106	40	63	84	105
35	52	56	92	108	31	65	88	100
36	29	58	81	90	32	56	70	95
37	29	60	83	94	33	58	72	98
40	32	67	91	100	36	63	72	102
43	36	76	81	98	30	55	71	98
45	38	81	84	103	32	58	74	102
48	41	87	91	110	34	63	80	110
50	43	93	97	119	37	67	84	117

	Pruning 40 percent of total tree height				Pruning 50 percent of total tree height			
Age	30	50	70	90	30	50	70	90
	square feet				*square feet*			
12	51	69	87	100	55	71	90	107
15	63	87	106	117	65	87	109	125
16	33	56	75	98	34	57	78	99
19	47	60	81	82	44	73	87	107
24	67	82	106	105	62	98	110	134
27	40	60	80	98	39	57	80	107
30	37	61	81	103	43	59	81	104
35	41	62	86	108	40	64	85	111
36	34	45	76	95	25	48	69	95
37	36	46	79	97	26	49	72	98
40	38	50	86	102	28	53	78	102
43	32	48	78	96	31	45	86	98
45	34	50	83	100	32	47	90	103
48	37	54	90	107	34	50	96	110
50	39	57	96	113	36	53	102	118

Age	Control
	square feet
27	123
30	145
35	158
36	157
37	163
40	134
43	142
45	147
48	154
50	160

Table 4—Cubic-foot volume per acre across four pruning treatments and four residual basal areas (30, 50, 70, and 90 square feet of basal area per acre) and a control where no thinning or pruning treatments were conducted

	No pruning				Pruning 25 percent of total tree height			
Age	30	50	70	90	30	50	70	90
	cubic feet				*cubic feet*			
12	1013	1346	1544	1841	952	1335	1517	1770
15	1544	1885	2296	2779	1427	1959	2189	2646
16	851	1354	1790	2326	781	1335	1705	2143
19	1508	1962	2484	2944	865	1262	1707	1997
24	2378	2875	3842	4084	1428	2026	2533	3055
27	1009	2028	2856	3429	1279	2100	2841	3802
30	1364	2165	3045	3865	1504	2394	3032	3848
35	1852	2308	3722	4369	1276	2580	3369	4048
36	1215	2503	3418	3843	1417	2508	3126	4181
37	1224	2598	3605	4064	1474	2598	3223	4361
40	1436	3022	4059	4598	1670	2937	3368	4745
43	1686	3528	3718	4622	1416	2668	3471	4625
45	1855	4067	4175	5579	1545	3043	3806	5133
48	2083	4398	4457	5652	1734	3269	4186	5781
50	2237	4765	4827	6077	1910	3578	4526	6220
	Pruning 40 percent of total tree height				Pruning 50 percent of total tree height			
Age	30	50	70	90	30	50	70	90
	cubic feet				*cubic feet*			
12	904	1200	1531	1750	973	1286	1574	1865
15	1447	1987	2388	2612	1498	2026	2503	2815
16	820	1336	1791	2303	821	1372	1885	2323
19	1286	1686	2279	2158	1206	2028	2394	2896
24	2115	2754	3509	3270	1924	3219	3604	4114
27	1366	2016	2731	3205	1277	1944	2740	3545
30	1371	2296	3141	3662	1487	2210	3108	3782
35	1551	2372	3520	4176	1453	2590	3432	4516
36	1422	1927	3303	4133	1036	2029	3041	4090
37	1486	1990	3461	4324	1092	2124	3174	4265
40	1711	2266	3940	4769	1236	2412	3549	4654
43	1443	2250	3725	4635	1388	2139	4072	4639
45	1517	2433	4308	5157	1491	2413	4710	5169
48	1753	2753	4528	5567	1702	2539	4959	5658
50	1909	3002	4951	6004	1832	2731	5294	6165

Age	Control
	cubic feet
27	3621
30	4519
35	5791
36	6246
37	6591
40	5733
43	6337
45	6991
48	7529
50	8061

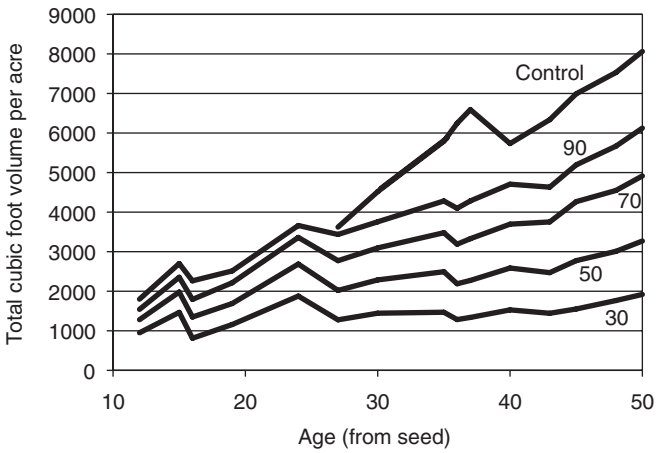

Figure 1—Standing cubic-foot volume per acre of four residual basal areas (30, 50, 70, and 90 square feet of basal area per acre) averaged across all pruning treatments. Control refers to the lack of both thinning and pruning.

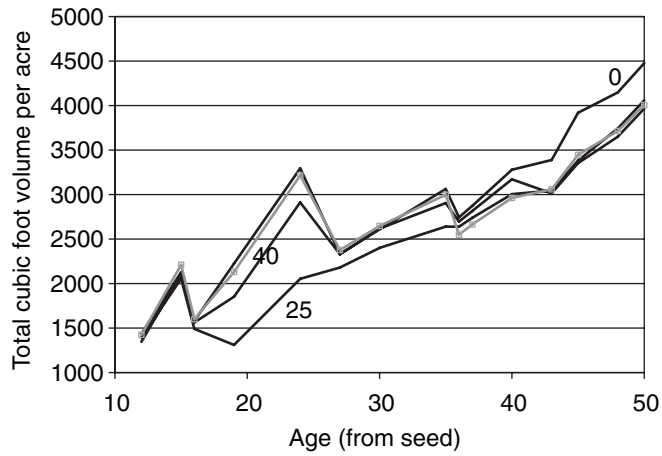

Figure 2—Standing cubic-foot volume per acre of four pruning treatments (no pruning–0, 25, 40, and 50 percent of total tree height) averaged across all thinning treatments (controls were not included in the no pruning–0 treatment for consistency among measurement ages). The gray line in the pruning figure is the 50-percent pruning treatment.

likely resulted in optimum crown conditions for diameter growth. At all ages, the control plots had a D much smaller than those of any of the thinned plots. However, differences in D were not consistently different among the pruning treatments.

Data from all ages showed that more intensive thinning regimes reduced BA (table 3). Similar results have been observed (Baldwin and others 2000). Standing cubic-foot volume was greatest in plots with less intensive thinning treatments (table 4, fig. 1) consistent with the findings of others (Feduccia and Mann 1976, Williston 1979). The heaviest pruning treatment (25 percent) substantially reduced volume up to age 35 (table 4, fig. 2). For the 40 percent pruning treatment, yields were not affected after age 27 when compared to the no pruning treatment.

CONCLUSIONS

Based on our results, it does not appear that moderate pruning substantially impacts yields. Hence, to improve wood quality, managers may want to consider pruning.

If the purpose of tree management is to have a higher standing yield, then the best stand density management regime would be to avoid thinnings. However, since revenues per tree are generally greater as diameter increases, thinning is an economically viable alternative. An optimal trade-off between standing yield and average diameter appears to be obtained when residual BAs are 50 or 70 square feet. Based on our results, differences in yields across a rotation for various residual stand density levels appears to be more related to differences in diameter than height growth.

ACKNOWLEDGMENTS

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LITERATURE CITED

Baldwin, V.C., Jr.; Peterson, K.D.; Clark, A. [and others]. 2000. The effects of spacing and thinning on stand and tree characteristics of 38-year-old loblolly pine. Forest Ecology and Management. 137: 91–102.

Bragg, D.C.; Shelton, M.G.; Zeide, B. 2003. Impacts and management implications of ice storms on forests in the Southern United States. Forest Ecology and Management. 186: 99–123.

Feduccia, D.P.; Mann, W.F., Jr. 1976. Growth following initial thinning of loblolly pine planted on a cutover site at five spacings. Res. Pap. SO-120. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 8 p.

Grosenbaugh, L.R. 1954. New tree-measurement concepts: height accumulation, giant tree, taper and shape. Occas. Pap. 134. New Orleans: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 25 p.

Peterson, J.A.; Seiler, J.R.; Nowak, J. [and others]. 1997. Growth and physiological responses of young loblolly pine stands to thinning. Forest Science. 43: 529–534.

Sharma, S.; Burkhart, H.E.; Amateis, R.L. 2002. Modeling the effect of density on the growth of loblolly pine trees. Southern Journal of Applied Forestry. 26: 124–133.

Sparks, R.C.; Linnartz, N.E.; Harris, H.E. 1980. Long-term effects of early pruning and thinning treatments on growth of natural longleaf pine. Southern Journal of Applied Forestry. 4: 77–79.

Williston, H.L. 1979. Growth and yield to age 37 in north Mississippi loblolly plantations. Southern Journal of Applied Forestry. 3: 127–130.

Zhang, S.; Burkhart, H.E.; Amateis, R.L. 1997. The influence of thinning on tree height and diameter relationships in loblolly pine plantations. Southern Journal of Applied Forestry. 21: 199–205.

INFLUENCE OF FOREST ROAD BUFFER ZONES ON SEDIMENT TRANSPORT IN THE SOUTHERN APPALACHIAN REGION

Johnny M. Grace III and Stanley J. Zarnoch[1]

Abstract—A gap exists in the understanding of the effectiveness of forest road best management practices (BMP) in controlling sediment movement and minimizing risks of sediment delivery to forest streams. The objective of this paper is to report the findings of investigations to assess sediment travel distances downslope of forest roads in the Appalachian region, relate sediment travel distances to BMP recommendations, and describe the deposition patterns within buffer zones. A total of 164 randomly selected sediment deposition zones were measured downslope of the road lead-off ditch structures for each forest. The mean distance sediment was deposited within buffer zones was 20 m for the Chattahoochee National Forest (Georgia) and 41 m for the Talladega National Forest (Alabama). Sediment transport distances were <30 m for 38 and 88 percent of sites evaluated on the National Forests in Alabama and Georgia, respectively. A small percentage (15 percent) of sediment from road sections terminated in streams downslope of the road sections in this investigation. Results indicate that current road BMPs are somewhat effective in reducing risks of road to steam connectivity in most cases. However, the deposition lengths within the buffers for both forests were >20 m which may exceed the buffer zone width requirements in the States in this investigation and many States in the United States. This research indicates that the connectivity issue requires additional research that specifically focuses on quantifying the fraction of forest road erosion reaching stream systems which include intermittent and perennial streams.

INTRODUCTION

Nonpoint source (NPS) pollution issues related to forest activities (silviculture) are not as extensive as the leading source activities (agriculture, resource extraction, urban stormwater, and construction) (West 2002). However, NPS issues, particularly related to soil erosion and suspended sediment, are a major concern in forest resource management (Binkley and Brown 1993, Marion and Ursic 1993, Patric 1976) because water is a key product of the southern forests' resource (Sun and others 2004). Water-quality problems related to sediment from forested landscapes are difficult to address due to the difficulty in locating the sediment source. Upslope erosion and stormwater routing can increase sediment transport as storm runoff travels downslope through natural and artificial drainages toward critical streams (Grace 2007; Swift 1985, 1988). These artificial drainages, i.e., culverts, roads, road sideslopes, and roadside drainage ditches, are often conduits for storm runoff and sources for accelerated erosion losses. The forest road prism is identified as a major contributor to NPS issues related to forest activities and have the potential to elevate NPS problems (Grace and Clinton 2007). Forest roads provide access to perform management prescriptions which make them critical elements in most forest management activities. Forest roads are beneficial in many aspects; however, roads can also result in environmental impacts on the nation's watersheds (Grace 2002b, Lane and Sheridan 2002). Water-quality issues related to sediments have been and continue to be a concern in forest road management strategies (Brown and others 1993, Grace 2005b, Neary and others 1989, Riedel and others 2004).

The forest floor is an effective filter of stormwater runoff from forest road systems based on previous short-term erosion and water-quality studies (Haupt 1959, Swift 1986). It is recognized that the forest floor can reduce sediment delivered to stream systems due to increased infiltration and trapping sediments. However, the trapping characteristics of the forest floor are temporal and diminish with each significant subsequent storm as sediment encroaches on forest water systems (Grace 2002a). Consequently, road systems can eventually have direct connectivity to streams resulting in degraded water quality. It is for this reason that forest roads continue to be reported as one of the major sources of sediment that reaches stream channels on forest lands (Van Lear and others 1998). Quantifying the extent and magnitude of sediment transport from forest roads has eluded scientists due to the complexity in assessing sediment transport across the forest floor and difficulty in defining the hydrologic connectivity between roads and streams.

Previous research has investigated factors influencing the movement and quantity of sediment traveling across the forest floor (Grace 2005a, Haupt 1959, Packer 1967, Swift 1986, Trimble and Sartz 1957). Researchers have presented several characteristics related to forest roads that can influence the distance sediment travels downslope and suggested minimal widths of buffer areas adjacent to forest streams. Based on this research, primary characteristics influencing sediment deposition distances are related to road section area, slope, and obstacles in the storm runoff flow path. Unfortunately, a gap remains in the understanding. The fundamental question that has not yet been answered is, "Do current management practices disconnect roads from stream systems and other water bodies?" This work reports an investigation to assess the movement of sediment from road systems in the Southern Appalachians and presents special issues related to the connectivity and the role of the

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forest buffer in minimizing the connection of forest roads and streams in this sensitive region.

METHODS

Study Area Description

The investigation was conducted on the Talladega and Chattahoochee National Forest lands in the Southern Appalachians located in Alabama and Georgia, respectively. This area is contained within the Appalachian Blue Ridge Forests ecoregion and consists of temperate broadleaf and mixed forests. Average annual temperatures in the region range from 4 °C to 15 °C.

The Talladega National Forest (TNF) study area is located at approximately lat. 33° and long. 85° where the long-term average annual precipitation is 1400 mm. Slopes in the TNF study area ranged from zero to 60 percent. Elevation of the study area is approximately 400 m above mean sea level (MSL).

The Chattahoochee National Forest (CNF) study area is located at approximately lat. 35° and long. 83° in the Southern Appalachian Mountains. Long-term average annual precipitation is 1800 mm. Slopes on the study site ranged from 10 to 60 percent. The elevation of the CNF study area is approximately 900 m above MSL.

Measurements

The initial phase of this investigation defined a representative sample of roads with similar construction standards, maintenance levels, traffic intensity, and drainage characteristics based on site reconnaissance and in consultation with national forest personnel. The potential study roads were constrained to crowned roads with native surfacing drained by lead-off ditches. Road maintenance primarily consisted of biannual grading with periodic ditch maintenance. Roads in the investigation ranged from 5 to >20 years. However, the age of roads was a difficult parameter to characterize due to records being essentially unavailable for older roads (>20 years). Traffic intensity for roads in the investigation ranged from low to moderate with intermittent periods of high traffic during periods of management activities.

Lead-off ditch (or road section drainage) structures were randomly selected from seven roads at the CNF and six roads at the TNF. The study design was a completely randomized design within each forest where roads were selected at random and each road was subsampled by site. The number of sites (or observations) selected was based on the number of sites required for statistical validity determined by a Neyman approximation using procedures presented by Grace (2005b). A total of 164 sites, 88 for the CNF and 76 for the TNF, were measured in the Appalachian region with replications of factors hypothesized to influence sediment movement downslope. These factors included road section length, road width, road gradient, downslope gradient, forest floor index, soil texture, and deposition length. However, this report concentrates on deposition length measurements within the forest buffer zones. The statistical analysis consisted of testing for differences between the

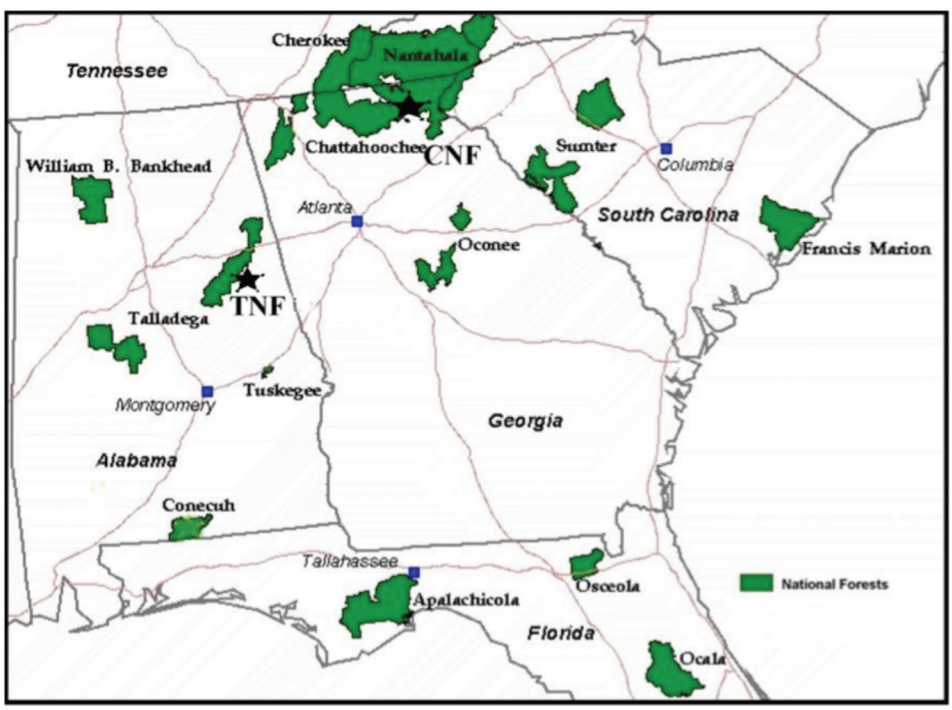

Figure 1—Location of study sites within the Southern Region highlighting the Talladega National Forest (TNF) site and the Chattahoochee National Forest (CNF) site.

national forests as a fixed factor with both roads nested within national forests and sites nested within roads as random factors. PROC MIXED (SAS Institute Inc. 2004) was used for the analysis of this mixed model. SAS TTEST procedures were used to test for differences in observed drainage structure spacing and best management practices (BMP) recommended drainage structure spacing.

The length that sediment deposition areas extended into the forest buffer was determined by first tracking the most remote deposition in the road stormwater flow path. Total deposition length was taken as the distance between the roadway edge and the most remote point of visible sediment deposition. The buffer distance from the road edge was also measured along the storm runoff flow path to determine the established buffer length for direct comparison with the observed deposition length.

RESULTS AND DISCUSSION

Results from the 13 experimental roads on the TNF and CNF are presented in table 1. These data were evaluated based on two different components of concern in forest management. First, the results were analyzed to evaluate the effectiveness of implementation of forest road BMPs for the sites, e.g., determining the implementation of current BMPs for forest roads. Initially analyzing the data for implementation effectiveness minimized risks associated with interpreting differences in application and implementation of BMPs as differences in deposition patterns within buffers. Using this approach allowed a direct comparison of spacing recommendations for a specific road section and observed spacing contributing to deposition within the buffers. Secondly, data were analyzed to evaluate the influence of buffers in trapping sediment eroded from contributing road sections on the forests. This procedure allowed comparisons of observed deposition within buffers with both recommended buffer strip widths and implemented buffer strip widths for the forests.

Previous research provides the guides for BMP recommendations for road drainage structure spacing that are a function of road gradient (Haupt 1959, Megahan and Ketcheson 1996, Swift 1986, Trimble and Sartz 1957). The observed mean drainage structure spacing for road sections was 55.5 and 43.1 m for the TNF and CNF road sections, respectively. The mean of recommended spacing for drainage structures for road sections on the TNF and CNF are 58.3 and 141.2 m, respectively. The drainage spacing and resultant drainage area for the TNF road sections were significantly greater than those for the CNF which is primarily attributed to the reduced road gradients (6.6 percent slope opposed to 4.4 percent slope) on the CNF road sections in the investigation ($P = 0.042$) based on SAS MIXED procedures (SAS 2004). Analysis, using SAS TTEST procedures, also detected a significantly closer observed spacing than BMP recommendations for the CNF site ($P < 0.0001$). No significant differences were detected between observed spacing and recommendation spacing for the TNF ($P < 0.584$). A plot of the observed drainage structure spacing for each forest vs. BMP recommended lead-off or drainage structure spacing illustrates the consistent or less conservative spacing than

Table 1—Means and statistics for road length, width, gradient, buffer gradient, and deposition length within buffer zones for the Talladega National Forest and Chattahoochee National Forest sites

Parameter	*n*	Mean	Standard deviation	COV
Talladega National Forest				
Deposition length, m	75	41.2 a	22.1	53.3
Buffer gradient	76	23.4 b	11.3	48.4
Road gradient	76	6.6 a	4.2	62.8
Road length, m	76	55.5 a	36.6	66.1
Road width, m	76	3.0 a	0.8	27.9
Chattahoochee National Forest				
Deposition length, m	88	19.6 b	13.9	71.2
Buffer gradient	88	34.5 a	22.8	66.1
Road gradient	88	4.4 b	3.1	70.2
Road length, m	88	43.1 a	20.6	47.8
Road width, m	88	3.1 a	0.5	17.1

Mean values for a given parameter with the same letter are not significantly different between the forests at the 0.05 significance level.

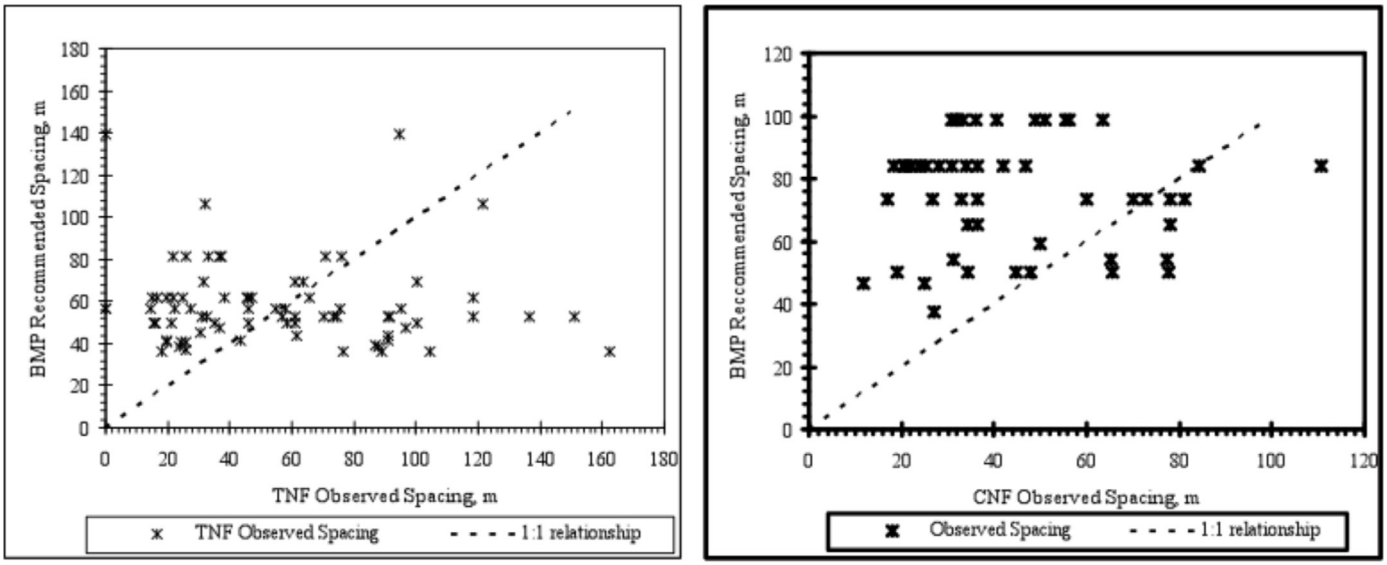

Figure 2—Observed drainage spacing for the TNF site (left) and the CNF site (right) versus BMP recommended spacing for each site.

recommended for the road sections in the investigations (fig. 2). The 1 to 1 line plotted along with data for each site represents consistency between observed and recommended spacing for road drainage structures; data points in perfect agreement would fall along this line. Observed spacing for the CNF was found to be more conservative than recommended drainage structure spacing for the majority (90 percent) of observations as indicated by points that fall to the left of the 1 to 1 line. Whereas, results show that only 59 percent of observed drainage spacing was more conservative than recommendations for the TNF. Based on these results, we would expect that deposition lengths within the buffers below road sections on both forests would be minimized due to consistent or closer (conservative) drainage structure spacing. Conservative spacing could be deemed as a built-in safety factor to maximize the distance that eroded sediment has to travel across the buffer to reach critical stream systems. Conservative spacing typically results in less drainage area contributing to each storm runoff pathway which translates to reduced energy to detach and transport sediment in the flow path. The combination of reduced energy to detach and transport sediment and reduced stormwater runoff from road sections results in decreased sediment deposition volumes and distances within buffer zones adjacent to streams. However, this was not always the case based on the analysis and results presented later in this paper.

The comparison of the observed and recommended spacing of road drainage structures addressed the adequacy of observed drainage spacing in satisfying the existing forest road BMP recommendations for the two forests in the investigation. Taking this into account, the sediment deposition patterns (deposition volume and length) are minimized below the forest road sections in the investigation based on the establishment criteria for the BMPs presented by previous research (Alabama Forestry Commission 1993, 2007; Georgia Forestry Commission 1999; Grace and Clinton 2007). The observed deposition lengths within buffers are presented for the TNF and CNF along with parameters hypothesized to influence the distance sediment travels into buffers toward stream systems (table 1). The mean distance that sediment was deposited in buffers downslope of road sections was 41.2 and 19.6 m for the TNF and CNF, respectively. Sediment deposition patterns within the buffers for the two forests were detected as significantly different at the 0.05 level of significance. In fact, sediment from road sections on the TNF encroached twice as far into the forest buffer than sediment from road sections on the CNF. This result is consistent with the results obtained from the spacing analysis which presented closer spacing of road drainage structures for the CNF road sections. The closer spacing resulted in reduced stormwater energy and runoff volume from the road sections on the CNF. Buffer gradient and road gradient were the only parameters, of those hypothesized to influence sediment deposition lengths in the buffers, detected as greater on the CNF in comparison to the TNF (table 1).

Surprisingly, road gradient was greater ($P = 0.042$) for the TNF which has less relief than the CNF. This likely had the greatest influence on the distance sediment traveled into the buffers for the sites. The differences in road parameters for the forests are a function of both design and management strategies utilized at the forests. Gradients for road sections on the CNF were minimized during the initial road construction which is likely due to the topography having greater relief. At the same time, spacing or road length was minimized on the CNF either at initial construction or at some point thereafter with the installation of additional drainage structures at closer spacing. It is recognized that the reduced road gradients at the CNF resulted in sediment deposition lengths within the buffers approximately half that of the TNF which had greater road gradients ($P = 0.042$) and reduced downslope gradients ($P = 0.025$). However, the deposition lengths within the buffers for both sites were ≥20 m which

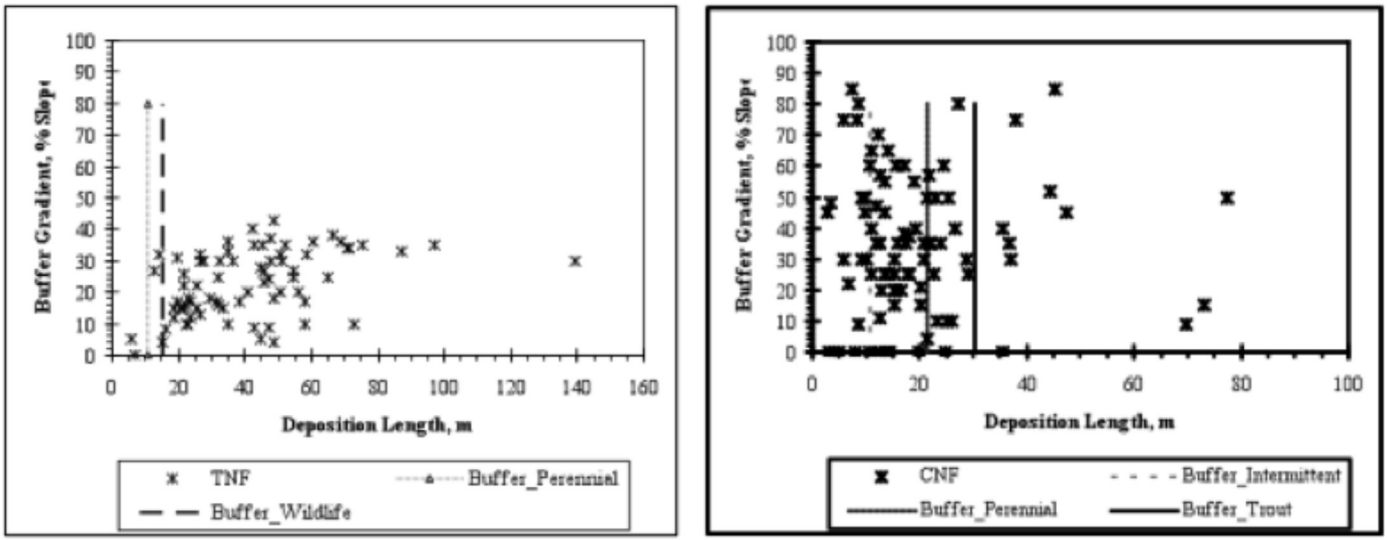

Figure 3—Deposition lengths within buffer zones vs. buffer gradients for the TNF site (left) and the CNF site (right). The BMP recommended buffer widths are presented for perennial streams (Buffer_Perennial), with a wildlife focus (Buffer_Wildlife) for the TNF site, and trout streams (Buffer_Trout) for the CNF site.

may exceed the buffer zone width requirements in the States in this investigation and many States in the United States.

Deposition data was compared to current forest road BMPs to gain a better understanding of the influence of forest buffer zones in this experiment on disconnecting roads from vital stream systems. A plot of deposition lengths in buffer zones vs. buffer gradients illustrates the relationship of deposition lengths to current minimum buffer zone widths recommendations for the two States in this investigation, Alabama and Georgia (fig. 3). Alabama's recommended streamside management zone (SMZ) widths are 11 m for perennial and intermittent streams and 15 m for management with a wildlife objective (Alabama Forestry Commission 2007) as represented by the vertical lines on figure 3. Georgia's recommended buffer zones range from 6 to 30 m based on slope approaching perennial, intermittent, or trout streams (Georgia Forestry Commission 1999). These BMP recommendations are also represented as vertical lines on figure 3. The majority of the deposition lengths within the buffers for the TNF are greater than the both SMZ width recommendations based on Alabama's BMPs for forestry. Specifically, 92 percent of deposition lengths into buffers observed at the TNF were greater than the most stringent Alabama SMZ recommendation of 15 m, and 62 percent of deposition lengths were >30 m (fig. 4). Conversely, results show that 88 percent of deposition lengths for the CNF were less than the trout stream minimum SMZ recommendation of 30 m. Sediment from a small percentage of road sections, 7 percent of TNF depositions, and 8 percent of CNF depositions emptied directly into streams. A t-test comparing the observed deposition lengths and BMP recommendations detected significantly ($P < 0.0001$) greater deposition lengths in comparison to both the minimum (11 m) and maximum (15 m) buffer zone recommendations for the TNF. Results were mixed for the CNF, deposition lengths were significantly greater than ($P < 0.0001$) minimum perennial stream (11 m) buffer zone recommendations and significantly less than ($P < 0.0001$) the maximum trout stream (30 m) recommendations.

Implications

Understanding the connectivity between roads and streams has been an area of focus of soil and water conservation engineering over the past 40 years. It is recognized that forest roads have increased risk associated with soil erosion and sediment delivery to stream systems. It follows that previous research has established forest roads as the major source of soil erosion from forest watersheds due to the many

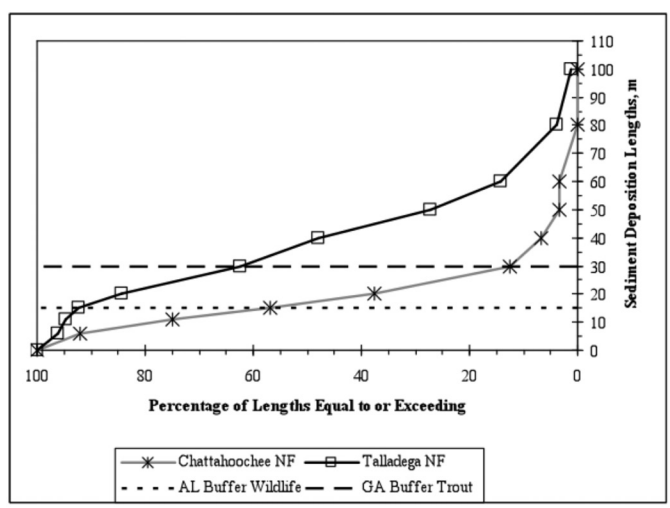

Figure 4—Buffer zone deposition length frequency duration curves observed for the TNF and CNF sites. BMP recommended buffer widths for Alabama with a wildlife emphasis and for Georgia trout streams are provided as reference lines.

factors that increase the potential for erosion losses from forest roads (Grace 2005b, Luce and Black 2001). However, sediment delivery from forest roads has not been as well studied and existing literature fails to directly link forest road erosion upslope to sediment delivery rates to downslope stream systems. The connectivity issue continues to require additional research that specifically focuses on quantifying the fraction of forest road erosion reaching stream systems which include intermittent and perennial streams. This research needs to additionally aim to determine the capacity of the forest floor filter or buffer zones, understand factors influencing this connectivity, and gain a better understanding of the benefit of sediment control in the connectivity issue.

In this investigation, the effectiveness of BMP recommended buffer zone widths in filtering and containing the sediment from forest roads within the forest buffers varies widely. Nearly 90 percent of the deposition lengths into the forest buffer at the CNF site were less than the most stringent buffer width recommendation of 30 m. Conversely, the most stringent buffer width recommendation of 15 m at the TNF site contained <10 percent of the deposition from upslope road drainage structures. Fortunately, the buffer zone widths at both sites exceeded the recommended buffer zone width recommendations and therefore had a built-in safety factor. These conservative buffer zone widths on both sites were quite effective in containing the majority of the sediment transport from adjacent roads in this investigation. This is supported by the fact that the investigation revealed that only a small percentage of the road sections emptied directly into streams at the sites.

Based on the findings of this investigation, buffer zone width recommendations require additional research to establish science-based support for existing recommendations. The fundamental question that must be addressed is "What percentage of buffer failure is acceptable environmentally, socially, and economically?" Answering this question would allow research to define buffer widths that satisfy the goals set forth by policy. Secondary questions that require consideration are "Is the acceptable level of buffer failure 75 percent, 50 percent, or 10 percent?", "What is the design period for buffer recommendations?", and "Are there alternative sediment control practices that can be utilized to minimize the buffer widths and risks associated with buffer failure?"

CONCLUSIONS

A total of 164 road sections and drainage structures from 13 roads from 2 forests in the Appalachian region were investigated to determine the influence of buffer zones sediment transport from forest roads in the region. Drainage structure spacing was similar to the recommended spacing for the TNF and closer than recommended for roads on the CNF. The TNF had greater road gradients than the CNF despite the fact that the CNF was at higher elevation and had greater relief. Decreased road gradients and closer spacing of drainage structures on the CNF resulted in a decreased drainage area and stormwater runoff volume which likely had the greatest influence on the shorter distance sediment traveled across the CNF buffer zones. In fact, CNF mean deposition length was 20 m which is less than half the mean of 41 m observed for the TNF deposition lengths.

The analysis revealed that the buffer width recommendations for the sites were somewhat effective in containing the sediment movement within the buffers. These results indicate that current BMPs are effective in most cases but may not be sufficient in all instances. Consequently, more than 90 percent of the observed deposition lengths exceeded the most stringent Alabama width recommendations on the TNF. The mean deposition length into buffers for TNF was greater than both the minimum (11 m) and maximum (15 m) BMP recommended buffer widths. Fortunately, forest managers or road engineers ensured that risks were minimized by having a built-in safety factor on buffer recommendations in the form of increased buffer widths below road sections. In contrast to the TNF results, only 8 percent of the deposition lengths exceeded the most stringent Georgia width recommendations on the CNF. The mean deposition length into buffers for CNF was less than both minimum and maximum BMP recommended buffer width. These results also revealed that a total of 12, or 15 percent, of the 164 road sections in this investigation had direct connectivity to stream systems.

This work has highlighted the need for additional research related to forest road buffer zone width. This research should focus on providing scientific support for current buffer zone width recommendations or the definition of science based buffer zone widths that consider risks associated with buffer failure. Future research and discussion needs to categorize the potential for buffer zone breaches associated with established recommendations; define the environmental, social, and economic risks associated with sediment extending beyond buffer zones; and define the acceptable level of sediment export from the buffers.

LITERATURE CITED

Alabama Forestry Commission. 1993. Alabama's best management practices for forestry. Montgomery, AL: Alabama Forestry Commission. 30 p.

Alabama Forestry Commission. 2007. Alabama's best management practices for forestry. Montgomery, AL: Alabama Forestry Commission. 31 p.

Binkley, D.; Brown, T.C. 1993. Forest practices as nonpoint sources of pollution in North America. Water Resources Bulletin. 29(5): 729–740.

Brown, T.C.; Brown, D.; Binkley, D. 1993. Laws and programs for controlling nonpoint source pollution in forested areas. Water Resources Bulletin. 29(1): 1–13.

Georgia Forestry Commission. 1999. Georgia's best management practices for forestry. Macon, GA: Georgia Forestry Commission. 69 p.

Grace, J.M., III. 2002a. Control of sediment export from the forest road prism. Transactions of the ASAE. 45(4): 1127–1132.

Grace, J.M., III. 2002b. Sediment movement from forest road systems. Resource. 13–14.

Grace, J.M., III. 2005a. Factors influencing sediment plume development from forest roads. In: Environmental Connection 05, Proceedings of conference 36. Steamboat Springs, CO: International Erosion Control Association: 221–230.

Grace, J.M., III. 2005b. Forest operations and water quality in the south. Transactions of the ASAE. 48(2): 871–880.

Grace, J.M., III. 2007. Modeling erosion from forest roads with WEPP. In: Environmental Connection 07, Proceedings of conference 36. Steamboat Springs, CO: International Erosion Control Association. 12 p.

Grace, J.M., III; Clinton, B.D. 2007. Protecting soil and water in forest road management. Trans. ASABE. 50(5): 1579–1584.

Haupt, H.F. 1959. Road and slope characteristics affecting sediment movement from logging roads. Journal of Forestry. 57(5): 329–332.

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

Luce, C.H.; Black, T.A. 2001. Spatial and temporal patterns in erosion from forest roads. In: Wigmosta, M.S.; Burges, S.J., eds. Influence of urban and forest land uses on the hydrologic-geomorphic responses of watersheds. Water Resour. Monogr. Washington, DC: American Geophysical Union: 165–178.

Marion, D.A.; Ursic, S.J. 1993. Sediment production in forest of the Coastal Plain, Piedmont, and Interior Highlands. In: Technical workshop on sediments; Proceedings of the EPA/Forest Service workshop. Washington, DC: Terrene Institute: 19–27.

Megahan, W.F.; Ketcheson, G.L. 1996. Predicting downslope travel of granitic sediments from forest roads in Idaho. Journal of the American Water Resource Association. 32(2): 371–382.

Neary, D.G.; Swank, W.T.; Riekerk, H. 1989. An overview of nonpoint source pollution in the Southern United States. In: Proceedings of the symposium, The forested wetlands of the Southern United States. Gen. Tech. Rep. SE-50. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 168 p.

Packer, P.E. 1967. Criteria for designing and locating logging roads to control sediment. Forest Science. 13(1): 2–18.

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

Riedel, M.S.; Vose, J.M.; Bolstad, P.V. 2004. Characterizing hysteretic water quality in Southern Appalachian streams. In: Proceedings, 2004 national water quality monitoring conference. Washington, DC: U.S. Advisory Committee on Water Information—National Water Quality Monitoring Council. 13 p.

SAS Institute. 2004. SAS online. Doc 9.1.3. Cary, NC: SAS Institute Inc.

Sun, G.; Riedel, M.; Jackson, R. [and others]. 2004. Influences of management of southern forests on water quantity and quality. In: Gen. Tech. Rep. SRS-75. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 195–234. Chapter 19.

Swift, L.W., Jr. 1985. Forest road design to minimize erosion in the Southern Appalachians. In: Blackman, B.G., ed. Forest and water quality: a mid-south symposium. Monticello, AR: University of Arkansas: 141–151.

Swift, L.W., Jr. 1986. Filter strip widths for forest roads in the Southern Appalachians. Southern Journal of Applied Forestry. 10(1): 27–34.

Swift, L.W., Jr. 1988. Forest access roads: design, maintenance, and soil loss. In: Swank, W.T.; Crossley, D.A., Jr., eds. Ecological studies. Forest hydrology and ecology at Coweeta. New York: Springer-Verlag: 313–324. Vol. 66.

Trimble, G.R., Jr.; Sartz, R.S. 1957. How far from a stream should a logging road be located? Journal of Forestry. 55(5): 339–342.

Van Lear, D.H.; Taylor, G.B.; Hansen, W.F. 1998. Sediment sources to the Chattooga River. In: Waldrop, T.A., ed. Proceedings of the ninth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 357–362.

West, B. 2002. Water quality in the South. In: Wear, D.N.; Greis, J.G., eds. Southern Forest Resource Assessment—technical report. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 455–477.

DEVELOPING EQUATIONS FOR ESTIMATING TREE COMPONENT BIOMASS FOR NATURALLY REGENERATED SHORTEAF PINE IN SOUTHEAST OKLAHOMA WITH APPLICATION TO BIOMASS PARTITIONING IN THINNED AND UNTHINNED STANDS

Nabin Gyawali, Thomas B. Lynch, and Rodney E. Will[1]

Abstract—Traditionally, the main focus of forest production has usually been to maximize allocation of biomass to merchantable stem wood. But the assessment of biomass partitioning in stands is needed to address management concerns such as stem production and allocation, carbon sequestration, wildland fire, whole tree harvesting, etc. Thinning mainly increases the bole diameter and crown area of the residual trees. This change results in changes in biomass partitioning to tree components. To meet all purposes of forest management it is imperative to understand the effects of thinning on the allocation of biomass to different aboveground tree components. Data were obtained from a thinning study established in even-aged naturally-regenerated shortleaf pine (*Pinus echinata*) stands consisting of 12 permanent plots established during 1988 to 1989 (site 1) and 9 permanent plots established in 1990 (site 2). The two sites are located in the Ouachita Mountains of Pushmataha County in southeastern Oklahoma on industrial forest lands. The sites consisted of even-aged stands 25 to 35 years old at the time of thinning. Site 1 has a site index of 17.4 m (base age 50) and site 2 has a site index of 22.2 m (Wittwer and others 1998). Tree component (branch, foliage, and tree bole) biomass equations were developed based on destructive measurement of 48 shortleaf pine trees, ranging from 5 to 33 cm in d.b.h. on site 1, and 36 shortleaf pine trees ranging from 7 to 40 cm in d.b.h. on site 2. Thinning treatments (low thinning) included unthinned control plots and plots thinned to 70-percent, 50-percent and 30-percent full stocking on site 1, and plots thinned to 70 percent, 50 percent, and unthinned control on site 2. The biomass equations were used to estimate biomass quantities (/ha) on plots to evaluate biomass partitioning. Analysis of variance (ANOVA) and multiple comparisons were done using the REML approach in SAS PROC MIXED (SAS Institute Inc., Cary NC). Since site-by-treatment interactions were significant, the interaction model was used for ANOVA to test simple effects. Multiple comparisons were conducted by using SLICE option under the LSMEANS statement in the SAS MIXED procedure with the null hypothesis of equality of the means being rejected if P-value ≤ 0.05 experiment wise type I error rate. The proportion of biomass partitioned to branches and bark was significantly affected by thinning. Unless heavily thinned, thinning treatments did not significantly affect the proportion of biomass allocation to bole. Biomass partitioning to branches was higher as thinning intensity increased. Thinned stands partitioned smaller proportions of total biomass to bark. Although thinning seemed to reduce the total foliage biomass, it was evident that heavily thinned stands increased the proportion of foliage to total biomass.

LITERATURE CITED

Wittwer R.F.; Lynch, T.B.; Huebschmann, M.M. 1998. Stand density index for shortleaf pine (*Pinus echinata* Mill.) natural stands. In: Waldrop, T.A., ed. Proceedings of the ninth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 590–596.

SAS Institute Inc. 2000–2004. Help and documentation. Version 9.1.3. Cary, NC: SAS Institute Inc.

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EFFECTS OF PLANTING DENSITY AND GENOTYPE ON CANOPY SIZE, CANOPY STRUCTURE, AND GROWTH OF 25-YEAR-OLD LOBLOLLY PINE STANDS IN SOUTHEASTERN OKLAHOMA

Thomas C. Hennessey, Rodney E. Will, Thomas B. Lynch, Robert Heinemann, Randal Holeman, Dennis Wilson, Keith Anderson, and Gregory Campbell[1]

Abstract—Leaf biomass and its display within the canopy are important driving variables of stand growth because they reflect a tree or stand's capacity to intercept radiation, reduce carbon dioxide, and transpire water. We determined the effects of planting density (4- by 4-, 6- by 6-, 8- by 8-, and 10- by 10-foot spacing) on annual needle fall biomass, intercepted radiation, and canopy openness and then related these measures of canopy size to current annual increment and basal area for 25-year-old stands planted with two loblolly pine (*Pinus taeda* L.) seed sources: North Carolina Coastal (NCC 8-01) and Oklahoma/Arkansas (O/A mix 4213). The study site is located on an excessively drained, mountain site in southeastern Oklahoma. Litter was measured using randomly located traps and summed for a phenological year (Apr. 1 to Mar. 31). Intercepted radiation and canopy openness were measured using hemispherical photographs and the WinScanopy canopy analysis software. After 25 years, initial planting density no longer significantly affected litterfall, intercepted radiation, or canopy openness. Canopy openness was greater for the North Carolina Coastal than the Oklahoma/Arkansas genotypes. Somewhat surprisingly, the measures of canopy size were not related to current annual increments. In contrast, basal area (or standing volume) was correlated to canopy size (R^2 = 0.12 for litterfall, R^2 = 0.45 for intercepted radiation, and R^2 = 0.13 for canopy openness). Possible reasons for lack of a relationship between canopy size and current annual increment could be due to varying and sometimes intense intraspecific competition, environmental variation across the site, varying and sometimes large amounts of woody biomass relative to foliage, and previous stochastic events. Correlation between basal area and canopy size indicates canopy variables have some ability to predict basal area.

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Effects of planting density and genotype on canopy size, canopy structure, and growth of 25-year-old loblolly pine stands in southeastern Oklahoma

Thomas C. Hennessey, Rodney E. Will, Thomas B. Lynch, Robert Heinemann, Keith Anderson, Dennis Wilson, Randal Holeman, Gregory Campbell & Ed Lorenzi

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ABSTRACT

Leaf biomass and its display within the canopy are important driving variables of stand growth because they reflect a tree or stand's capacity to intercept radiation, reduce carbon dioxide, and transpire water. We determined the effects of planting density (4 x 4, 6 x 6, 8 x 8, and 10 x 10 foot spacing) on annual needle-fall biomass, intercepted radiation, and canopy openness and then related these measures of canopy size to current annual increment and basal area for 25-year-old stands planted with two loblolly pine seed sources: North Carolina Coastal (NCC 8-01) and Oklahoma/Arkansas (O/A mix 4213). The study site is located on an excessively drained, mountain site in southeastern Oklahoma. Needle litter was measured using randomly located traps and summed for a phenological year (1 April to 31 March). Intercepted radiation and canopy openness were measured using hemispherical photographs and the WinScanopy Canopy Analysis Software. Somewhat surprisingly, the measures of canopy size were not related to current annual increment. In contrast, basal area (or standing volume) was correlated to canopy size.

OBJECTIVE

1) Determine the effects of planting density (4 x 4, 6 x 6, 8 x 8, and 10 x 10 foot spacing) on annual needle-fall biomass, stand-level basal area, intercepted radiation, and canopy openness for 25-year-old stands planted with two loblolly pine seed sources: North Carolina Coastal (NCC 8-01) and Oklahoma/Arkansas (O/A mix 4213).

2) Determine the relationships between measures of canopy size and stem current annual increment.

METHODS

The research site, near Broken Bow, Oklahoma is a droughty, mountain soil outside the native range of loblolly pine. Soils were classified as the Goldston–Carnasaw–Sacul association, which are upland, gravelly, moderately steep, silt loams with low water-holding capacity. The average annual rainfall of the area is 125 cm (49 in.), and the average annual temperature is 17 °C (63°F). Rainfall is usually adequate through May, but droughts 2–6 weeks in duration are common from June through October. Except for subsoiling at planting, no other stand-level treatments were applied. Stands were established in 1983. Treatments were a combination of spacing (4 x 4, 6 x 6, 8 x 8, and 10 x 10 foot) and seed source (Oklahoma-Arkansas vs. North Carolina Coastal). Between 3 and 5 plots per spacing x genotype combination were measured.

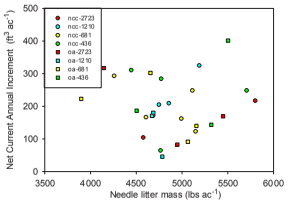
Diameters of all trees and heights of a subset of trees were periodically measured. For trees with height measurements, volume inside bark was calculated. Subsequently, treatment specific regressions between individual tree volume and basal area were used to calculate the volumes of the remaining trees. Current annual increment was calculated from successive dormant season measures of volume. Needle litter was measured using five randomly located 1 m^2 traps per plot and summed for a phenological year (1 April to 31 March). Canopy openness, an estimate of percent canopy not obscured by foliage, was measured using hemispherical photographs and the WinScanopy Canopy Analysis Software. Intercepted radiation was measured in September using the SunScan Canopy Analysis System. All variables were calculated at the plot-level. Regressions between canopy variables and current annual increment and basal area were conducted.

RESULTS

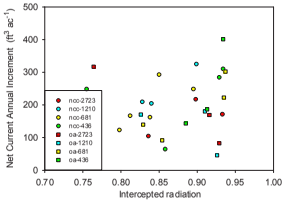
After 25 years, initial planting density no longer significantly affected needlefall, intercepted radiation, or canopy openness. Canopy openness was greater for the NCC than the OA genotypes (Tab. 1). [For more detailed analysis of volume and basal area, see poster by Will et al.] The canopy variables were not related to net current annual increment (Fig. 1) or gross current annual increment (data not shown). However, the canopy variables were correlated to basal area (Figure 2) and standing volume (data not shown).

Table 1. Mean attributes for stands from two genotypes (*Geno*) planted at four densities (*Spacing*). Genotypes were from North Carolina (NCC) and Arkansas/Oklahoma (OA). *BA 08* is basal area at the end of the 2008 growing season. *CAI 08 net* is net current annual increment for the 2008 growing season. *CAI 08 gr* is gross current annual increment (volume of trees that died during the 2008 growing season were added back into calculations. *Litter 06* is the foliage cohort that developed during the 2006 growing season and remained on the trees into early 2008. *Int rad* is the percent of radiation intercepted by the canopies. *Can open* is the percent of canopy not obscured by foliage.

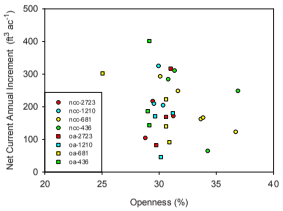
Geno	Spacing (tpa)	n	BA 08 ($ft^2 ac^{-1}$)	(s.e.)	CAI 08 net ($ft^3 ac^{-1} y^{-1}$)	(s.e.)	CAI 08 gr ($ft^3 ac^{-1} y^{-1}$)	(s.e.)	Litter 06 ($lbs ac^{-1}$)	(s.e.)	Int rad (%)	(s.e.)	Can open (%)	(s.e.)
NCC	436	4	180.2	12.8	226.8	26.4	283.7	2.5	4257	92	87	0.4	33.3	9.8
NCC	681	5	185.7	8.8	198.5	15.9	211.6	10.3	4171	35	84	0.2	33.2	12.9
NCC	1210	3	215.7	23.2	246.0	10.0	273.5	7.1	4264	25	86	0.1	30.0	2.1
NCC	2723	3	234.0	14.9	164.3	8.9	344.2	15.2	4342	54	89	0.2	29.8	5.7
OA	436	3	196.4	18.2	243.6	26.7	243.6	11.7	4418	39	91	0.1	29.1	0.6
OA	681	4	197.7	11.2	189.1	19.6	194.6	8.9	4059	53	89	0.3	29.3	12.0
OA	1210	3	199.4	13.7	132.3	14.2	153.7	4.4	4077	8	89	0.2	30.3	3.4
OA	2723	3	200.7	16.7	189.5	17.1	394.4	6.3	4191	69	87	0.3	30.5	2.1



Litter trap in front right

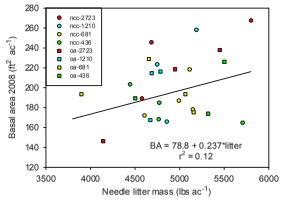


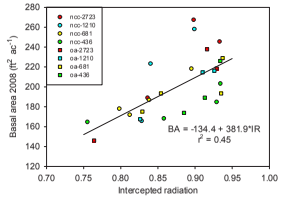
SunScan Canopy Analysis System (ceptometer)



Example hemispherical photograph

Figure 1. Relationship between current annual increment for the 2008 growing season and litter mass of the 2006 foliage cohort, radiation intercepted by the canopy, and canopy openness. Numbers in legend correspond to planting density. OA refers to Oklahoma/Arkansas and NCC refers to North Carolina Coastal.





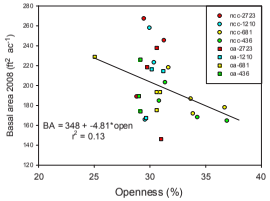


Figure 2. Relationship between basal area following the 2008 growing season and litter mass of the 2006 foliage cohort, radiation intercepted by the canopy, and canopy openness. Numbers in legend correspond to planting density. OA refers to Oklahoma/Arkansas and NCC refers to North Carolina Coastal.

DISCUSSION

While common in younger and/or more vigorous stands, we found no relationship between current annual increment and canopy size in these 25-year-old, dense stands. Possible reasons for lack of a relationship could be due to varying and sometimes intense intraspecific competition, environmental variation across the site, varying and sometimes large amounts of woody biomass relative to foliage, differences in partitioning to biomass components such as branch and roots, and previous stochastic events. Correlation between basal area and canopy size indicates canopy variables have some ability to predict basal area.

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We thank Phil Dougherty and Weyerhaeuser for establishing the study plots and the U.S. Forest Service for continued site maintenance.

USING EXISTING GROWTH MODELS TO PREDICT RCW HABITAT DEVELOPMENT FOLLOWING SITE PREPARATION: PITFALLS OF THE PROCESS AND POTENTIAL GROWTH RESPONSE

Benjamin O. Knapp and Joan L. Walker[1]

Abstract—Land managers throughout the Southeast are interested in restoring the longleaf pine (*Pinus palustris* Mill.) ecosystem, due in part to its value as habitat for the endangered red-cockaded woodpecker (*Picoides borealis*). In 2003, we established a study at Camp Lejeune, NC, to determine the effects of common site preparation treatments (mounding, bedding, herbicide, and chopping) on longleaf pine restoration on wet sites. We monitored mortality and measured root-collar diameter and height after three growing seasons. Although we found early increases in seedling growth in response to site preparation, it is unclear if these differences will persist throughout stand development. We extrapolated the results using existing growth-and-yield models to predict possible outcomes of site preparation on survival, dominant height, basal area, and quadratic mean diameter. Our projections suggest that stand structure for suitable foraging habitat could potentially be reached around 25 years earlier on the treatment with the most early growth (chopping/herbicide/bedding) when compared to the untreated check. However, liberal application of the models generates considerable uncertainty regarding interpretation of the results; we discuss the sources of error and suggest areas for needed research.

INTRODUCTION

Throughout the Southeastern United States, forest managers on lands supporting red-cockaded woodpeckers (*Picoides borealis*) (RCW) are increasingly interested in maintaining or creating RCW habitat. Favorable RCW habitat is commonly associated with a canopy dominated by longleaf pine (*Pinus palustris* Mill.), but historical land use and management practices have resulted in widespread conversion of longleaf pine forests to forests dominated by faster growing species such as loblolly pine (*Pinus taeda* L.) (Frost 1993). To increase RCW habitat quality, many land managers are now interested in rapidly reestablishing longleaf pine on sites dominated by other species.

Site preparation treatments are potentially useful management tools for increasing tree growth. Because site preparation is typically a single event that takes place just before seedlings are planted, seedling response is the strongest, and most often quantified, in the early years of stand establishment. A number of past studies have demonstrated the effectiveness of site preparation for increasing early growth rates and/or reducing early mortality of planted longleaf pine (e.g., Boyer 1988, Haywood 2007, Knapp and others 2006) and other southern pine seedlings (e.g., Knowe and others 1992, Pritchett 1979, Rahman and Messina 2006). In production forestry, rapid establishment and early growth shortens time to financial maturity and thereby increases the landowners' investment. However, land managers wishing to restore RCW habitat must consider the effects of site preparation on a temporal scale that depends on the ecological requirements of the RCW rather than economic returns.

To facilitate restoration of RCW habitat, site preparation must shorten the time required for a stand to develop from seedlings to trees of the size and structure utilized by RCWs. Although RCWs generally favor older trees in the forest for use as cavity trees (often 80 to 150 years old), stand criteria for good-quality foraging habitat may be reached substantially sooner. According to U.S. Fish and Wildlife recovery standard guidelines (U.S. Fish and Wildlife Service 2003), a group of RCWs will use from 49 to 120 ha of forest surrounding cavity trees as foraging habitat, depending on site productivity and habitat quality. Stand structure for good-quality foraging habitat includes, but is not limited to: (1) at least 45 pines/ha that are >35 cm in diameter at breast height (d.b.h.), 60 years old, and total at least 4.6 m^2/ha basal area; (2) basal area of all pines ≥25 cm d.b.h. is at least 9.2 m^2/ha; and (3) basal area of pines ≤25 cm d.b.h. is lower than 2.3 m^2/ha and below 50 stems/ha. In general, these guidelines describe stands that are dominated by large, old pines and include low densities of smaller pines or hardwoods. The quality of foraging habitat generally improves with tree size, as indicated by the requirement of a minimum number of large (>35 cm d.b.h.), old (≥60 years old) trees. However these guidelines suggest that 9.2 m^2/ha basal area of 25 cm d.b.h. trees is an important structural characteristic that may be a threshold for stands becoming RCW foraging habitat. It is not clear when artificially regenerated stands will reach the required structure for foraging habitat, or whether short-term effects of site preparation on longleaf pine seedlings will result in long-term differences in stand establishment.

The objectives of this study were to: (1) project theoretical growth and stand structure following site preparation using existing longleaf pine growth-and-yield models to predict development of RCW habitat and (2) discuss problems we encountered that introduced error and uncertainty into the results. This modeling approach was based on several assumptions: (1) the effects of site preparation persist throughout stand development, (2) survival and growth of current trees are solely determined by the size and number

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of trees in the previous time step, and (3) tree size variation within a stand is minimal so the quadratic mean diameter (QMD) and mean d.b.h. are approximately equal. Although a number of growth-and-yield models exist for longleaf pine, most are for stands ≥20 years old, and application is often restricted to specific site and stand conditions. Additionally, the biology of longleaf pine presents unique challenges for developing models of stand growth at young ages, due to the extended and often variable period of time in the grass stage (Goelz 2001). Consequently, we were liberal in application of existing models, resulting in greater error in our results. However, this exercise demonstrates theoretical scenarios for longleaf pine stand development after site preparation and clearly shows our need for a better understanding of the dynamics of stand development.

METHODS

Study Site

This study was conducted on Marine Corps Base Camp Lejeune, NC, located within the Atlantic Coastal Flatlands section of the Outer Coastal Plain Mixed Forest Province (Bailey 1995). All study sites were on Leon fine sand (sandy, siliceous, thermic, Aeric Alaquod), which is characterized by light-gray-to-white sand within the first 30 to 60 cm, underlain by a dark B horizon of organic accumulation. The B horizon was sufficiently cemented to form a hardpan of varying thickness (15 to 25 cm), and consequently this soil type is poorly drained, with internal drainage impeded by the hardpan layer (Barnhill 1992).

Experimental Design

The study design was a randomized complete block, with location as the block factor. Eight common site preparation treatments were randomly assigned to approximately 0.4-ha measurement plots in each block, with 15-m buffers between plots to reduce treatment overlap. Prior to site preparation, all blocks were harvested and sheared to remove standing vegetation. The eight experimental treatments were applied in August 2003: a check (no-site preparation), six treatments that combined two initial vegetation control treatments (chopping or herbicide) with three planting site conditions [flat (no additional treatment), mounding, or bedding], and a more intense treatment including chopping, herbicide, and bedding. In this paper, the treatments are often referred to by their initials as follows: flat or check (F), chopping and flat (CF), herbicide and flat (HF), chopping and mounding (CM), herbicide and mounding (HM), chopping and bedding (CB), herbicide and bedding (HB), and chopping, herbicide, and bedding (CHB).

Study plots were handplanted in December 2003 with container-grown seedlings from locally collected seed. Prior to planting, seedlings were culled to remove individuals of low vigor; average root-collar diameter of planted seedlings was 6.6 mm with a standard deviation of 1.2 mm.

Data Collection

Seedling survival was monitored in 2005, after 2 years of growth. In 2006, a subsample of 20 seedlings was randomly selected for third-year growth measurements. We used digital calipers to measure root-collar diameter (RCD) and a height pole to measure height to the terminal bud of all seedlings selected for measurement. Seedlings were determined to be in height growth when the terminal bud reached a height of 15 cm (Boyer 1988, Nelson and others 1985). Because most of the seedlings were in the grass stage, we calculated mean dominant height as the tallest half of surviving trees per plot. Boyer (1983) found that this fraction of grass stage seedlings represented a large number of vigorous seedlings that would likely become dominant and codominant canopy trees. Mean survival, RCD, and dominant height are summarized by treatment in table 1.

At four additional 10-year-old longleaf pine plantations, we randomly selected two 100-m^2 sampling plots to measure tree growth at age 10. Within each sampling plot, we marked each tree with a numbered aluminum tag and recorded RCD, d.b.h., and total height. The 10-year-old plantations were either bedded or not prepared prior to planting, and all plantations were on Leon soils.

Model Selection and Application

We searched the literature for the most appropriate models for our stand and site types. To our knowledge, Brooks and Jack (2006) developed the only model available to project stand growth and development for stands younger than 9 years old. Because models do not exist for the specific conditions of our study sites, we were liberal with model application and describe model assumptions that may be violated in table 2.

Table 1—Mean trees/ha, root-collar diameter, and dominant height used as starting points for projecting growth

Treatment	Trees/ha[a]	Root-collar diameter	Dominant height
		mm	*m*
CB	876	29.1	0.188
CF	819	18.9	0.031
CHB	782	35.8	0.645
CM	788	25.4	0.126
F	812	17.5	0.018
HB	776	34.0	0.400
HF	795	23.6	0.101
HM	777	30.6	0.299

CB = chopping and bedding; CF = chopping and flat; CHB = chopping, herbicide, and bedding; CM = chopping and mounding; F = flat; HB = herbicide and bedding; HF = herbicide and flat; HM = herbicide and mounding.

[a] Trees/ha were calculated from second-growing season survival (2005); root-collar diameter and dominant height were measurements taken 3 years after planting (2006).

Table 2—Description of models used to project stand growth for our study

Model	Variables[a]	Stand characteristics[b]	Site description[b]	Possible model violations[c]
Brooks and Jack (2006)	- Survival - Dominant height - Basal area	- Age 2 to 19 - Stand density 674 to 2,322 TPH - Basal area 1.2 to 31.2 m^2/ha	- Well-drained soils - Southwest Georgia	- Seedlings used in model development were ≥1.4 m tall, i.e., all were out of the grass stage. Our measurements were primarily seedlings in the grass stage; we calculated basal area from root-collar diameter. - Our study sites are poorly drained. Growth may differ based on drainage.
Lohrey and Bailey (1977)	- Survival	- Age 16 to 38 - Planting density from 618 to 6,178 TPH - Surviving density from 74 to 3,823 TPH - Unthinned plantations	- Site indices (25 years) from 9 to 22 m - Central LA and east TX	- We used this model to project survival to age 50, extrapolating past the maximum age used in model development - The model was developed in a different region than our study.
Farrar (1985)	- Basal area	- Age 11 to 90 - Basal area from 3.7 to 37.0 m^2/ha - Even-age natural stand - Period thinning	- Site indices (50 years) from 14 to 29 m - Regionwide study from east gulf region	- Model developed in naturally regenerated stands in the east Gulf Region. Site and stand conditions are different from our study. - Model developed from stands thinned on 5-year intervals; the author suggests restricting use of this model to short growth periods, not to exceed 30 years.

TPH = trees/ha.

[a] Represent the variables we used each model to project.

[b] Describe important information about the stands/sites used in model development.

[c] Describes some possible sources of error introduced into our projections.

Projections of QMD and basal area were used as a gauge of RCW habitat suitability, assuming that 9.2 m^2/ha basal area of 25 cm d.b.h. longleaf pine trees is an appropriate threshold for good-quality foraging stand structure.

Survival—To project survival to age 19, we used a model that projects future number of trees from stand age and current number of trees, developed by Brooks and Jack (2006) (equation 1):

$$N_2 = N_1 * Exp\left\{\alpha_1\left(\left(\frac{A_2}{10}\right)^{\alpha_2} - \left(\frac{A_1}{10}\right)^{\alpha_2}\right)\right\} \quad (1)$$

where
N_2 = projected survival in trees/ha at age A_2
N_1 = current trees/ha at age A_1
A_1 = stand age at the start of the growth period
A_2 = stand age at the end of the growth period
$\alpha_1 = -0.206745$, and $\alpha_2 = 0.360652$

To extend survival projections from age 19 to age 60, we used a model developed for unthinned longleaf pine plantations by Lohrey and Bailey (1977) (equation 2):

$$N_2 = N_1\left\{Sin^2\left[\frac{\pi}{2} + \left(1 - \frac{A_1}{A_2}\right) * \left(\beta_1 + \beta_2 * \sqrt{N_1} + \beta_3 * A_1 + \beta_4 * (A_1)^2\right)\right]\right\} \quad (2)$$

where
$\beta_1 = -2.827365$
$\beta_2 = -0.032141$
$\beta_3 = 0.221332$
$\beta_4 = -0.004125$

Dominant height—Brooks and Jack (2006) used a modified Chapman-Richards height/age projection function for other southern pines (Pienaar and Shiver 1980) to predict dominant height. Future dominant height is projected from stand age and current dominant height for plantations ages 2 to 19, as follows (equation 3):

$$DHT_2 = DHT_1\left[\frac{1 - Exp(\lambda_1 * A_2)}{1 - Exp(\lambda_1 * A_1)}\right]^{\lambda_2} \quad (3)$$

where
DHT_2 = projected dominant height at age A_2
DHT_1 = current dominant height at age A_1
$\lambda_1 = -0.07576$
$\lambda_2 = 2.099041$

Basal area—We used a model developed by Brooks and Jack (2006) to project basal area to age 19. This model predicts future basal area from current basal area, current and future dominant height, and current and future survival for plantations age 2 to 19 (equation 4):

$$BA_2 = Exp\{Ln(BA_1)+\delta_1(Ln(DHT_2)-Ln(DHT_1))+\delta_2(Ln(N_2)-Ln(N_1))\} \quad (4)$$

where
BA_2 = projected basal area at age A_2
BA_1 = projected basal area at age A_1
δ_1 = 1.817699
δ_2 = 7.398342

In applying this model, we calculated current basal area from measurements of RCD, with the assumption that basal area calculated from RCD could be used in place of basal area calculated from d.b.h. However, taper of the tree stem will cause diameter at the root collar to be larger than d.b.h., and consequently, basal area projected from RCD would be substantially larger than basal area calculated from d.b.h.

To rectify this, we followed a number of steps to convert basal area calculated from RCD to an estimated basal area from d.b.h. First, we converted the basal area projected to age 10 (from equation 4) to QMD, which would represent mean RCD at age 10. Then we used the data we collected from 10-year-old plantations and simple linear regression to develop the following relationship between RCD and d.b.h. at age 10 (equation 5):

$$DBH = -0.6526 + 0.7405(RCD) \quad (5)$$

$$r^2 = 0.86;\ n = 143;\ \text{SSE} = 312.12;\ P < 0.0001$$

Using this relationship, we predicted d.b.h. at age 10 from the projected RCD and converted this back to basal area. Under the assumption that the relationship between RCD and d.b.h. was independent of age, we projected basal area at age 10 backward to age 3 and forward to age 19 using equation 4.

The model we selected for projecting basal area past age 19 was developed for a variety of stand ages (11 to 90), site indices (13.7 to 29.0 m, base age 50) and densities (3.7 to 37.9 m^2/ha basal area) in the east gulf region (Farrar 1985) (equation 6):

$$BA_2 = \left\{\frac{\theta_1}{\theta_2} - \left[\frac{\theta_1}{\theta_2} - (BA_1)^{(1-\theta_3)}\right] * \left(\frac{A_2}{A_1}\right)^{(-\theta_2(1-\theta_3))}\right\}^{\left(\frac{1}{1-\theta_3}\right)} \quad (6)$$

where
θ_1 = –1.0007
θ_2 = –5.6643
θ_3 = 1.3213

This model was designed to predict longleaf pine growth in natural stands with periodic thinning. In this model, basal area is predicted from stand age and current basal area using a modified form of the Chapman-Richards growth function (Pienaar and Turnbull 1973).

Quadratic mean diameter—Projected basal areas were converted to QMD and plotted for each treatment.

RESULTS AND DISCUSSION

Both models used to project longleaf pine survival followed a reverse "J" shaped curve, with mortality greatest early in the growth period and slowing down over time (fig. 1). Previous studies have reported the greatest longleaf pine mortality in

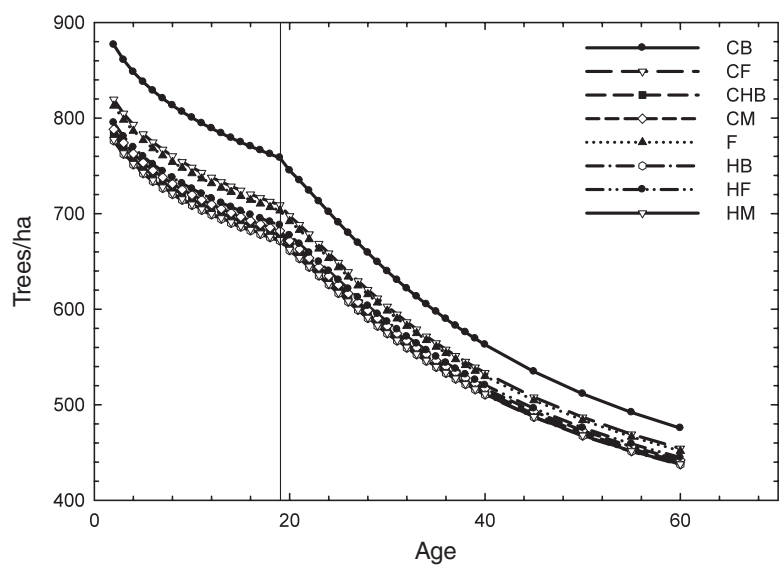

Figure 1—Trees per hectare projected from age 2 to age 60. Vertical line at age 19 represents a change in model from Brooks and Jack (2006) to Lohrey and Bailey (1977). (CB = chopping and bedding; CF = chopping and flat; CHB = chopping, herbicide, and bedding; CM = chopping and mounding; F = flat; HB = herbicide and bedding; HF = herbicide and flat; HM = herbicide and mounding)

the first year after planting (Boyer 1988, Knapp and others 2006), followed by a fairly low mortality rate through age 20 (Wilhite 1976). By age 60, projected survival ranged from 437 to 475 trees/ha, a level of stocking that would be unusually high for stands managed for RCWs. For example, a uniform stand with 25-cm d.b.h. trees requires only around 270 trees/ha to maintain 13.8 m^2/ha of basal area. It is likely that managers would periodically harvest to reduce stand density, allowing residual trees more resources for growth. Tree density would therefore be dictated by management activities rather than natural mortality and would not limit RCW habitat development.

Traditional growth models commonly use site index functions to predict dominant height (Farrar 1981, U.S. Forest Service 1976) but are unable to accurately account for changes in site quality caused by site preparation. Boyer (1980, 1983) compared height over age curves of young longleaf pine plantations established on old fields, mechanically prepared cutover forests, and unprepared cutover forests and found that site index curves were affected by site history/preparation as well as site quality. The Brooks and Jack (2006) model (equation 3) projected future dominant height from current dominant height rather than site index, thereby allowing us to account for differences in site quality resulting from site preparation.

Projected dominant height at age 19 was quite variable among the treatments, ranging from virtually no height growth on CF and F to over 10 m on CHB (fig. 2). Projections for some treatments were lower than expected. For example, it is unlikely that dominant height of a 19-year-old stand would remain below 2 m, as projected for CM, CF, HF, and F, unless seedlings never emerged from the grass stage. On sites with intense competition, it is possible that grass stage emergence would not occur without site improvement, i.e., site preparation. However, it is also likely that error was introduced into our projections by applying the Brooks and Jack (2006) model (equation 3) to data from grass stage seedlings. Treatments with age-3 mean dominant height >15 cm (CHB, HB, HM; table 1) were likely to have a greater proportion of seedlings out of the grass stage and result in more accurate projections of dominant height. On CHB, in which the majority of seedlings had emerged from the grass stage by age 3, our projection of dominant height at age 19 was similar to the dominant height of 19-year-old longleaf pine reported in a study conducted on Leon sand in northeastern Florida (Wilhite 1976), suggesting that model accuracy may be greatly improved as seedlings emerge from the grass stage.

Basal area and QMD growth projections were very different among the treatments, ranging from 5.3 to 23.7 m^2/ha basal area (fig. 3) and 9.8 to 21.1 cm QMD (fig. 4) at age 19. In the Wilhite (1976) study, 20-year-old longleaf pine plantations had a basal area of 14.5 m^2/ha and d.b.h. of approximately 12.7 cm. Prior to planting, those sites were prepared by scarifying the soil several times with an agricultural disk harrow and mechanically removing saw palmetto [*Serenoa repens* (Bartram) Small]. Such site preparations would fall within the range of site preparation intensity used in our study, and therefore it is not surprising that the values reported by Wilhite (1976) are within the range of projected values for basal area and QMD reported in our study.

When considering RCW habitat suitability, all treatments were projected to reach a basal area of 9.2 m^2/ha by around age 25 (fig. 3), suggesting that tree diameter will be a more

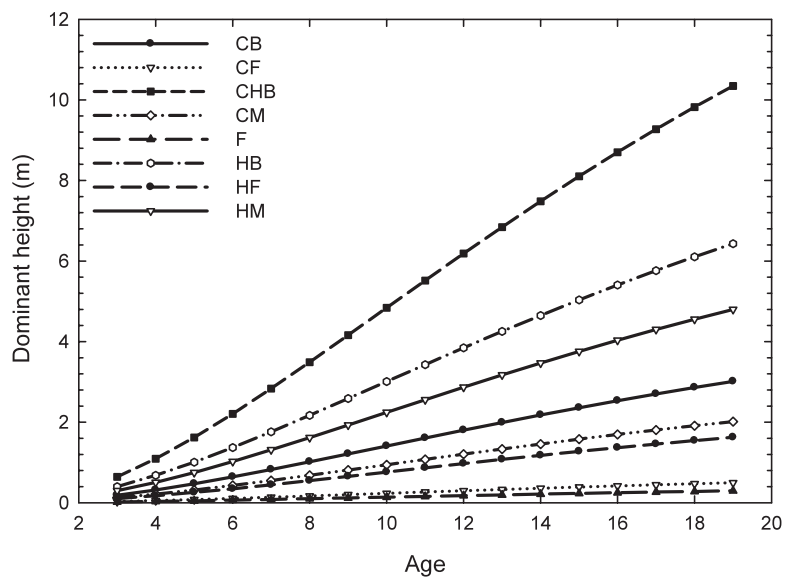

Figure 2—Dominant height (m) projected for ages 3 to age 19 using the model developed by Brooks and Jack (2006). (CB = chopping and bedding; CF = chopping and flat; CHB = chopping, herbicide, and bedding; CM = chopping and mounding; F = flat; HB = herbicide and bedding; HF = herbicide and flat; HM = herbicide and mounding)

important indicator of when these stands will become good-quality foraging habitat. For instance, CHB is projected to reach a basal area of 9.2 m^2/ha around age 11, at which point QMD is only 13.8 cm (fig. 4). Assuming that stands will first become usable as RCW habitat when QMD reaches 25 cm, our growth projections indicate drastic treatment differences in time to habitat suitability. Three treatments, CHB, HB, and HM may be expected to reach suitable size for foraging habitat by around age 30, with the fastest growing treatment (CHB) projected to reach 25 cm QMD at around age 25. On the other hand, the slowest growing treatments, F and CF, will not be suitable for RCW habitat until around age 50.

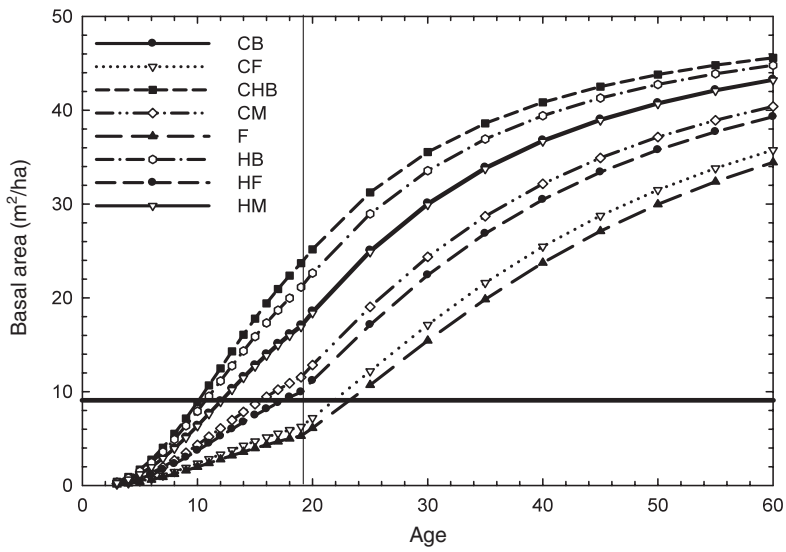

Figure 3—Basal area (m^2/ha) projected from age 3 to age 60. The vertical line at age 19 represents a change in model from Brooks and Jack (2006) to Farrar (1985). The horizontal line at 9.2 m^2/ha represents the lower basal area limit recommended for good-quality RCW habitat. (CB = chopping and bedding; CF = chopping and flat; CHB = chopping, herbicide, and bedding; CM = chopping and mounding; F = flat; HB = herbicide and bedding; HF = herbicide and flat; HM = herbicide and mounding)

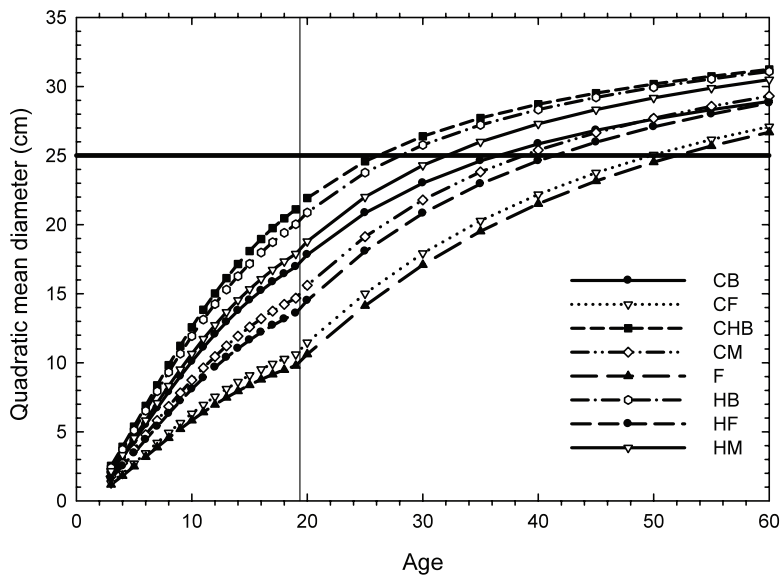

Figure 4—Quadratic mean diameter (cm) projected from age 3 to age 60. The vertical line at age 19 represents a change in model from Brooks and Jack (2006) to Farrar (1985). The horizontal line at 25 cm represents the lower basal area limit recommended for good-quality RCW habitat. (CB = chopping and bedding; CF = chopping and flat; CHB = chopping, herbicide, and bedding; CM = chopping and mounding; F = flat; HB = herbicide and bedding; HF = herbicide and flat; HM = herbicide and mounding)

Our results demonstrate theoretical differences in stand development following site preparation, but we acknowledge the uncertainty introduced by such liberal application of existing models (table 2). Error in our projections caused by exceeding model limitations was compounded by combining models for long-term extrapolation. Perhaps the most serious problem with modeling stand development of young longleaf pine is the unpredictable growth of grass stage seedlings. Currently, no models are available to translate grass stage measurements (primarily RCD) to projections of sapling/tree measurements (height/d.b.h.). Although it is accepted that grass stage emergence typically occurs when the root collar approaches 2.5 cm (Boyer 1990), emergence at the stand level and subsequent growth patterns are not fully understood and therefore difficult to model. The Brooks and Jack (2006) model (equation 4) was developed to project stand growth from a young age but assumes that seedlings have reached a height of at least 1.4 m (d.b.h. height). Many seedlings in our study were measured in the grass stage, violating that assumption and reducing the reliability of resulting model projections.

In this modeling exercise, we assume that effects of site preparation will last throughout stand development; however, there is evidence that early increases in longleaf pine growth following site preparation do not persist throughout stand development (Boyer 1996). For example, Boyer (1985) studied the effects that timing of release from competition had on short- and long-term longleaf pine growth response by comparing growth following complete hardwood competition control applied at ages 1, 2, 3, 4, and 8, and an unreleased check. At age 10, dominant tree height was greatest on treatment plots released at age 1 and decreased with each subsequent year of release. By age 31, however, dominant height was similar among all released treatments but remained significantly greater than the unreleased check. It is possible that as stands develop and canopies close, competition from understory species is reduced and growth is more strongly influenced by site productivity and intraspecific competition than by understory competition (Boyer 1983). However, it remains unclear how long the effects of mechanical treatments that change microtopography, i.e., bedding and mounding, would impact site productivity and tree growth.

An important benefit of increased early growth is a reduction in the length of time that seedlings remain in the grass stage. Longleaf pine seedlings have the ability to persist in the grass stage for over 10 years in unfavorable conditions (Pessin 1944) and in extreme cases may never enter height growth. Emergence from the grass stage is critical to stand establishment, and site preparation treatments may be one way to ensure successful height growth. On sites with extreme competition, improved chances for emergence may justify use of site preparation, regardless of subsequent growth benefits. It is logical that early grass stage emergence should correspond with shorter time to maturity. However, the ability of longleaf pine to make up for early growth deficits (Boyer 1983, 1985, 1996) suggests that this may not be the case and highlights our lack of knowledge about the early stages of stand development.

CONCLUSIONS

Our model projections demonstrate theoretical differences in stand development following site preparation but also make clear some problems associated with modeling growth of young longleaf pine. Assuming that stands become suitable foraging habitat when trees ≥25 cm d.b.h. reach a basal area of 9.2 m^2/ha, we projected that CHB would become habitat 25 years faster than the untreated check. Our results suggest that site preparation may be a useful tool for land managers wishing to shorten the time required to grow longleaf pine plantations into RCW habitat on this site type. However, we acknowledge the uncertainty of our results and intend for this study to raise questions for future research rather than provide concrete management guidelines for landowners.

ACKNOWLEDGMENTS

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LITERATURE CITED

Bailey, R.G. 1995. Description of the ecoregions of the United States. 2d ed. Rev. Misc. Publ. 1391. Washington, DC: U.S. Department of Agriculture. 108 p.

Barnhill, W.L. 1992. Soil survey of Onslow County, North Carolina. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 139 p.

Boyer, W.D. 1980. Interim site-index curves for longleaf pine plantations. Res. No. SO-261. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 5 p.

Boyer, W.D. 1983. Variations in height-over-age curves for young longleaf pine plantations. Forest Science. 29: 15–27.

Boyer, W.D. 1985. Timing of longleaf pine seedlings release from overtopping hardwoods: a look 30 years later. Southern Journal of Applied Forestry. 9: 114–116.

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 For. Res. Pap. 76. [Place of publication unknown]: Georgia Forestry Commision. Research Division. 8 p.

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

Boyer, W.D. 1996. Longleaf pine can catch up. In: Kush, J.S., comp. Proceedings of the first Longleaf Alliance conference—longleaf pine: a regional perspective of challenges and opportunities. Longleaf Alliance Report 1: 28–29.

Brooks, J.R.; Jack, S.B. 2006. A whole stand growth and yield system for young longleaf pine plantations in southwest Georgia. 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: 317–318.

Farrar, R.M., Jr. 1981. A site-index function for naturally regenerated longleaf pine in the east gulf area. Southern Journal of Applied Forestry. 5: 150–153.

Farrar, R.M., Jr. 1985. Volume and growth predictions for thinned even-aged natural longleaf pine stands in the east gulf area. Res. Pap. SO-220. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 12 p.

Frost, C.C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. In: Hermann, S.M., ed. Proceedings of the 18th Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station: 17–43.

Goelz, J.C.G. 2001. Longleaf pine plantations: growth and yield modeling in an ecosystem restoration context. In: LeMay, V.; Marshall, P., eds. Proceedings of the forest modeling for ecosystem management, forest certification and sustainable management conference. Vancouver, BC, Canada: University of British Columbia, Forest Resources Management Department: 219–232.

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.

Knapp, B.O.; Wang, G.G.; Walker, J.L.; Cohen, S. 2006. Effects of site preparation treatments on early growth and survival of planted longleaf pine (*Pinus palustris* Mill.) seedlings in North Carolina. Forest Ecology and Management. 226: 122–128.

Knowe, S.A.; Shiver, B.D.; Kline, W.N. 1992. Fourth-year response of loblolly pine following chemical and mechanical site preparation in the Georgia Piedmont. Southern Journal of Applied Forestry. 16: 99–105.

Lohrey, R.E.; Bailey, R.L. 1977. Yield tables and stand structure for unthinned longleaf pine plantations in Louisiana and Texas. Res. Pap. SO-133. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 53 p.

Nelson, R.L.; Knowe, S.A.; Gjerstad, D.H. 1985. Planted longleaf pine seedlings respond to herbaceous weed control using herbicides. Southern Journal of Applied Forestry. 9: 236–240.

Pessin, L.J. 1944. Stimulating early height growth of longleaf pine seedlings. Journal of Forestry. 42: 95–98.

Pienaar, L.V.; Shiver, B.D. 1980. Dominant height growth and site index curves for loblolly pine plantations in the Carolina flatwoods. Southern Journal of Applied Forestry. 4: 54–59.

Pienaar, L.V.; Turnbull, K.J. 1973. The Chapman-Richards generalization of von Bertalanffy's growth model for basal area growth and yield in even-aged stands. Forest Science. 19: 2–22.

Prichett, W.L. 1979. Site preparation and fertilization of slash pine on a wet savanna soil. Southern Journal of Applied Forestry. 3: 86–90.

Rahman, M.S.; Messina, M.G. 2006. Intensive forest management affects loblolly pine (*Pinus taeda* L.) growth and survival on poorly drained sites in southern Arkansas. Southern Journal of Applied Forestry. 30: 79–85.

U.S. Department of Agriculture Forest Service. 1976. Volume, yield, and stand tables for second-growth southern pines. Misc. Publ. 50. Washington, DC. 202 p.

U.S. Fish and Wildlife Service. 2003. Red-cockaded woodpecker (*Picoides borealis*) recovery plan. 2d rev. Atlanta. 296 p.

Wilhite, L.P. 1976. Slash, loblolly, longleaf, and sonderegger pines 20 years after planting on Leon sand in northeastern Florida. Res. Pap. SE-153. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 8 p.

REGENERATING OAK-DOMINATED FORESTS USING IRREGULAR, GAP-BASED SILVICULTURAL SYSTEMS

John M. Lhotka, Michael R. Saunders, John M. Kabrick, and Daniel C. Dey[1]

Abstract—Throughout the Eastern United States, practitioners have primarily focused on using uniformly applied even-aged approaches to regenerate oak species. Irregular, gap-based silvicultural systems offer an alternative that retains continuous canopy cover, creates heterogeneous forest structure, and provides multiple income flows over a rotation. Although commonly used in Europe, there are few documented applications of these systems in oak forests of North America. Our objective is to establish a regionwide trial of two variants of an irregular group shelterwood system that successively expands gaps outward until the entire stand has been regenerated. Both variants will use cutting cycles that are 10 percent of the rotation length and harvest approximately 20 percent of the area at each entry. Initial gap size for both variants will be 2.0 to 2.2 times the dominant canopy height, and the overstory will be removed in either one or two stages. In addition, the midstory canopy will be removed from areas of the surrounding forest matrix scheduled for harvest in the next cycle for both variants. Soil scarification, competition control, and/or underplanting may be used in areas lacking sufficient advance reproduction. These treatments will create a light gradient extending from the gap center into the surrounding matrix that should allow intolerant species, like yellow-poplar (*Liriodendron tulipifera* L.) or shortleaf pine (*Pinus echinata* Mill.), to prevail in areas high in direct beam radiation and oaks to dominate in the diffuse light conditions created on the gap margins and within the surrounding matrix. Successive expansions will release the developing oak reproduction and will result in a stand with a diverse mixture of commercial species.

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Regenerating Oak Dominated Forests Using Irregular, Gap-Based Silvicultural Systems

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Project Overview: Throughout the eastern United States, practitioners have primarily focused on using uniformly applied even-aged approaches to regenerate oak species. Irregular, gap-based silvicultural systems offer an alternative that retains continuous canopy cover, creates heterogeneous forest structure, and provides multiple income flows over a rotation. Although commonly used in Europe, there are few documented applications of these systems in oak forests of North America. Our objective is to establish a region-wide trial of two variants of an irregular group shelterwood system that successively expand gaps outward until the entire stand has been regenerated. Both variants will use cutting cycles that are 10% of the rotation length and harvest approximately 20% of the area at each entry. Initial gap size for both variants will be 2.0-2.2 times the dominant canopy height and the overstory will be removed in either one or two stages. In addition, the midstory canopy will be removed from areas of the surrounding forest matrix scheduled for harvest in the next cycle for both variants. Soil scarification, competition control, and/or underplanting may be used in areas lacking sufficient advance reproduction. These treatments will create a light gradient extending from the gap center into the surrounding matrix that should allow intolerant species, like yellow-poplar or shortleaf pine, to prevail in areas high in direct beam radiation and oaks to dominate in the diffuse light conditions created on the gap margins and within the surrounding matrix. Successive expansions will release the developing oak reproduction and will result in a stand with a diverse mixture of commercial species.

Background and Justification

- Gap-based, hybrid silvicultural systems incorporate aspects of even- and uneven-aged regeneration methods and are an attractive alternative for oak dominated forests
- Expanding group shelterwood is a hybrid system where cuttings are applied in a pattern of expanding gaps or patches
- Treatment formulation builds upon shelterwood, group selection, a patch clearcutting research
- Expanding group shelterwood results in resource gradients from gap centers extending into the forest matrix (Figure 1)
- Size and juxtaposition of gaps, and timing of gap expansions can be manipulated to regenerate a wide variety of species, from intolerants in gap centers to intermediates and shade tolerants along gap edges
- Potential benefits: 1) maintenance of continuous forest cover, 2) species diversity and structural complexity, and 3) multiple income flows over rotation

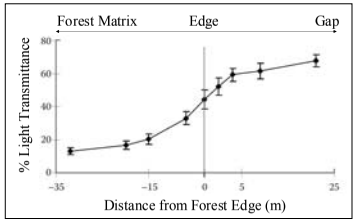


Figure 1. Hypothesized relationship between light transmittance and distance from forest edge (Adapted from Schmid, I., K. Klumpp, and M. Kazda. 2005. Light distribution within forest edges in relation to forest regeneration. J. For. Sci. 51(1):1-5)

Objectives

The long-term goal of this project is to develop alternative silvicultural systems that can be used to successfully regenerate oak within the Central Hardwood Forest Region. This study is novel in that it uses common treatment regimes and monitoring protocols at multiple sites around the region that differ significantly in species composition, competitors, and environments. It is hoped that this design will lead to interdisciplinary investigations of other components of oak ecosystems. Primary objectives include:

1. Test the effectiveness of two expanding group shelterwood variants for regeneration of oak-dominated communities in a variety of ecosystems and forest types.
2. Characterize biotic and abiotic factors that affect the abundance and spatial distribution of advance oak regeneration within natural stands. Estimate the range of conditions under which oak can compete successfully and be recruited into the successive stand.
3. Determine the effect of expanding group shelterwood systems on other biotic components of these ecosystems.

We actively seek collaborators interested in expanding the regional extent of the study and/or quantifying the response of wildlife and other biotic components to the study's gap-based silvicultural treatments

Common Treatment Regimes

Study Areas (Figure 2):

- Mature, oak-dominated upland stands without harvests over past 10-20 years
- A minimum of 30 acres to accommodate treatments

Expanding Group Shelterwood Treatments:

- Initial Gap/Group Size: 2.0 to 2.2 times dominant tree height (0.5 – 1.0 ac)
- Two Variants of Group Shelterwood Tested:
  1. Entire overstory removed in single cut
  2. Two-stage removal with an initial shelterwood cut of 40 to 50 % residual stocking
- Preparatory Cuts: Midstory removal in areas scheduled for harvest in next cycle
  - Goals: Work in synergy with gap treatments to increase light availability and seedling survival/growth in forest matrix
- Cutting Regime: Area controlled with 5 to 6, ten-year cutting cycles (Figure 3)
- Soil scarification, underplanting, and/or chemical competition control may be used in areas lacking sufficient advance reproduction.

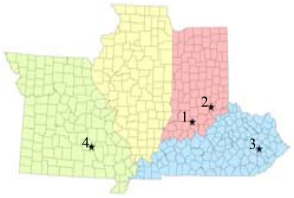
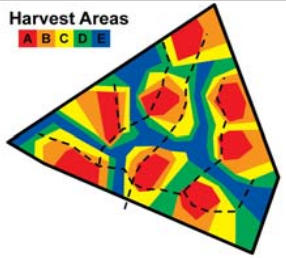


Figure 2. Proposed Study Locations

1. Southern Indiana Purdue Agricultural Center
2. Southwest Purdue Agricultural Center
3. Robinson Forest (University of Kentucky)
4. Sinkin Experimental Forest (Forest Service)



Harvest Area	Variant 1		Variant 2		
	Prep	Overstory	Prep	Establish	Overstory
A	-	1*	-	1	2
B	1	2	1	2	3
C	2	3	2	3	4
D	3	4	3	4	5
E	4	5	4	5	6

* Ten-year harvest cycle number

Figure 3. Conceptual harvest layout and cutting cycles for expanding group shelterwood systems

Harvest areas (A – E) and cutting cycle for two variants of expanding group shelterwood. Harvests include a preparatory cut removing all midstory trees (Prep), an establishment cut with 40 to 50% residual stocking (Establish), and an overstory removal cut (Overstory). Cutting cycles are 10 years. Dashed black lines are one potential permanent skid trail layout.

Monitoring Protocols

Stand-Level

- Stands sampled with minimum of 5% area sample on a five year inventory cycle
- Overstory trees, saplings, tree regeneration, and herbaceous vegetation (Figure 4)

Gap/Group Level

- Sampling transects extending from gap center into surrounding matrix
  1. Understory environmental factors such as photosythetically active radiation
  2. Establishment and development of natural regeneration

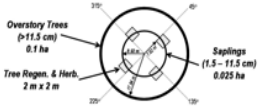


Figure 4. Sample Plot Layout

DEVELOPMENT OF VOLUME EQUATIONS USING DATA OBTAINED BY UPPER STEM DENDROMETRY WITH MONTE CARLO INTEGRATION: PRELIMINARY RESULTS FOR EASTERN REDCEDAR

Thomas B. Lynch, Rodney E. Will, and Rider Reynolds[1]

Abstract—Preliminary results are given for development of an eastern redcedar (*Juniperus virginiana*) cubic-volume equation based on measurements of redcedar sample tree stem volume using dendrometry with Monte Carlo integration. Monte Carlo integration techniques can be used to provide unbiased estimates of stem cubic-foot volume based on upper stem diameter measurements obtained at randomly selected stem heights. Monte Carlo integration with importance sampling and antithetic variates was used to obtain sample tree volume estimates in this study. Importance sampling is a variance reduction technique that uses a proxy taper function (in this case a paraboloid) to randomly select the most influential upper stem sample diameters. Antithetic variates use negatively correlated upper stem measurements to further reduce the variance of stem volume estimates. The estimator was revised to use tree d.b.h. rather than stump diameter. Since volume estimates from these techniques are unbiased, they can be used as dependent variables to estimate the parameters in a standard volume equation for eastern redcedar. Data were obtained from more than 30 sample trees using a Wheeler Pentaprism to obtain two upper stem diameter measurements for each sample tree. Sample tree total height and d.b.h. were also measured. Preliminary results indicate an R^2 (0.85), which is somewhat lower than that usually obtained in studies where trees are felled and measured deterministically. However, since the Monte Carlo integration estimates are unbiased the mean regression line is probably well located even though the variance about the line may be somewhat greater than would be the case for felled tree data.

INTRODUCTION

A cubic-foot volume equation for eastern redcedar (*Juniperus virginiana*) was desired for use in the Payne County, OK, area. Since no suitable equations had been developed, it was decided to use upper stem dendrometry on standing redcedar trees to develop a cubic-foot volume equation. Monte Carlo integration with importance sampling and antithetic variates (Van Deusen and Lynch 1987) provides an unbiased and efficient method of estimating individual tree stem volumes from only two randomly chosen upper stem measurements. Importance sampling selects upper stem diameter measurements using a proxy taper function so that measurements representing greater volume are more likely to be chosen (generally these are lower on the bole). Through use of the antithetic variates technique, the two randomly chosen upper stem diameter measurements are negatively correlated. This results in variance reduction of the volume estimate because the variance of the sum of two negatively correlated random variables is less than the sum of their variances.

For this application it was desired to modify the estimating equations described by Van Deusen and Lynch (1987) so that the proxy taper function is based on tree total height and d.b.h. rather than stump diameter. As suggested by Van Deusen and Lynch (1987) a paraboloid was used as a proxy taper function. The following formula can be used to compute the cubic-foot volume of a paraboloid having the same height and d.b.h. as the tree of interest:

$$V_{par} = \frac{\pi D^2}{2 \times 576}\left(\frac{H}{1-\frac{4.5}{H}}\right)$$

where
V_{par} = cubic-foot volume of paraboloid
D = d.b.h. (4.5 feet) in inches
H = total tree height in feet

The estimation process begins by selecting a random variable u which is uniformly distributed between zero and 1. This number is used to select two antithetic random heights at which upper stem diameters will be measured. The following formula is used to generate these two heights:

$$h_1 = H \left[1-\sqrt{u}\right]$$
$$h_2 = H \left[1-\sqrt{(1-u)}\right]$$

where
h_1 = upper stem height 1 in feet
h_2 = upper stem height 2 in feet
u = random number uniformly distributed between 0 and 1
H = total tree height in feet

A Wheeler Pentaprism was used to measure upper stem heights on each redcedar sample tree at heights h_1 and h_2.

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The paraboloid volume given above was then adjusted by using ratios of measured squared diameter to paraboloid squared diameter:

$$\hat{V} = \frac{V_{par}}{2}\left[\frac{d_{m1}^2}{d_{p1}^2} + \frac{d_{m2}^2}{d_{p2}^2}\right]$$

where

$d_{p1} = D\sqrt{\frac{H-h_1}{H-4.5}}$, upper stem diameter (inches) of paraboloid at h_1

$d_{p2} = D\sqrt{\frac{H-h_2}{H-4.5}}$, upper stem diameter (inches) of paraboloid at h_2

d_{m1} = is measured upper stem diameter (inches) at h_1

d_{m2} = is measured upper stem diameter (inches) at h_2

h_2 = is upper stem height 2 in feet

$\hat{V}$ = estimated redcedar cubic-foot total stem volume

RESULTS AND DISCUSSION

At two locations in Payne County, OK, a total of 38 redcedar trees were subsampled on BAF = 10-factor point sample plots to provide a test of these methods. Upper stem diameters were measured at two randomly chosen antithetic heights so that estimated stem volumes could be obtained by using the equation above. Linear regression techniques were then used to estimate the parameters in the following combined variable (Avery and Burkhart 2002 p. 173) total volume equation:

$$\hat{V} = \underset{(0.535629)}{0.859397} + \underset{(0.000188)}{0.002674}\, D^2H$$

where

$\hat{V}$ = predicted cubic-foot volume outside bark

D = d.b.h. (inches)

H = total height (feet)

Numbers in parentheses under the equation above are the standard errors of estimated coefficients.

Fit statistics for the equation above were: coefficient of determination $R^2 = 0.85$, standard error $S_{y.x} = 1.81$, number of observations $n = 38$, calculated "F" statistic is $F = 201$, (significant at levels <.0001)

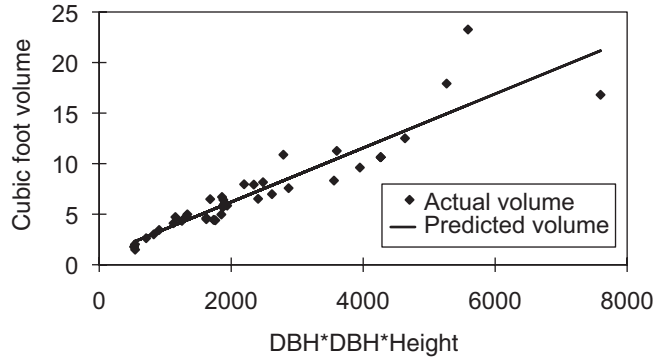

Figure 1—Actual vs. predicted cubic-foot volume for 38 redcedar sample trees.

The plot of the regression relationship vs. sample data indicated reasonable fit (fig. 1).

SUMMARY AND CONCLUSIONS

Estimation of sample tree volume by Monte Carlo integration with importance sampling and antithetic variates was adequate for preliminary development of a combined variable volume equation on this small redcedar dataset. The $R^2 = 0.85$ was not as good as that usually seen in datasets that use felled tree data, but since Monte Carlo estimation is unbiased, this method should give adequate results in terms of fitting the mean volume line. This method of obtaining volume observations for volume equation studies may be of interest in situations where the cost and time required to obtain felled tree data is prohibitive.

LITERATURE CITED

Avery, T.E.; Burkhart, H.E. 2002. Forest measurements. 5th ed. New York: McGraw-Hill. 456 p.

Van Deusen, P.C.; Lynch, T.B. 1987. Efficient unbiased tree-volume estimation. Forest Science. 33: 583–590.

DIAMETER-HEIGHT AND CROWN RELATIONSHIPS FOR LOBLOLLY PINE FROM NORTH CAROLINA AND OKLAHOMA-ARKANSAS SEED SOURCES NEAR THE WESTERN EDGE OF THE NATURAL RANGE

Thomas B. Lynch, Rodney E. Will, Thomas C. Hennessey, Robert Heinemann, Randal Holeman, Dennis Wilson, Keith Anderson, and Gregory Campbell[1]

Abstract—In southeastern Oklahoma, loblolly pine (*Pinus taeda* L.) has been planted extensively outside the western boundary of its natural range. Furthermore, many plantings have been based on seed sources such as North Carolina Coastal (NCC) rather than Oklahoma-Arkansas (OA). NCC plantings are also frequent in nearby areas of Oklahoma and Arkansas which are within the loblolly pine natural range. A loblolly pine seed source and density trial planted in 1983 on Carter Mountain near Broken Bow, OK, provided the opportunity to compare individual tree characteristics of NCC vs. OA seed sources in a location just beyond the western edge of the natural range. The study site is a rocky, mountain soil. North Carolina and OA seed sources were planted at 4- by 4-feet, 6- by 6-feet, 8- by 8-feet, and 10- by 10-feet spacings. Diameter at breast height (d.b.h.) of sample trees ranged from 4 to 13 inches, with heights ranging from approximately 30 to 70 feet. The study consists of 19 plots for which individual tree measurements were made in 2000, 2001, 2002, 2005 and 2008. D.b.h. was measured on each tree while heights and heights to live crown base were measured on a subset of trees on each plot. Nonlinear regression techniques were used to develop models relating tree heights and crown lengths to d.b.h. Dummy variables representing loblolly pine seed sources (NCC and OA) were included in height, crown, and local volume models where significant to represent seed source effects on total height, crown length, and cubic-foot volume inside bark. Dependent variables for regression analysis included measured height, crown length, and computed inside bark volume on individual trees. Inside bark cubic-foot total volumes were obtained by using the equation of Tasissa and others (1997). Individual tree volumes were then regressed against individual tree basal area with dummy variables to indicate seed source. Regression relationships between individual tree d.b.h. and height indicated a significant difference due to seed source, with the North Carolina source being approximately 7 percent taller across the d.b.h. range in these data. This indicates that the North Carolina seed source is outperforming the OA seed source in height growth even at the extreme western edge of the loblolly natural range. The average d.b.h.-height relationship also was significantly affected by density, but the North Carolina seed source was taller on average per given d.b.h. for all densities. In addition to being taller, for a given d.b.h. the average individual North Carolina trees had a significantly longer live crown (approximately 2 to 3 feet longer at d.b.h. 5 to 10 inches) and significantly more cubic stem content (from 6.6 percent at 12 inches d.b.h. to 8.8 percent at 5 inches d.b.h.) than the average individual tree from the Oklahoma seed source.

LITERATURE CITED

Tasissa, G.; Burkhart, H.E.; Amateis, R.L. 1997. Volume and taper equations for thinned and unthinned loblolly pine trees in cutover, site-prepared plantations. Southern Journal of Applied Forestry. 21: 146–152.

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RESPONSE TO PRESCRIBED BURNING OF 5-YEAR-OLD HARDWOOD REGENERATION ON A MESIC SITE IN THE SOUTHERN APPALACHIAN MOUNTAINS

W. Henry McNab, Erik C. Berg, and Ted M. Oprean III[1]

Abstract—Five years after a Southern Appalachian cove was regenerated, vegetation was dominated by a dense stand of yellow-poplar (*Liriodendron tulipifera*), which averaged 9,181±13,042 stems per acre, and other mesophytic hardwood seedlings and saplings. The stand was prescribed burned during late spring to improve habitat for turkey by reducing density of saplings to stimulate greater production of grasses and herbs. The prescribed fire completely killed about 69 percent of the saplings; the others were topkilled and produced basal sprouts that reclaimed much of the canopy growing space after one growing season. Regression analysis indicated that over a range of fire intensities the probability of mortality for yellow-poplar saplings was about twice that of other species. Results suggest that additional prescribed burns will be necessary to achieve the desired low density of arborescent vegetation to allow development of herbaceous species beneficial for wildlife.

INTRODUCTION

Recently regenerated stands on mesic sites in the Southern Appalachians are typically dominated by a mixture of mesophytic hardwoods consisting of yellow-poplar (*Liriodendron tulipifera*), black locust (*Robinia pseudoacacia*), sweet birch (*Betula lenta*), and red maple (*Acer rubrum*) (Beck and Hooper 1986). Although the dense sapling stands provide browse for white-tailed deer (*Odocoileus virginianus*), their forage value for other wildlife declines rapidly because canopy shading excludes desirable grasses, herbs, and legumes (Beck and Harlow 1981). Prescribed burning is sometimes used to manipulate vegetation for wildlife habitat with favorable results on dry pine-hardwood sites, (Keetch 1944, Van Lear and Waldrop 1989). Little information is available, however, on the response of arborescent vegetation to prescribed burning for wildlife habitat objectives in young, recently regenerated stands on mesic sites in the Southern Appalachians.

This report documents results of an operational prescribed burn to reduce density of hardwood regeneration for wildlife purposes on a mesic site in the Southern Appalachian Mountains. Our study of the prescribed burn had two objectives: (1) determine the change in density of yellow-poplar regeneration caused by the prescribed burn and (2) determine factors associated with mortality of hardwood saplings. The scope of our case study was limited to results 1 year following the prescribed fire.

METHODS

Study Area and Treatment

The study site was located in a large east-facing cove at 2,750 feet elevation in the Pisgah District of the Pisgah National Forest, adjacent to the Bent Creek Experimental Forest. Within the cove, topography was hilly and aspects ranged from northwest to southeast. Site index (index age of 50 years) was 90 feet for northern red oak (*Quercus rubra* L.) and 110 feet for yellow-poplar. The mature stand averaged 100 years of age and contained about 14,300 board feet per acre of sawtimber that consisted primarily of yellow-poplar (83 percent), with smaller amounts of northern red oak (10 percent), and chestnut oak (*Q. prinus*) (5 percent). Red maple (1 percent) and other low-grade hardwoods accounted for the remainder. The stand was regenerated using a shelterwood with reserves system in June 1995, with basal area of the residual overstory trees averaging about 80 square feet per acre. Merchantable timber was harvested from 16.4 acres of the 21-acre stand using a rubber-tired skidder to remove log-length products. Residual basal area was further reduced to about 40 square feet per acre in December 1996, as a result of salvaging windthrow caused by the remnants of Hurricane Opal (October 1995). Site preparation after harvest consisted of chainsaw-felling residual unmerchantable trees >3 inches diameter at breast height (d.b.h.); stems <3 inches d.b.h. and over 4.5 feet tall were killed with herbicide.

The stand had been regenerated successfully by advance regeneration released by removal of the overstory and newly established seedlings, mostly yellow-poplar. A low (<4 feet height) canopy of xerophytic species was present on about 29 percent of the study area, mainly on upper slopes. Mesophytic species, primarily yellow-poplar, dominated a similar proportion of the stand on moist lower slopes, forming dense “dog-hair” thickets of tall, spindly saplings (fig. 1); the remainder was a mixture of species. Wildlife management objectives specified greater amounts of grasses and herbaceous species suitable for “bugging” by turkey (*Meleagris gallopavo*) poults. Prescribed burning was selected as an appropriate method for reducing density of hardwood saplings to allow increased light to reach the forest floor and stimulate growth of grasses and forbs.

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Figure 1—Five years following the regeneration harvest on the cove study area, much of the vegetation on mesic sites consisted of thickets of yellow-poplar saplings that ranged from 5 to 15 feet in height and up to 2 inches d.b.h.

The stand was burned between 10 a.m. and 4 p.m. on May 3, 2001, using backing and flanking fires. Air temperature ranged from 75 °F to 80 °F, relative humidity was about 55 percent, and winds were variable, averaging 2 to 3 miles per hour from the southeast to the northeast. Fuel loading was estimated as 1 to 2 tons per acre each of humus, leaf litter, and logging debris, for a total loading of 3 to 6 tons per acre. Visual fire intensity ranged from small unburned patches to almost complete consumption of large logging debris. Estimated fuel consumption averaged about 85 percent of the preburn leaf litter, half of the litter layer and most of the logging residues.

Field Samples

Data were collected from sample plots established after the area was burned; the schedule for burning did not allow for their preburn installation. We used two methods of sampling to achieve study objectives: permanent plots and temporary transects. Permanent plots were established in early summer to determine (1) estimated density of live yellow-poplar saplings present before burning and (2) fire-related mortality of yellow-poplar saplings and density and composition of regeneration after burning. A uniform grid of 21 circular 0.01-acre plots (11.8 feet radius) was established at 200- by 200-foot intervals throughout the stand for estimation of vegetative composition and stem density. Saplings were inventoried by species in five height classes (<1 foot, 1 to 2 feet, 2 to 3 feet, 3 to 4 feet, and >4 feet). Dead, e.g., nonsprouting, yellow-poplar saplings were also recorded to provide a conservative estimate of the live population of saplings present before burning. The numbers of dead yellow-poplar saplings, particularly those >1 foot tall, could be inventoried accurately because the succulent green stems were not consumed by the fire and species could be determined with certainty from the distinctive bark and stem characteristics. The numbers of dead yellow-poplar saplings and live saplings of all species were summed by height class to obtain a conservative estimate of saplings present after the prescribed burn.

Transects were established through the burned stand in late summer to investigate the second objective: factors associated with the mortality of hardwood saplings. Sample plots 3.28 feet in radius (0.000776 acre) were systematically established along transects that extended through areas of the stand that exhibited the range of fire intensity. Intensity was quantified by measuring the height aboveground of burned (or charred) bark on the stems of saplings (Waldrop and Brose 1999). Stem char was relatively uniform throughout the small plots, and a single value was recorded for each sample site. Data collected from all saplings on each plot included life status (alive or dead, as indicted by the presence or absence of basal sprouts), total height, and species. Correlation analysis was used to determine the association of sapling size with selected stand variables. Logistic regression was used to determine significant ($P < 0.05$) relationships between sapling mortality and the independent variables.

RESULTS

Density and Species Composition

The preburn density of yellow-poplar saplings was estimated as 9,181 stems per acre (table 1). Sapling height was directly correlated with density ($r = 0.62$, $P < 0.01$); the tallest saplings (>4 feet) occurred in dense thickets. Approximately 6,304 yellow-poplar saplings per acre were killed by the prescribed burn, resulting in an average mortality of about 69 percent. The proportion of saplings killed ranged from 27 to 50 percent for the four shortest height classes, but increased to 87 percent for saplings >4 feet. Approximately half of the postburn population of 5,476 saplings per acre consisted of yellow-poplar.

Sixteen arborescent species were recorded on the 21 sample plots following the prescribed burn (table 2). Three species occurred on more than half of the plots: yellow-poplar, sassafras (*Sassafras albidum*), and black locust, which together accounted for 86 percent of the stem density. Yellow-poplar was present in greatest numbers, but black locust was most widespread, occurring on 90 percent of the plots. Except for black locust, most species tended to occur in patches as indicated by the coefficients of variation, which averaged around 200 percent.

Table 1—Preburn density of yellow-poplar saplings and postburn mortality and density by species and height class following a spring prescribed burn on a mesic site in the Southern Appalachian Mountains (*n* = 21 plots)

		Height class (feet)					
Inventory	Species	<1	1–2	2–3	3–4	>4	Total±SD
		trees per acre					
Preburn	Yellow-poplar	343	628	1238	1505	5467	9181±13,042
Mortality	Yellow-poplar	171	171	514	705	4743	6304±7969
Postburn	Yellow-poplar	172	457	724	800	724	2877±5998
	Miscellaneous	690	852	419	295	343	2599±2470
Total		862	1309	1143	1095	1067	5476±5669

SD = standard deviation.

Table 2—Mean stem density and stocking by species of 5-year-old tree regeneration following a prescribed burn on a mesic site in the Southern Appalachian Mountains (*n* = 21 plots)

Species	Density±SD	Density CV	Stocking
	trees per acre	*percent*	
Yellow-poplar (*Liriodendron tulipifera*)	2876±5998	208	62
Sassafras (*Sassafras albidum*)	1104±2040	185	71
Black locust (*Robinia pseudoacacia*)	714±707	99	90
Sweet birch (*Betula lenta*)	219±676	309	24
Red maple (*Acer rubrum*)	152±225	148	48
Blackgum (*Nyssa sylvatica*)	143±340	238	38
Northern red oak (*Quercus rubra*)	105±166	158	10
White oak (*Q. alba*)	67±203	303	19
Sourwood (*Oxydendrum arboreum*)	38±74	195	29
Flowering dogwood (*Cornus florida*)	19±68	358	10
Cucumber-tree (*Magnolia accuminata*)	10±30	300	10
Chestnut oak (*Q. prinus*)	10±30	300	10
Other[a]	19±22	116	5
All species	5476±5669		

SD = standard deviation; CV = coefficient of variation (SD/mean)*100.

[a] Other = one sapling each of American chestnut (*Castanea dentata*), American holly (*Ilex opaca*), black oak (*Quercus velutina*), and hickory (*Carya* spp.).

Effects of Fire Intensity

A total of 197 saplings were inventoried on 20 sample plots established along transects to investigate causes of mortality. The majority of saplings sampled were yellow-poplar (65 percent), followed by red maple, black locust, and northern red oak (each 7 percent). Sample sizes were sufficient (>30) for analysis by species only for yellow-poplar (n = 128); all others were pooled (n = 69). Sapling d.b.h. averaged 0.30 inch (range from 0.1 to 2.0 inches), and total height averaged 6.3 feet (range from 1.6 to 14 feet). Over 99 percent of the inventoried saplings in this data set were topkilled, but mortality (indicated by absence of basal sprouts) was only 34 percent. Field observations suggested that mortality was associated with fire intensity, stem size, and species.

Logistic regression indicated that sapling life status (LS) (topkilled or dead) was a function of species group and stem char height:

$$LS = -4.8906 + 2.6662SpG + 0.3303SCH \qquad (1)$$

where

SpG = species group (miscellaneous, zero; yellow-poplar, one)

SCH = stem char height (feet)

Both variables were highly significant ($P < 0.001$). The model was developed with values of char height ranging mainly from 1.6 to 9.8 feet. Standard errors of the model constant, *SpG* and *SCH* are 0.925, 0.701, and 0.072, respectively. Figure 2 displays results of solving equation (1) for probability of sapling mortality in exponential form: $P_m = \exp^{(LS)}/(1 + \exp^{(LS)})$ with a range of values of stem char height for the two species groups.

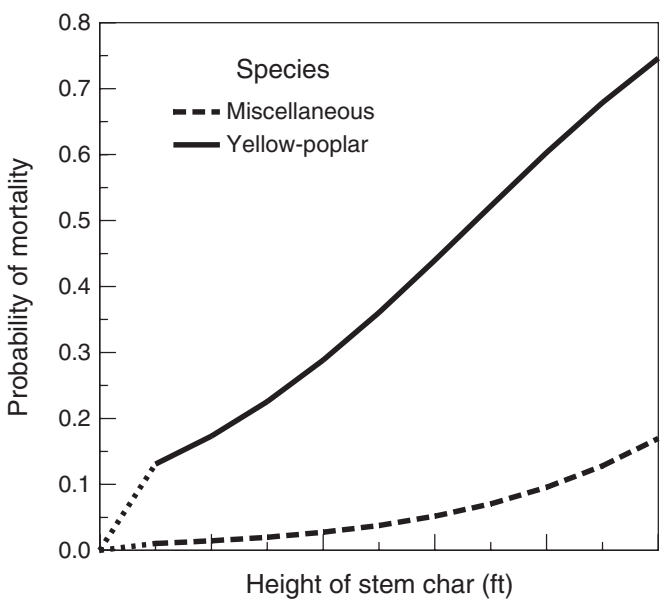

Figure 2—Probability of hardwood sapling mortality in relation to species and height of stem char resulting from a prescribed fire on a mesic site in the Southern Appalachian Mountains.

For example, probability of mortality for a miscellaneous species resulting from stem char height of 3 feet from a prescribed burn on a mesic site 5 years after a regeneration cut is estimated as: $P_m = \exp^{(-3.9)}/(1 + \exp^{(-3.9)}) = 0.02$. Inadequate field data for saplings with char height of zero (unburned) did not allow predictions of the model to pass through the origin.

DISCUSSION

The prescribed burn achieved the desired objective of reducing sapling density, but the level of density remained relatively high—over 5,000 stems per acre. The reduction in stem density is likely to be temporary because we observed newly germinated yellow-poplar seedlings soon after the fire on about half of the permanent plots. The new seedlings likely originated from seeds stored in the litter layer (Shearin and others 1972). Barnes and Van Lear (1998) also reported an increase of yellow-poplar stems following a single spring burn and found that repeated burns were needed to achieve a lasting reduction in density. Keetch (1944) found, however, that neither sprouting capacity nor height growth of sprouts was diminished following three prescribed burns on a dry site. Our results suggest that a series of relatively intense prescribed burns will be required to reduce hardwood stems to a desired density and maintain that density to accomplish the desired wildlife habitat objectives.

Frequent prescribed fires of sufficient intensity to cause additional mortality of yellow-poplar saplings will likely be difficult to achieve on this site. Albrecht and Mattson (1977) found that loading of fuels on mountain cove sites averaged about 4.2 ± 0.5 tons per acre and consisted mostly of leaf litter with little material provided by shrubs. Assuming first that half or less of the average fuel loading reported for coves would be consumed during a typical prescribed burn, and second that total loading on the recently burned study site will be less than occurs on a recently unburned site, then it appears that fire intensity may be marginal to attain adequate sapling mortality. For example, insufficient available fuel was a contributing factor to an unsuccessful second prescribed fire attempted at the study site in early May 2002.

The equation for estimating the probability of sapling mortality provides a quantitative method that managers can use to prescribe a burn of the intensity needed to achieve the desired effects on the sapling stand. The model indicates the probability of mortality of hardwood saplings is directly related to stem char height and species group. Application of the model for planning a prescribed burn to reduce sapling density, for example, indicates that fire intensity resulting in stem char height of about 6.5 feet would be needed to obtain a mortality probability level of 0.5 for a recently regenerated sapling stand on a mesic site. A variable quantifying sapling size was not included in the final regression model because it produced illogical results.

Our results appear to imply that sapling mortality from prescribed burning increases with increasing size, e.g., height (table 1). This relationship is likely an artifact of the dataset that resulted from an interaction between the occurrence

of large saplings and fuel loading. In our study area sapling height was correlated with stand density particularly in thickets of yellow-poplar where trees tended to be tall and spindly (fig. 1). We observed that ground fuels in thickets consisted mostly of fallen deciduous foliage with little or no green herbaceous content, and loading appeared to be higher there than elsewhere. Fire intensity, therefore, was likely greater in thickets than elsewhere in the burned area resulting in greater mortality of tall saplings, where d.b.h. ranged up to 2 inches. Even though the larger saplings presumably had slightly thicker bark than the smaller saplings not in thickets (Hengst and Dawson 1994), the increased thickness likely was not sufficient to insulate the cambium from reaching lethal temperature from the pulse of heat produced by rapid combustion of the fine fuels. Generally, tree mortality from prescribed burns is inversely related to their size (Green and Shilling 1987, Hare 1965, McNab 1977).

In summary, results from our study suggest that prescribed burning to reduce density of hardwood regeneration density on mesic sites can be successful, particularly if recent logging residues are present. But, obtaining the desired mortality may be difficult with a single fire. As other studies have shown, a series of prescribed burns will likely be necessary to prevent domination of the site by aggressive yellow-poplar seedlings, saplings, and sprouts. A prediction model provides managers with a means to plan prescribed fires of specified intensity to achieve the desired level of hardwood sapling mortality.

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LITERATURE CITED

Albrecht, M.H.; Mattson, K.E. 1977. Fuel weights by types for Piedmont and mountain regions of North Carolina. For. Note 30. Raleigh, NC: North Carolina Forest Service, Office of Forest Resources. 19 p.

Barnes, T.A.; Van Lear, D.H. 1998. Prescribed fire effects on advanced regeneration in mixed hardwood stands. Southern Journal of Applied Forestry. 22: 138–142.

Beck, D.E.; Harlow, R.F. 1981. Understory forage production following thinning in Southern Appalachian cove hardwoods. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies. 35: 185–196.

Beck, D.E.; Hooper, R.M. 1986. Development of a Southern Appalachian hardwood stand after clearcutting. Southern Journal of Applied Forestry. 10: 168–172.

Green, T.A.; Shilling, C.L. 1987. Predicting girdling probability for pine and hardwood saplings in low-intensity backfires. Forest Science. 33: 1010–1021.

Hare, R.C. 1965. Contribution of bark to fire resistance of southern trees. Journal of Forestry. 63: 248–251.

Hengst, G.; 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: 686–696.

Keetch, J.J. 1944. Sprout development on once-burned and repeatedly-burned areas in the Southern Appalachians. Tech. Note 59. Asheville, NC: U.S. Department of Agriculture Forest Service, Appalachian Forest Experiment Station. 3 p.

McNab, W.H. 1977. An overcrowded loblolly stand thinned with fire. Southern Journal of Applied Forestry. 1: 24–26.

Shearin, A.T.; Bruner, M.H.; Goebel, N.B. 1972. Prescribed burning stimulates natural regeneration of yellow-poplar. Journal of Forestry. 70: 482–484.

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. 20 p.

Waldrop, T.A; Brose, P.H. 1999. A comparison of fire intensity levels for stand replacement of Table Mountain pine (*Pinus pungens* Lamb.) Forest Ecology and Management. 113: 155–166.

IMPACT OF RAINFALL ON THE MOISTURE CONTENT OF LARGE WOODY FUELS

Helen H. Mohr and Thomas A. Waldrop[1]

Abstract—This unreplicated case study evaluates the impact of rainfall on large woody fuels over time. We know that one rainfall event may decrease the Keetch-Byram Drought Index, but this study shows no real increase in fuel moisture in 1,000-hour fuels after just one rainfall. Several rain events over time are required for the moisture content of large woody fuels to increase thereby impacting fire behavior.

INTRODUCTION

A prolonged drought in the Southeastern United States increased the severity of southern pine beetles (*Dendroctonus frontalis*) throughout the region. Since 2001, South Carolina alone has detected over 92,000 beetle-killed spots and the death of approximately 25.5 million trees. After these attacks as dead trees fall, a particular concern is heavy fuel loading and how to predict when large fuels will ignite. Fire behavior models typically don't account for these fuels, but when ignited, logs can increase fire intensity and/or duration and become a problem for smoke management (Haywood and others 2004).

Land managers are commonly faced with prescribed fires that have higher intensities than expected after prolonged droughts. One reason for this discrepancy is that land managers are using the Keetch-Byram Drought Index (KBDI) alone to estimate fire behavior. The KBDI is part of the National Fire Danger Rating System and is the most widely used drought index for the fire danger rating. The KBDI was developed by John Keetch and George Byram to look at the effects of long-term drying on litter and duff and subsequently, on fire activity. The index ranges from zero to 800 with zero being saturated and 800 the worst drought condition. The index measure is in hundredths of an inch and is based on a measurement of 8 inches of available moisture in the upper soil layers. The available moisture can be used by vegetation for evapotranspiration. The index indicates deficit inches of available water in the soil. A KBDI of 250 means that there is a deficit of 2.5 inches of ground water available to the vegetation (Melton 1989). During a long dry period KBDI could be in the 600+ range and large fuels have experienced deep drying. A single rain event could cause KBDI to drop into the 200 to 300 range. While the 1- and 10-hour fuels would be immediately impacted by the rain event the larger 100- and 1,000-hour fuels would still be extremely dry on the interior. Fire behavior could be much different than expected if the KBDI alone is used as a fire behavior predictor.

In Georgia, the number of fires and acres burned between 1957 and 2000 was highest in the months of February and March. This is when KBDI is typically lowest with August having the highest KBDI. Wildland fire incidents in Georgia corresponded especially to the spring fire season not the high KBDI months of summer (Chan and others 2004). The influence of fuel moisture on fire behavior is not well known for 100- and 1,000-hour fuels. This is particularly a problem in the Southeastern United States where there is extensive southern pine beetle damage. Many southern forests have heavy 100- and 1,000-hour fuel loading due to the southern pine beetle. This study applied six different rainfall events and monitored fuel moisture of 1,000-hour logs to see how moisture contents changed with different rainfall amounts and durations.

METHODS

This study was conducted on the Clemson Experimental Forest in Clemson, SC. We collected two types of logs, several species of the red oak group (Section Lobatae) and loblolly pine (*Pinus taeda* L.). The loblolly pine collection consisted of live and dead tree logs whereas the red oak logs were all collected from live trees. The three log size classes were: (1) 3 to 5 inches, (2) 5 to 7 inches, and (3) 7 to 9 inches in diameter. Each log was 3 feet in length with the cut ends untreated. Trees were cut down and then cut to meet the size classes required. The dead loblolly logs were collected from the forest floor and cut according to size class as well. The dead loblolly logs were left from a southern pine beetle cut and had been lying on the forest floor for approximately 1 year.

The cut logs were placed into a barn where they were allowed to dry well protected from rain. This barn was not temperature controlled. The logs stayed in the barn for a total of 2 years. In the second year fans were used to circulate air across the logs for 24 hours a day to aid in the drying process. Log moistures were regularly checked over 2 years to monitor the drying process. When the logs reached equilibrium they were removed from the barn to the treatment location as needed.

Rainfall treatment events began on April 7, 2008, and ran until complete on June 18, 2008 (table 1). Rainfall was applied

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Table 1—Rainfall treatments and treatment dates

Treatments	Treatment dates 2008
4 inches one time in 1 day	April 7
2 inches two times in 1 week	April 10, 15
1 inch four times in 1 week	April 15, 16, 17, 21
1 inch four times in 2 weeks	April 23, 25, 29
1 inch four times in 3 weeks	May 5, 11, 14, 21
1 inch four times in 4 weeks	May 29; June 3, 12, 18

with a Colorado State rainfall simulator which was placed on an open gravel lot in full sun (fig. 1). The gravel was covered with 3 inches of wood mulch to simulate the forest floor. Natural rainfall was not a factor as we did not have any during our experiment times. The rainfall simulator applied rain at a rate of about 1 inch per hour. Twenty-four logs from each log type (a total 72 logs) were placed on the mulch for each of the 6 rainfall events. The logs were placed on the mulch in a random order for each of the six events. We applied six rainfall events: (1) 4 inches in 1 day, (2) 2 inches two times in 1 week, (3) 1 inch four times in 1 week, (4) 1 inch four times in 2 weeks, (5) 1 inch four times in 3 weeks, and (6) 1 inch four times in 4 weeks.

Log moisture was sampled in two locations on each log prior to and after each rainfall event. The posttreatment log moistures were taken 24 hours following each rainfall event to allow time for the exterior of the logs to dry. We used a Delmhorst® J-Series compact wood moisture meter with a type 26-ES two-pin hammer probe electrode. Fuel moisture measurements were taken 1 foot in from each end of the log on top of the log through the bark at a depth of 1 1/8 inches.

Figure 1—Colorado State rainfall simulator with 72 logs underneath.

Equipment limitations made it impossible to replicate this study. We had only one rainfall simulator and limited logs for this study. This unreplicated case study simply shows mean moisture changes for each rainfall event in all three species and size classes of logs.

RESULTS

Pretreatment log moisture ranged from 11.2 to 23.2 percent with an average of 15.4 percent. Dead pines had the lowest pretreatment moisture contents with the live pines slightly higher. The dead pines had the highest posttreatment moisture contents for the 1 inch four times in 2 weeks and the 1 inch four times in 3 weeks. The oak and live pine log moisture contents stayed close after the first rainfall in all treatments. The dead pine had the highest increase in log moistures. The 4 inches in 1 day treatment showed the dead pines with higher moisture content than the live pines and oaks (fig. 2). The 1 inch four times in 1 week treatment rained on logs every other day for 1 week. For this treatment log moisture contents of all classes increased at a steady rate with the dead pines having the highest moisture contents around 32 percent (fig. 3). The 2 inches two times in 1 week treatment followed the same trend with increased log moistures after each rainfall event (fig. 4). The oaks once again show a slower increase in moisture content than the live and dead pine. One inch of rain four times in 2 weeks produced the wettest logs after the last treatment (fig. 5). The watering for this treatment had more time between rainfall events but not enough time to let the logs dry. All pine logs and the 3- to 5-inch oak logs all had moisture contents greater than 30 percent. Watering the logs 1 inch four times in 3 weeks showed the same steady increase with slightly lower moisture contents than the 1 inch four times in 2 weeks treatment (fig. 6). With all of the treatments there was a steady increase in fuel moisture over time except for the 1 inch four times in 4 weeks treatment (fig. 7). The amount of time between rainfalls may have been too great allowing the logs to dry before the next rainfall could be applied. Our treatments show that one rainfall event only impacts the log moisture content slightly except for the 4 inches one time in 1 day treatment. To reach log moistures in the high 20 to low 30 percent range we had to apply 1 to 2 inches of rain frequently over a 2- to 3-week period. We obtained the greatest moisture contents by applying 1 inch of rainfall four times over 2 weeks. The logs that received 2 inches of rain two times in 1 week also had a higher moisture content of 27.1 percent.

CONCLUSIONS

We know that KBDI alone is not a good indicator of fire behavior and intensity in the Southeast. After a single rainfall event KBDI can drop drastically but moisture contents of large woody fuels do not all increase appreciably. In this preliminary case study we were able to monitor log moisture content over time with different rainfall treatments. This study gives us a picture that one rainfall event will not greatly increase the moisture content of 100- and 1,000-hour fuels unless it is an event producing large amounts of rain. It takes several 1- to 2-inch rainfall events over time. The next step in this

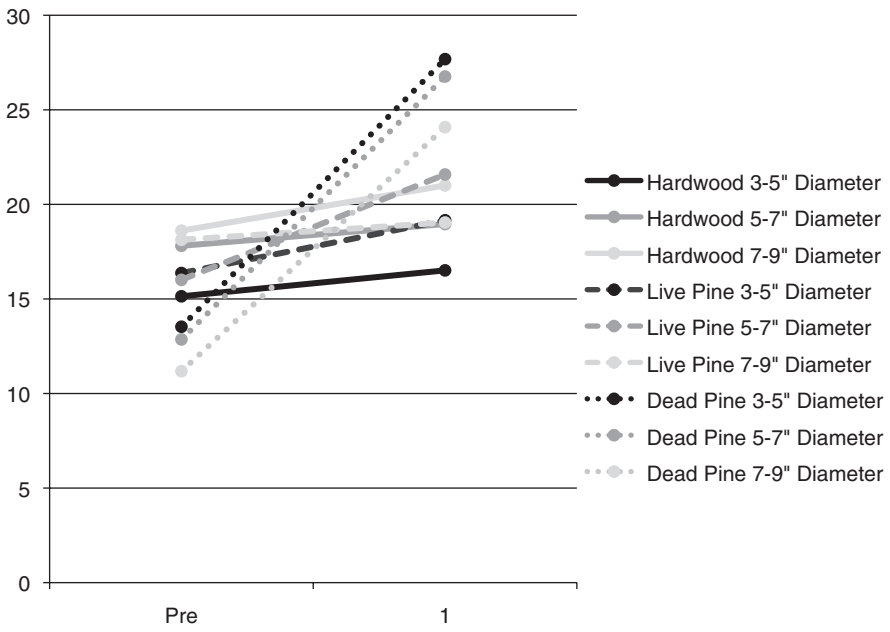

Figure 2—Average log moisture content (percent) prior to treatment (pre) and 24 hours after the rainfall event (1) for the treatment of 4 inches of rain applied one time in 1 day.

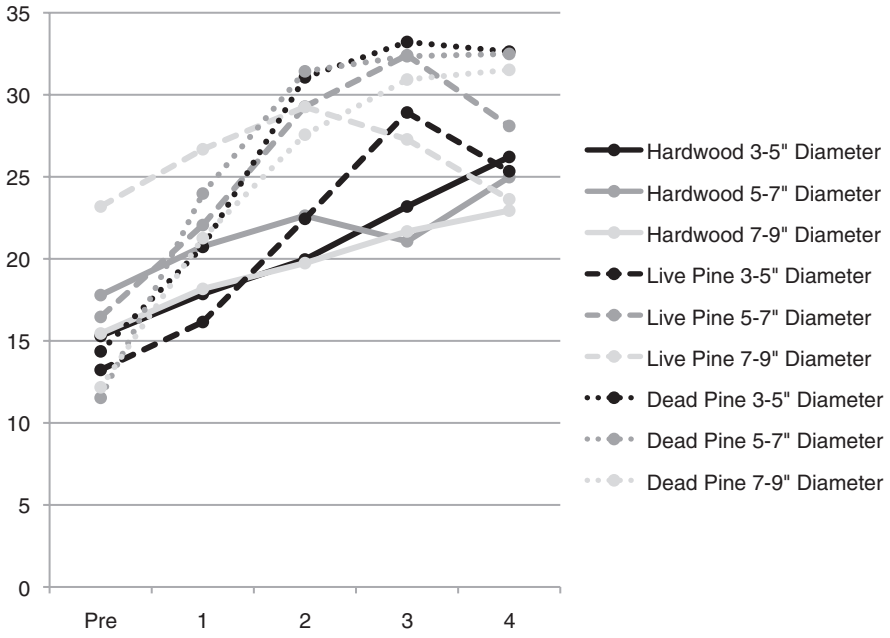

Figure 3—Average log moisture content (percent) prior to treatment (pre) and 24 hours after each rainfall event (1, 2, 3) for the treatment of 1 inch of rain applied four times over a 1-week period.

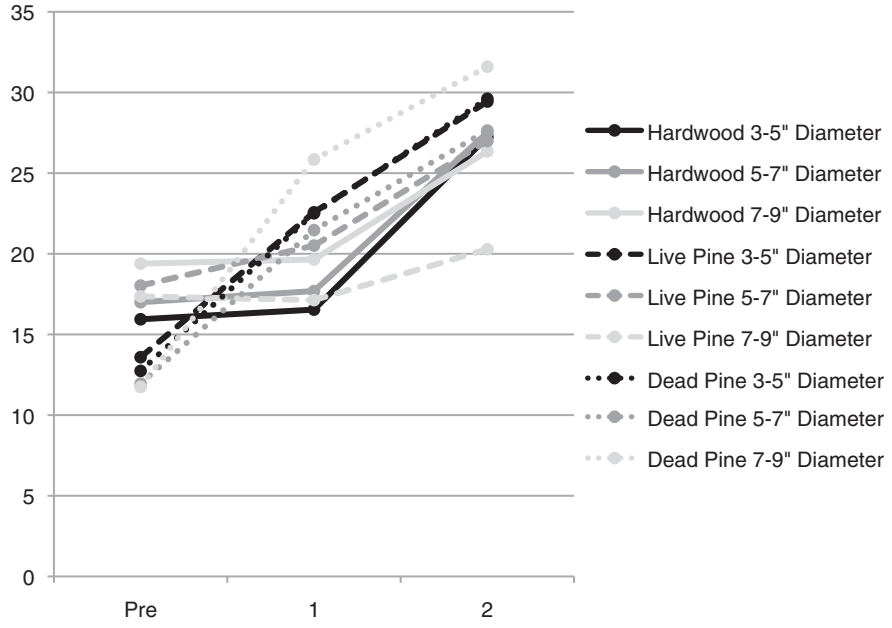

Figure 4—Average log moisture content (percent) prior to treatment (pre) and 24 hours after each rainfall event (1, 2) for the treatment of 2 inches of rain applied two times over 1 week.

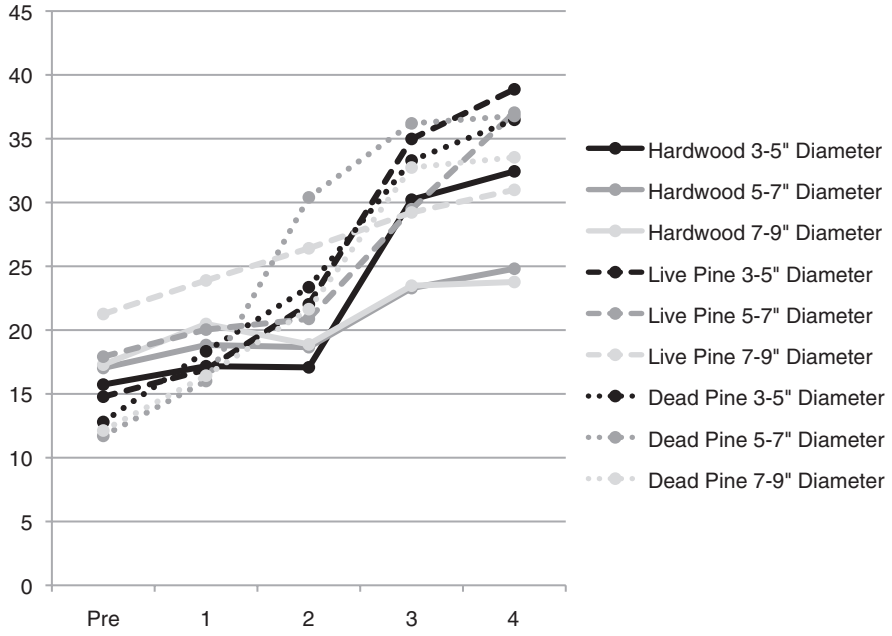

Figure 5—Average log moisture content (percent) prior to treatment (pre) and 24 hours after each rainfall event (1, 2, 3, 4) for the treatment of 1 inch of rain applied four times over 2 weeks.

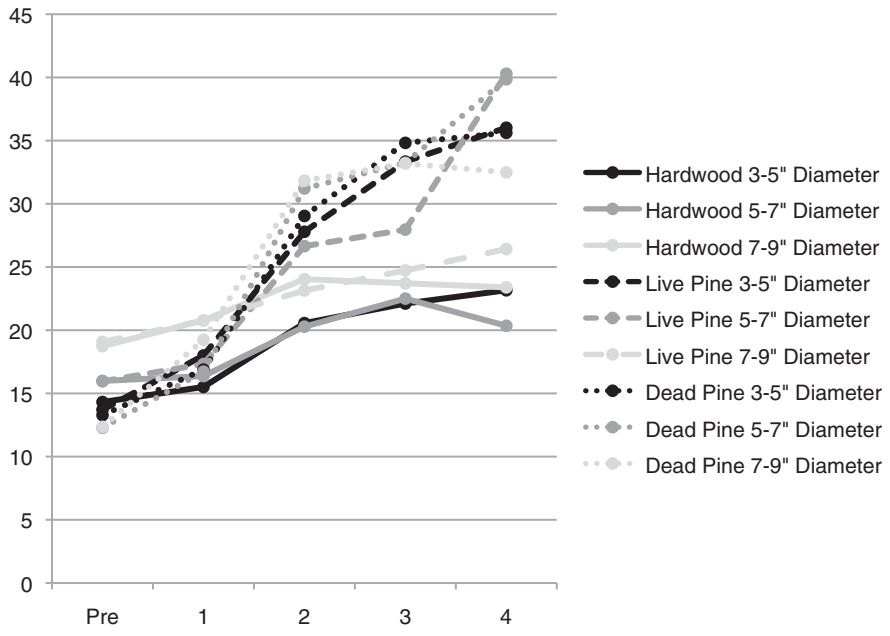

Figure 6—Average log moisture content (percent) prior to treatment (pre) and 24 hours after each rainfall event (1, 2, 3) for the treatment of 1 inch of rain applied four times over 3 weeks.

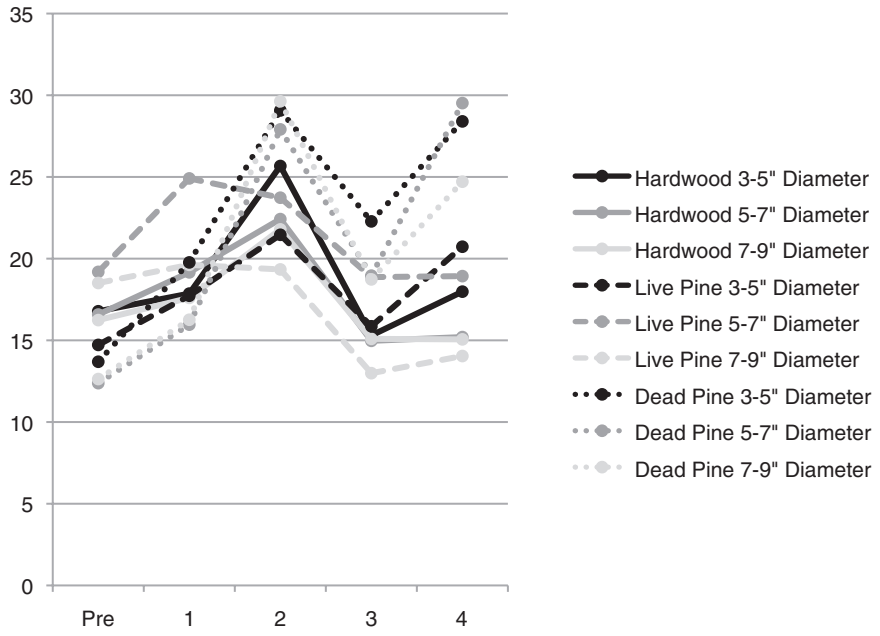

Figure 7—Average log moisture content (percent) prior to treatment (pre) and 24 hours after each rainfall event (1, 2, 3, 4) for the treatment of 1 inch of rain applied four times over 4 weeks.

study would be to replicate the treatments and add more rainfall events. Taking the wet logs to wooded areas that are scheduled for prescribed burns would then be the next step. Burning these logs in forested conditions would allow us to measure consumption due to the fire. The combination of log moisture data and log consumption may give us a clearer picture of fire behavior due to large woody fuels.

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LITERATURE CITED

Chan, David W.; Paul, James T.; Dozier, Alan. 2004. Keetch-Byram drought index: can it help predict wildland fires. Fire Management Notes. 64(2): 39–42.

Haywood, James D.; Stagg, Richard H.; Tiarks, Allen E. 2004. Relationship between Palmer's drought severity index and the moisture index of woody debris in the southern Coastal Plain. In Connor, Kristina 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: 39–43.

Melton, Mike. 1990. Keetch-Byram drought index: a guide to fire conditions and suppression problems. Fire Management Notes. 50(4): 30–34.

OCCURRENCE AND SPREAD OF NONNATIVE INVASIVE PLANTS IN STANDS TREATED WITH FIRE AND/OR MECHANICAL TREATMENTS IN THE UPPER PIEDMONT OF SOUTH CAROLINA

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Abstract—Increasing numbers of nonnative invasive plant species and the expansion of existing nonnative plant populations provide challenges for land managers trying to achieve commercial and restoration goals. Some methods used to achieve these goals, e.g., prescribed fire and mechanical treatments, may result in disturbances that promote the establishment and spread of invasive species. Natural disturbances, e.g., insect infestations, can also provide opportunities for nonnative plant expansion. We examined the effects of fuel-reduction treatments on the occurrence and abundance of nonnative invasive plants for mixed *Pinus taeda*/*P. echinata* stands that had sustained southern pine beetle (*Dendroctonus frontalis*) infestations and those that had not. Invasive plant abundance appeared to be greatest 3 to 5 years after disturbance. For stands not affected by southern pine beetles, the combination of mechanical treatment plus burning resulted in the largest increases for invasive species. Stands suffering pine beetle damage and subjected to mechanical treatment showed higher invasive abundance as compared to other treatments. Some invasive species responded differently to treatments. This information can help direct land management decisions.

INTRODUCTION

Invasions of nonnative plant species have received considerable attention over the last few decades as land managers are faced with increasing issues of exotic plant control, which can affect biodiversity, forest productivity, and disturbance regimes (Gordon 1998, Levine and others 2003, Vitousek 1990). Fuel-reduction treatments intended to reduce fuel loading and restore community composition and structure may be contributing to nonnative plant invasion and expansion (e.g., Metlen and Fiedler 2006); therefore, managers need information on the changes to community structure, environmental variables, and invasive plant dynamics in response to these treatments.

Natural disturbances can also provide opportunities for nonnative plant expansion. Outbreaks of southern pine beetle (*Dendroctonus frontalis* Zimm.) occur periodically in the Southern United States with severe outbreaks causing extensive damage to large areas of pine forests. In South Carolina, southern pine beetle infestations from 2000 until the winter of 2002 affected over 2.2 million ha (U.S. Department of Agriculture Forest Service 2003a) and caused losses over $250 million for 2002 alone (U.S. Department of Agriculture Forest Service 2002). For existing nonnative plant populations in the understory of these affected areas, overstory mortality can lead to expansion of these plant populations as limiting resources become more available.

It has been estimated that economic impacts of invasive species exceed more than $4 billion a year (U.S. Department of Agriculture Forest Service 2003b). With invasive species comprising up to 48 percent of the total flora for some States and the expected increase of nonnative species as globalization continues and climate conditions change (Dukes and Mooney 1999), dealing with these species will continue to be a major issue for land managers.

We examined the effect fuel-reduction treatments had on nonnative invasive plant abundance and how this was influenced by additional natural disturbances, i.e., southern pine beetle infestation. We also looked at the changes in understory species composition over time as it related to differences in stand structure with particular emphasis on nonnative invasive species.

STUDY SITE

The study site is in the South Carolina Piedmont on the Clemson Experimental Forest (34°40' N, 82°49' W) in Anderson, Oconee, and Pickens Counties. The dominant forest type is *Pinus taeda* L. and *P. echinata* Mill. growing over highly degraded soils. Most of the forest is second- or third-growth timber resulting from reforestation programs of abandoned agricultural fields in the early 1900s. The area also had a history of intentional introductions of nonnative plants for erosion control, wildlife forage, and/or horticultural purposes (Sorrells 1984). The climate of the region is characterized by mean monthly temperatures between 5 °C and 26 °C, and mean annual precipitation of 1372 mm distributed evenly throughout the year (National Oceanic and Atmospheric Administration 2002).

METHODS

We used a randomized complete block design with each treatment replicated three times. The treatments for intact pine stands, i.e., no presence of pine beetles at study initiation, included: (1) mechanical treatment by means of a single entry thinning from below (conducted in the winter of 2000 to 2001) with a target basal area of 18 m^2/ha,

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(2) prescribed burning during the spring every 3 years, (3) the combination of mechanical treatment plus burning, and (4) an untreated control. The first prescribed burns were performed in 2001 for the burn-only treatment, whereas the burns for the mechanical+burn treatment occurred the following year. A second round of burns was conducted in 2004 and 2005 for the burn treatment and mechanical+burn treatment, respectively. Pine beetle damage was so extensive in the original burn treatment that a second set of burn treatments was established (designated as "burn1" for the original and "burn2" for the replacements). For the initial burns, we recorded maximum temperatures of 253 °C to 399 °C in the "burn1" treatment; 177 °C to 253 °C for the "burn2" treatment, and 177 °C to 253 °C in the mechanical+burn treatment. Maximum temperatures for the second burn in the mechanical+burn treatment ranged from 204 °C to 816 °C. Details on fire behavior and weather are reported by Phillips and Waldrop (2008). Additional treatment units were selected from stands that sustained southern pine beetle damage, where all overstory pines had been killed over an area of at least 15 ha and that had active pine beetles within 2 years prior to plot establishment. To reduce fuel loading for these stands, we used: (1) a mastication treatment, which removed all dead overstory trees and turned large woody fuel into mulch, (2) low-intensity site prep burns conducted in the late spring of 2006, and (3) high-intensity site prep burns, also in the spring of 2006. An untreated control was established for comparison. Maximum temperatures measured at 1 m above the forest floor for the beetle-killed prescribed burns ranged from 181 °C to 216 °C for the low-intensity burn treatment and from 291 °C to 320 °C for the high-intensity burn treatment.

Understory vegetation (<1.4 m tall) was measured on twenty 1-m^2 subplots nested within 0.1-ha modified Whittaker plots prior to treatment (year 0), immediately following treatment (year 1), and every year thereafter (year 2, year 3, etc.) for up to 7 years. For the intact pine stands we measured ten 0.1-ha plots per treatment area, whereas the size of the beetle-killed stands permitted only two 0.1-ha plots. All understory vegetation was recorded by cover class: 1 = <1 percent; 2 = 1 to 10 percent; 3 = 11 to 25 percent; 4 = 26 to 50 percent; 5 = 51 to 75 percent; and 6 = >75 percent. Each cover class was assigned the value of the class midpoint and averaged across plots for data analysis. We selected a subset of nonnative invasive species to analyze based on maximum abundance, number of occurrences, and threat classification. All species selected are classified as "severe threat" for South Carolina, defined by the SC Exotic Pest Plant Council (2008) as posing significant risk to composition, structure, or function of natural areas. These species included: *Lonicera japonica* Thunb., *Lespedeza bicolor* Turcz., *Microstegium vimineum* (Trin.) A. Camus, *Ligustrum sinense* Lour., *Albizia julibrissin* Durazz., *Ailanthus altissima* (Mill.) Swingle, and *Pueraria montana* (Lour.) Merr. Diameter at breast height (d.b.h.) for all live overstory trees on one-half of the 0.1-ha plot was measured and converted to basal area for each treatment area.

We applied repeated measures analysis of variance to test differences in percent cover of selected nonnative invasive plants and their responses to each fuel-reduction treatment. To account for pretreatment differences, we subtracted pretreatment values from each posttreatment measurement. We made post-hoc comparisons using linear contrasts to test differences between each treatment and interpreted significant treatment and/or treatment by year interactions at $\alpha = 0.05$ as evidence of treatment effects. Data from stands unaffected by pine beetles were analyzed separately from beetle-killed stands.

Nonmetric multidimensional scaling (NMS) was used to examine changes in total understory vegetation composition over time due to treatment effects with particular emphasis on nonnative species. After removing rare species (occurring in <2 percent of sampled plots) from the dataset, we conducted ordinations using the Sørensen distance measure with 250 runs of real data and 250 runs of randomized data in 6 dimensions (McCune and Mefford 2006).

RESULTS AND DISCUSSION

Prior to fuel-reduction treatments, total cover of nonnative invasive species was considerably less in stands not impacted by pine beetles (0.8 percent) as compared to those that had sustained southern pine beetle damage (3.9 percent). Pine beetle infestations caused extremely high mortality for all overstory pines resulting in basal areas of ≤1 m^2/ha, whereas intact stands were characterized by basal areas of 25 m^2/ha or greater (Phillips and Waldrop 2008).

Time since disturbance appeared to be the most important factor affecting species abundance with species responding differently to the treatments (table 1). The greatest increases in nonnative invasive species cover were observed 3 to 5 years after treatment, primarily in the mechanical+burn treatment, as species were able to take advantage of more available resources quickly recovering from treatment disturbances or becoming established following treatment. *L. japonica* was significantly greater in the mechanical+burn treatment than all other treatments (P-values ≤ 0.0313) 3 years after treatment but significantly less than the mechanical treatment (P-value ≤ 0.0001) and control (P-value = 0.0008) 5 years after treatment. *L. sinense* also showed significant increases in the mechanical+burn treatment as compared to the "burn1" treatment (P-value = 0.0084), mechanical treatment (P-value = 0.0065), and control (P-value = 0.0087) after 3 years, but not the "burn2" treatment (P-value = 0.0823). *M. vimineum* was only recorded in the mechanical+burn treatment and the control with significant differences observed for the sampling period 3 years after treatment (P-value = 0.0074). Burning appeared to encourage growth and/or establishment for *L. bicolor*, *L. sinense*, and *A. julibrissin*, whereas *L. japonica* demonstrated a decrease in cover in the burn treatments but showed a slight increase in abundance after initial mechanical+burn treatment. However, successive burns for this treatment reduced cover to below pretreatment values. In contrast, the mechanical treatment resulted in continued growth of *L. japonica* as it was able to take advantage of more available light.

Table 1—Mean invasive plant cover (and standard error) for intact pine stands treated with fuel-reduction techniques in the upper Piedmont of South Carolina

Years since treatment[a]	Treatment	LONIJAP	LESPBIC	MICRVIM	LIGUSIN	ALBIJUL	AILAALT	PUERMON
0	Control	0.47 (0.16)	0.00 (0.00)	0.01 (0.01)	0.09 (0.05)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Mechanical	0.65 (0.22)	0.00 (0.00)	0.00 (0.00)	0.05 (0.03)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Burn 1	0.16 (0.11)	0.00 (0.00)	0.00 (0.00)	0.01 (0.00)	0.00 (0.00)	0.01 (0.01)	0.00 (0.00)
	Burn 2	1.18 (0.43)	0.00 (0.00)	0.00 (0.00)	0.02 (0.01)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Mechanical+burn	0.72 (0.24)	0.00 (0.00)	0.19 (0.19)	0.41 (0.30)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
1	Control	0.62 (0.20) a[b]	0.00 (0.00)	0.00 (0.00)	0.01 (0.01)	0.00 (0.00) b	0.00 (0.00) c	0.00 (0.00)
	Mechanical	0.37 (0.10) ab	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00) b	0.00 (0.00) c	0.00 (0.00)
	Burn 1	0.04 (0.02) ab	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00) b	0.22 (0.06) a	0.00 (0.00)
	Burn 2	0.36 (0.14) b	0.10 (0.10)	0.00 (0.00)	0.16 (0.10)	0.15 (0.09) ab	0.00 (0.00) c	0.00 (0.00)
	Mechanical+burn	0.38 (0.15) ab	0.53 (0.29)	0.31 (0.31)	0.24 (0.17)	0.35 (0.21) a	0.14 (0.07) b	0.00 (0.00)
3	Control	0.66 (0.16) b	0.00 (0.00) b	0.00 (0.00) b	0.03 (0.01) b	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Mechanical	0.84 (0.28) b	0.00 (0.00) b	0.00 (0.00) b	0.02 (0.01) b	0.00 (0.00)	0.01 (0.00)	0.00 (0.00)
	Burn 1	0.13 (0.04) b	0.00 (0.00) b	0.00 (0.00) b	0.00 (0.00) b	0.01 (0.01)	0.05 (0.04)	0.00 (0.00)
	Burn 2	0.44 (0.15) b	0.07 (0.06) b	0.00 (0.00) b	0.26 (0.18) ab	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Mechanical+burn	1.77 (0.67) a	1.16 (0.84) a	1.84 (1.84) a	1.13 (0.78) a	0.26 (0.22)	0.00 (0.00)	0.00 (0.00)
5	Control	2.43 (0.56) a	0.00 (0.00) b	0.13 (0.13)	0.08 (0.05) b	0.00 (0.00) b	0.00 (0.00)	0.00 (0.00)
	Mechanical	2.99 (0.94) a	0.00 (0.00) b	0.00 (0.00)	0.08 (0.05) b	0.00 (0.00) b	0.00 (0.00)	0.00 (0.00)
	Burn 1	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Burn 2	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Mechanical+burn	1.12 (0.30) b	5.73 (2.91) a	0.97 (0.97)	1.48 (0.84) a	0.51 (0.31) a	0.01 (0.00)	0.00 (0.00)
7	Control	0.45 (0.10)	0.00 (0.00)	0.00 (0.00)	0.02 (0.01)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Mechanical	1.13 (0.39)	0.00 (0.00)	0.00 (0.00)	0.05 (0.03)	0.00 (0.00)	0.03 (0.03)	0.00 (0.00)
	Burn 1	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Burn 2	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Mechanical+burn	0.59 (0.14)	0.02 (0.01)	0.10 (0.10)	0.50 (0.30)	0.25 (0.15)	0.00 (0.00)	0.00 (0.00)

Species codes: LONIJAP = *Lonicera japonica*; LESPBIC = *Lespedeza bicolor*; MICRVIM = *Microstegium vimineum*, LIGUSIN = *Ligustrum sinense*; ALBIJUL = *Albizia julibrissin*; AILAALT = *Ailanthus altissima*; PUERMON = *Pueraria Montana*.

n/a = data not available.

[a] Corresponds to pretreatment sampling (0); first year immediately after initial treatment (1); 3 years after initial treatment (3), etc.

[b] Within sample year, values followed by different letters indicate significant change in percent cover from pretreatment level at $\alpha = 0.05$.

Our results are consistent with other studies showing greater abundance of nonnative invasive plants in areas subjected to mechanical treatment combined with prescribed burning (Dodson and Fiedler 2006, Griffis and others 2001). Large reductions in overstory basal area and disturbance to the forest floor provided suitable seedbed habitat for nonnative species to germinate and expand. Implications from these findings suggest fuel-reduction treatments could potentially have effects opposite from intended purposes, e.g., ecosystem restoration, by negatively impacting natural species regeneration (Oswalt and others 2007) and altering disturbance regimes (Brooks and others 2004).

Results from the fuel-reduction treatments in beetle-killed stands (table 2) cover only the first 2 years since disturbance; therefore, conclusions are preliminary. Initial reductions in invasive cover did not persist over time as relatively few significant differences between pretreatment and 2-year

Table 2—Mean invasive plant cover (and standard error) in southern pine beetle-killed pine stands treated with fuel-reduction techniques in the upper Piedmont of South Carolina

Years since treatment[a]	Treatment	LONIJAP	LESPBIC	MICRVIM	LIGUSIN	ALBIJUL	AILAALT	PUERMON
0	Control	4.74 (1.66)	0.00 (0.00)	0.00 (0.00)	0.01 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Mastication	8.07 (4.36)	0.00 (0.00)	0.00 (0.00)	0.15 (0.15)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Low burn	2.32 (1.60)	0.00 (0.00)	0.00 (0.00)	0.16 (0.16)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	High burn	0.20 (0.14)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
1	Control	7.36 (2.54) a[b]	0.00 (0.00)	0.00 (0.00)	0.06 (0.03)	0.13 (0.10)	0.00 (0.00)	0.00 (0.00) b
	Mastication	1.35 (0.74) b	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00) b
	Low burn	1.53 (1.26) a	0.00 (0.00)	0.00 (0.00)	0.15 (0.10)	0.00 (0.00)	0.00 (0.00)	0.11 (0.11) a
	High burn	0.25 (0.12) a	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00) b
2	Control	4.48 (1.48)	0.00 (0.00)	0.00 (0.00)	0.10 (0.06) a	0.05 (0.05)	0.00 (0.00)	0.00 (0.00)
	Mastication	7.57 (3.68)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00) b	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Low burn	1.63 (1.02)	0.00 (0.00)	0.00 (0.00)	0.06 (0.05) ab	0.05 (0.05)	0.00 (0.00)	0.00 (0.00)
	High burn	0.22 (0.15)	0.05 (0.05)	0.00 (0.00)	0.00 (0.00) ab	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)

Species codes: LONIJAP = *Lonicera japonica*; LESPBIC = *Lespedeza bicolor*; MICRVIM = *Microstegium vimineum*, LIGUSIN = *Ligustrum sinense*; ALBIJUL = *Albizia julibrissin*; AILAALT = *Ailanthus altissima*; PUERMON = *Pueraria montana*.

[a] Corresponds to pretreatment sampling (0); first year immediately after initial treatment (1); 3 years after initial treatment (3), etc.

[b] Within sample year, values followed by different letters indicate significant change in percent cover from pretreatment level at $\alpha = 0.05$.

posttreatment values were evident. *L. japonica* was the most abundant species with other species periodically recorded during the study period. Surrette and Brewer (2008) demonstrated that high cover of *L. japonica* was associated with areas of high disturbance, low fire frequency, and high soil compaction, which are characteristics of both the mechanical and mastication treatments. For these stands, *L. japonica* is likely to out-compete other species preventing stand development unless fire, or another means for limiting its growth (herbicides or manual removal), is incorporated. Burning successfully reduced *L. sinense*; however, *P. montana* and *L. bicolor* became established following treatment. Continued burning will likely increase the abundance of *L. bicolor* (Tesky 1992) as well as *P. montana* (Munger 2002) unless accompanied by herbicide application (Miller 2003).

Distinct differences in community composition between intact pine stands and beetle-killed pine stands were evident prior to treatment, but these differences appeared to diminish over time, except for the control and mastication treatments for the beetle-killed stands (fig. 1). The final NMS ordination converged on three axes with a final stress of 7.4. The amount of variance explained by each axis was 36.3 percent for axis 1, 29.0 percent for axis 2, and 28.9 percent for axis 3 (cumulative R^2 = 94.2 percent). Over time plots generally moved from left to right across axis 1, which may be an increasing gradient of soil moisture. Data on soil moisture were not available, but we speculate there were differences between stands that were not affected by beetles and those which had beetle infestations. These changes in stand structure and disturbance to the forest floor may have affected soil moisture thus influencing species composition. Axis 3 appeared to be associated with light availability as demonstrated by the positive correlation to basal area (r = 0.829) and the movement of plots down this axis for intact pine stands (increased light reaching the forest floor) contrasted by the progression of beetle-killed plots up this axis.

With respect to invasive species, *A. julibrissin* and *L. sinense* were positively associated with axis 1, whereas *M. vimineum* showed positive correlation with axis 3 (table 3). *L. japonica* was correlated with both axis 1 (r = 0.829) and axis 3 (r = –0.535). Even though *L. japonica* can persist under low-light conditions, it is more prolific in high-light environments as demonstrated by a strong negative correlation with axis 3. In contrast, *M. vimineum*, which is well adapted to low-light conditions and prefers disturbed sites that are shaded and more mesic (Barden 1987), had a positive association with this axis.

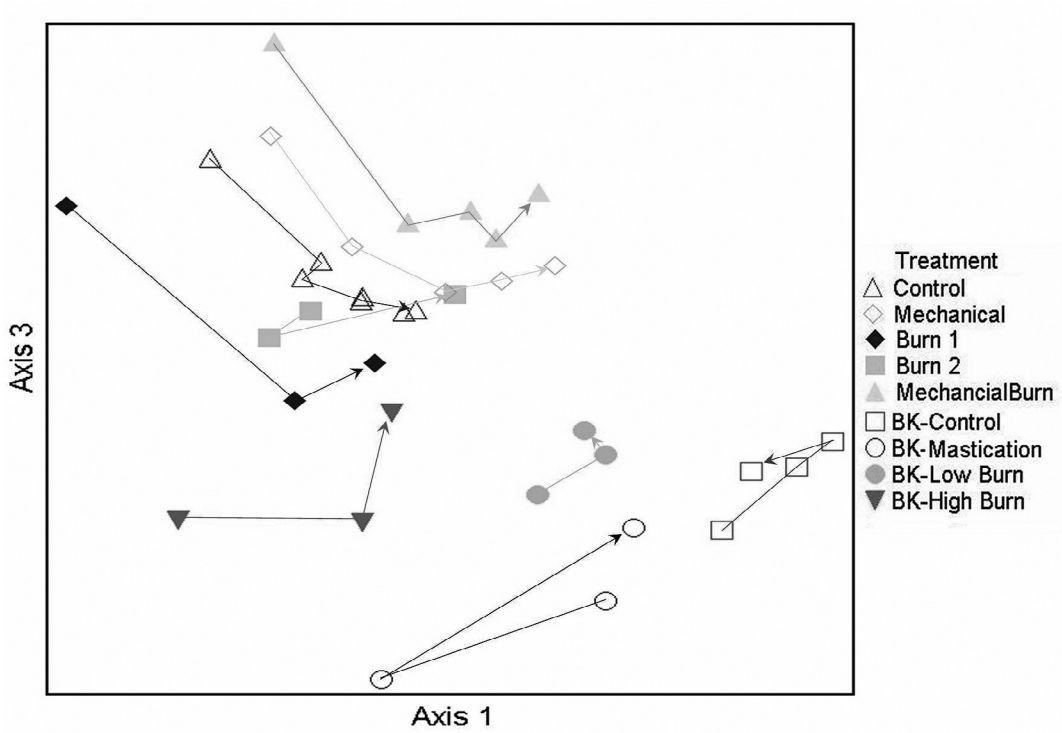

Figure 1—Nonmetric multidimensional scaling (NMS) ordination of plots in species space depicting change over time for pine stands treated with fuel-reduction treatments in the upper Piedmont of South Carolina. Axis 1 appeared to be related to soil moisture (decreasing from left to right) and axis 3 was associated with light availability (increasing from top to bottom).

Table 3—Parametric (Pearson's *r*) and nonparametric (Kendall's tau) correlations of invasive species with NMS axes for pine stands in the upper Piedmont of South Carolina

	Axis 1		Axis 2		Axis 3	
Species	*r*	tau	*r*	tau	*r*	tau
AILAALT	–0.190	–0.129	0.020	0.097	0.119	0.209
ALBIJUL	0.430	0.340	–0.406	–0.413	0.069	–0.038
LESPBIC	0.035	0.048	–0.317	–0.389	0.232	0.240
LIGUSIN	0.423	0.304	–0.210	–0.178	0.276	0.175
LONIJAP	0.829	0.613	–0.080	–0.060	–0.535	–0.263
MICRVIM	0.030	–0.096	–0.181	0.029	0.353	0.435
PUERMON	0.187	0.216	–0.356	–0.293	–0.148	–0.149

Species codes: LONIJAP = *Lonicera japonica*; LESPBIC = *Lespedeza bicolor*; MICRVIM = *Microstegium vimineum*, LIGUSIN = *Ligustrum sinense*; ALBIJUL = *Albizia julibrissin*; AILAALT = *Ailanthus altissima*; PUERMON = *Pueraria montana*.

NMS = nonmetric multidimensional scaling.

Abundances of some of these nonnative species are relatively small and statistically significant differences may not represent biological differences, but their presence and responses to these treatments indicate that land managers need to consider these species when planning hazard-reduction or restoration treatments. The establishment of nonnative species following treatment, even at low levels, needs to be addressed in a timely manner as cost for eradication increases over time (Rejmánek and Pitcairn 2002). Invasion by nonnative species can cause a variety of changes in the ecosystem, from altered disturbance regimes (Mack and D'Antonio 1998) to changes in soil nutrient cycling and decomposition (Ehrenfeld 2003).

CONCLUSIONS

Based on understory vegetation composition, it appears that pine stands treated with prescribed fire and mechanical fuel-reduction techniques and those sustaining southern pine beetle damage are more or less converging over time; however, the effects of these treatments on nonnative invasive species differed. Burning decreased *L. japonica* cover; mechanical+burning stimulated growth and/or establishment for *L. bicolor*, *L. sinense*, and *A. julibrissin*; whereas the mechanical treatments, both thinning and mastication, resulted in the greatest abundance of *L. japonica*. Depending on the presence of invasive species prior to fuel-reduction treatment, land managers can decide which treatment is best suited for preventing invasive expansion while accomplishing the goal of hazard fuel reduction.

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LITERATURE CITED

Barden, L.S. 1987. Invasion of *Microstegium vimineum* (Poaceae), an exotic, annual, shade-tolerant C4 grass, into a North Carolina floodplain. American Midland Naturalist. 118: 40–45.

Brooks, M.L.; D'Antonio, C.M.; Richardson, D.M. [and others]. 2004. Effects of invasive alien plants on fire regimes. BioScience. 54: 677–688.

Dodson, E.K.; Fiedler, C.E. 2006. Impacts of restoration treatments on alien plant invasion in *Pinus ponderosa* forests, Montana, USA. Journal of Applied Ecology. 43: 887–897.

Dukes, J.S.; Mooney, H.A. 1999. Does global change increase the success of biological invaders? Trends in Ecology and Evolution. 14: 135–139.

Ehrenfeld, J.G. 2003. Effects of exotic plant invasions on soil nutrient cycling processes. Ecosystems. 6: 503–523.

Gordon, D.R. 1998. Effects of invasive, non-indigenous plant species on ecosystem processes: lessons from Florida. Ecological Applications. 8: 975–989.

Griffis, K.L.; Crawford, J.A.; Wagner, M.R.; Moir, W.H. 2001. Understory response to management treatments in northern Arizona ponderosa pine forests. Forest Ecology and Management. 146: 239–245.

Levine, J.M.; Vilà, M.; D'Antonio, C.M [and others]. 2003. Mechanisms underlying the impacts of exotic plant invasions. Proceedings of the Royal Society of London Series B. 270: 775–781.

Mack, M.C.; D'Antonio, C.M. 1998. Impacts of biological invasions on disturbance regimes. Trends in Ecology and Evolution. 13: 195–198.

McCune, B.; Mefford, M.J. 2006. PC-ORD. Multivariate analysis of ecological data. Version 5.14. Gleneden Beach, OR: MjM Software.

Metlen, K.L.; Fiedler, C.E. 2006. Restoration treatment effects on the understory of ponderosa pine/Douglas-fir forests in western Montana, USA. Forest Ecology and Management. 222: 355–369.

Miller, J.H. 2003. Nonnative invasive plants of southern forests: a field guide for identification and control. Gen. Tech. Rep. SRS-62. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 93 p.

Munger, G.T. 2002. *Pueraria montana* var. *lobata*. Fire effects information system [online]. [Fort Collins, CO]: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/. [Date accessed: January 8, 2009].

National Oceanic and Atmospheric Administration. 2002. Climatography of the United States series no. 81: monthly station normals of temperature, precipitation, and heating and cooling degree days, 1971–2000 (South Carolina–38). Asheville, NC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Climatic Data Center. 23 p.

Oswalt, C.M.; Oswalt, S.J.; Clatterbuck, W.K. Effect of Migrostregium vimineum (Trin.) A Camus on native woody species density and diversity in a productive mixed-hardwood forest in Tennessee. Forest Ecology and Management. 242: 727–732.

Phillips, R.J.; Waldrop, T.A. 2008. Changes in vegetation structure and composition to fuel reduction treatments in the South Carolina Piedmont. Forest Ecology and Management. 255: 3107–3116.

Rejmánek, M.; Pitcairn, M.J. 2002. When is eradication of exotic pest plants a realistic goal? In: Veitch, C.R.; Clout, M.N., eds. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. Gland, Switzerland and Cambridge, UK: IUCN: 249–253.

SC Exotic Pest Plant Council. 2008. Invasive plant pest species of South Carolina. http://www.se-eppc.org/southcarolina/scinvasives.pdf. [Date accessed: October 10, 2008].

Sorrells, R.T. 1984. The Clemson Experimental Forest: its first fifty years. Clemson, SC: Clemson University, College of Forest and Recreation Resources. 52 p.

Surrette, S.B.; Brewer, J.S. 2008. Inferring relationships between native plant diversity and *Lonicera japonica* in upland forests in north Mississippi, USA. Applied Vegetation Science. 11: 205–214.

Tesky, J.L. 1992. *Lespedeza bicolor*. In: Fire effects information system [online]. [Fort Collins, CO]: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/. [Date accessed: January 08, 2009].

U.S. Department of Agriculture Forest Service. 2002. 2002 forest insect and disease conditions for the Southern Region. Washington, DC. http://www.fs.fed.us/r8/foresthealth/2002conditions/index.html. [Date accessed: October 10, 2008].

U.S. Department of Agriculture Forest Service. 2003a. Forest insects and disease conditions in the United States 2002. Washington, DC. 124 p.

U.S. Department of Agriculture Forest Service. 2003b. USDA Forest Service strategic plan for fiscal years 2004–08. FS-810. Washington, DC. 32 p.

Vitousek, P.M. 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. Oikos. 57: 7–13.

FUEL LOADING FOLLOWING FUEL-REDUCTION TREATMENTS AND IMPACTS FROM NATURAL DISTURBANCES

Ross J. Phillips and Thomas A. Waldrop[1]

Abstract—A long-term study of fuel-reduction treatments (mechanical fuel removal, prescribed burning, and the combination of mechanical treatment and burning) was begun in 2000 and 2001 for sites located in the Piedmont of South Carolina and the Southern Appalachian Mountains of North Carolina, respectively. During this time multiple natural disturbances [southern pine beetle (*Dendroctonus frontalis*) infestations and ice storms] occurred that allowed us to observe effects that fuel-reduction treatments had on impacts from these disturbances at these two sites. After 8 years and multiple natural disturbances, the mechanical treatment at the Piedmont site showed little difference in fine fuel and 1,000-hour fuel loadings from the control, whereas the mechanical+burn treatment had significantly less fuel. For the Appalachian site, an ice storm event in 2005 resulted in large inputs of fine fuels in the mechanical treatment and control units. Two years later, fine fuel loadings in those treatments were still significantly higher than that measured in the burn and mechanical+burn treatments; however, units treated with prescribed fire had greater 1,000-hour fuel loadings. Predicted fire behavior following fuel-reduction treatments, ice storms, and/or pine beetle infestations was lowest for the mechanical treatment at the Piedmont site and for the burn and mechanical+burn treatments at the Appalachian site. Changing fuel loadings through fuel-reduction techniques can have important effects on fire behavior by altering fuel structure and may influence the impact of natural disturbances in treated stands.

INTRODUCTION

With frequent occurrences of southern pine beetle (SPB) (*Dendroctonus frontalis*) infestations (8 to 10 years) and ice storms (5 to 20 years) (Abell 1934, Travis and Meentemeyer 1991) for the southeastern Piedmont and Southern Appalachian regions, these types of disturbances can significantly impact forest composition, structure, and fuel loads. Previous work has suggested these disturbances impact forest succession and species composition (Boerner and others 1988, Lafon 2006, Lafon and Kutac 2003) with relatively few studies focused on fuel loadings (Waldrop and others 2007). Given current forest conditions of southeastern forests—increased fuel loadings as a result of fire suppression over the past century—forest managers have incorporated fuel-reduction techniques to reduce the risk of severe fire occurrence and decrease stand density. However, common natural disturbances, e.g., SPB infestations and ice storms, can significantly alter available fuel, vertical fuel structure, and fuel dynamics, but the degree to which some of these disturbances influence fuel loadings may be affected by forest management practices.

Beginning in 2000, the Southeast experienced a significant SPB outbreak which impacted States from Alabama to Virginia. A SPB epidemic that occurred from 2000 to 2002 devastated over 6 million ha in South Carolina, resulting in over $250 million in losses for 2002 alone (U.S. Department of Agriculture Forest Service 2003). In addition to this large-scale disturbance, several small-scale ice storms occurred in these areas from 2000 to 2008. Two major ice events affected the Piedmont of South Carolina (Dec. 4–5, 2002, and Jan. 26–30, 2004), each producing >2.5 cm of frozen precipitation. The 2004 ice storm resulted in >$95 million in timber losses for over 930 000 ha in South Carolina prompting a Presidential declaration of major disaster (South Carolina Forestry Commission 2004). The Southern Appalachian site experienced one major ice event (Dec.14–15, 2005) with ice accumulations of 0.5 cm to >2.0 cm, the largest accretion occurred around Hendersonville, Saluda, and Tryon, NC (National Weather Service 2005).

The objectives for this paper were to identify differences among four different fuel-reduction treatments with respect to fuel loadings and subsequent fire behavior in the context of real world natural disturbances. How did these treatments influence fine fuel loadings? What impacts did SPB and ice storms have on large, 1,000-hour fuels in stands subjected to fuel-reduction treatments? How did changes in fuel structure and composition affect predicted fire behavior for these stands?

STUDY AREA

The study sites are located in the South Carolina Piedmont on the Clemson Experimental Forest (34°40' N, 82°49' W) in Anderson, Oconee, and Pickens Counties and in the Southern Appalachian Mountains of North Carolina on the Green River Game Land (35°17' N, 82°19' W) in Polk County (fig. 1). The Clemson site is dominated by loblolly pine (*Pinus taeda*) and shortleaf pine (*P. echinata*) growing over highly degraded soils. Most of the forest is second- or third-growth timber resulting from reforestation programs in the early 1900s with stand ages ranging from 15 to 120 years. The climate of the region is warm continental with mean monthly temperatures between 5 °C to 26 °C and mean annual precipitation of 1372 mm distributed evenly throughout the year (National Oceanic and Atmospheric Administration 2002b).

The Green River site supports a variety of oaks (*Quercus* spp.) and hickories (*Carya* spp.), red maple (*Acer rubrum*), yellow-poplar (*Liriodendron tulipifera*), sourwood

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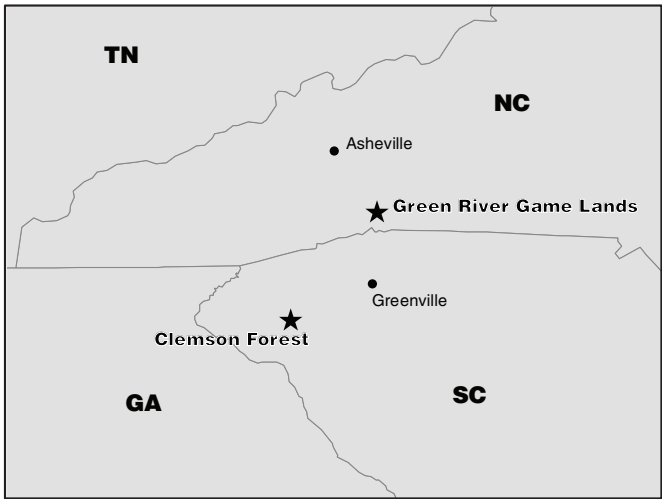

Figure 1—Location for the Piedmont and Southern Appalachian study sites.

(*Oxydendrum arboreum*), pitch pine (*P. rigida*) and a dense layer of ericaceous shrubs—mountain laurel (*Kalmia latifolia*) and rosebay rhododendron (*Rhododendron maximum*), which act as vertical fuels, potentially causing wildfires to reach the tree canopy (Waldrop and Brose 1999). The forests of the study area were 80 to 120 years old, and no indication of past agriculture or recent fire was present, though the historical fire return interval in the area prior to 1940 was approximately 10 years (Harmon 1982). Mean monthly temperatures range between 2 °C and 23 °C; mean annual precipitation averages 1438 mm distributed evenly throughout the year (National Oceanic and Atmospheric Administration 2002a).

METHODS

We used a randomized complete block design with each of 4 treatments present in 3 blocks (12 treatment units per study site). Initial fuel-reduction treatments (mechanical removal of fuel, prescribed burning, a combination of mechanical removal and burning, and a control) were applied from 2000 to 2002 at Clemson and 2001 to 2003 at Green River. We used different techniques for the mechanical treatment for the two sites to address the issues of fuel accumulation. At the Clemson site, we reduced stand density to a residual basal area of 18 m^2/ha using a single entry thinning from below during the winter of 2000 to 2001. Slash created from this treatment was distributed throughout the site. To reduce the vertical buildup of fuels at Green River, contract chainsaw crews felled all tree stems >1.8 m tall and <10.2 cm diameter at breast height (d.b.h.), as well as all shrub stems (predominantly mountain laurel and rhododendron), regardless of size, during the winter of 2001 to 2002. Prescribed burning was conducted on a 3-year cycle with the burn treatment units receiving initial treatment in April 2001 and March 2003 for Clemson and Green River, respectively. The mechanical+burn treatment units were initially burned in 2002 at Clemson and 2003 at Green River. Due to high overstory mortality from SPB in the burn treatment at Clemson, a second set of burn treatment areas was established in 2003 (designated as burn1 for the original and burn2 for the replacements). A second burn was conducted at Clemson in March to April 2004 (burn treatment) and March to May 2005 (mechanical+burn treatment) and at Green River in February to March 2006 (burn and mechanical+burn). Details on prescribed fire behavior are given by Phillips and Waldrop (2008) for the Clemson site and Waldrop and others (2008) for the Green River site.

Approximately 120 fuel transects were established in each treatment unit and were measured using the planar intercept method (Brown 1974) every year or every other year depending on treatment schedule. Fuels were classified by size class: 1-hour fuels (0 to 0.6 cm), 10-hour fuels (0.6 to 2.5 cm), 100-hour fuels (2.5 to 7.6 cm) and 1,000-hour fuels (7.7+ cm). Along the transect, 1- and 10-hour fuel intercepts were counted along the first 2 m and 100-hour fuels were counted along the first 4 m. Fuels in the 1,000-hour class were recorded by species, diameter, and decay class (sound or rotten) along the entire transect (15.2 m). Litter and duff depths were measured at three points along each transect. Fuel counts were converted to mean weights per treatment area with equations given by Brown (1974).

We used repeated measures analysis of variance (SAS Institute Inc. 2002) to identify differences in fine fuel loadings (litter, 1-, 10-, and 100-hour fuels) and large fuel loadings (1,000-hour fuels) and made post-hoc comparisons using linear contrasts for each site separately. We interpreted significant treatment and/or treatment × year interactions ($\alpha = 0.05$) as evidence of treatment effects. As much of these data did not meet the assumption of normality, it was necessary to use data transformations to normalize the distributions. Logarithmic and square root transformations were used in these analyses; however, all reported means were calculated using the nontransformed data.

Custom fuel models were developed for each treatment and fire behavior predictions (based on 80th-percentile weather during the wildfire season for each study site) were made using BehavePlus 4.0 (Andrews and others 2008). Eightieth-percentile weather conditions from February to early April calculated from observations from the Greenville/Spartanburg airport (approximately 72 km from the Clemson study site) included a high temperature of 22 °C, low relative humidity of 34 percent, and peak 5-minute wind speed of 13 m/second. For the Green River study site, values calculated from observations at the Asheville Regional airport (approximately 25 km from the study site) included a high temperature of 13 °C, minimum relative humidity of 42 percent, and peak 5-minute wind speed of 9.4 m/second. We used fuel moisture scenarios representative of conditions in these regions given the above described weather parameters: 1-hour fuel moisture content was 6 percent; 10-hour moisture content was 7 percent; and 100-hour moisture content was 8 percent. BehavePlus 4.0 provided estimates of flame length, rate of spread, spread distance, area burned, and scorch height.

RESULTS AND DISCUSSION

Southeastern Piedmont

Immediately following fuel-reduction treatments at the Clemson site, fine fuel loadings (litter, 1-, 10-, and 100-hour fuels) decreased across all treatments (fig. 2A). The decrease in fine fuels for the mechanical treatment is misleading because 1-, 10-, and 100-hour fuels actually increased after treatment (0.12 Mg/ha, 0.46 Mg/ha, and 2.02 Mg/ha, respectively), but litter decreased 2.92 Mg/ha—a result of less input and the manipulation of the existing forest floor (Waldrop and others 2004). Eight years after treatment and multiple natural disturbances, fine woody fuels were greatest in the mechanical treatment (27.8 Mg/ha) and the control (26.4 Mg/ha) with the mechanical+burn treatment containing significantly less fuel (22.0 Mg/ha) than all other treatments (P-values ≤ 0.019).

Large woody fuels (1,000-hour fuels) increased immediately following mechanical treatment (mechanical and mechanical+burn), whereas the burn treatment decreased and the control showed little change (fig. 2B). Large increases in 1,000-hour fuels were observed 2 to 3 years

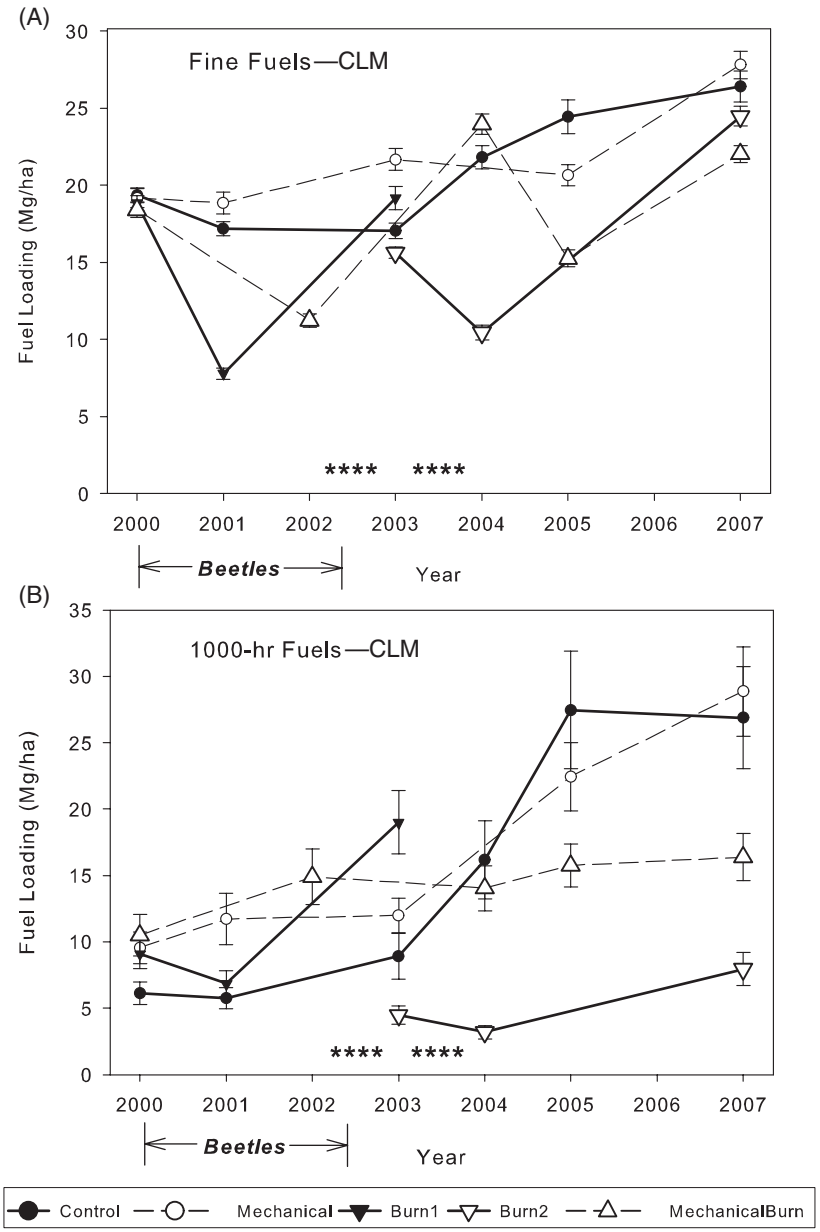

Figure 2—Loadings (Mg/ha) for (A) fine fuels (litter, 1-hour, 10-hour, and 100-hour) and (B) 1,000-hour fuel in stands affected by fuel reduction treatments and natural disturbances in the Piedmont of South Carolina. Southern pine beetle infestations lasted from September 2000 to October 2002. Two ice storm events (****) affected these stands in December 2002 and January 2004. Fuel reduction treatments were applied during 2000 to 2002 and 2004 to 2005.

after SPB infestations for the mechanical treatment and the control resulting in significantly more 1,000-hour fuels in these treatment areas as compared to the other treatments (*P*-values ≤ 0.0010). By year 8, large fuel loadings for all treatments were (in decreasing order) mechanical (28.9 t/ha), control (26.9 t/ha), mechanical+burn (16.4 t/ha), and burn2 (7.9 t/ha). However, the burn2 treatment had only been subjected to a single prescribed fire and was 3 years behind schedule compared to the other treatments.

Over the duration of this study, fuels increased approximately 100 percent in the mechanical and control, roughly 60 percent in burn2, almost 40 percent for burn1, and 33 percent for mechanical+burn treatments which could have significant consequences on fire behavior. Based on fuel models created for the Clemson site, potential fire behavior 8 years after initial treatment was lowest for the mechanical treatment (table 1) even though the mechanical+burn treatment had significantly lower fuel loadings for all fine fuels. These results indicate the sensitivity of BehavePlus to fuel depth as this variable was the only factor lower in the mechanical (12.4 cm) vs. the mechanical+burn (15.4 cm) for that sample period. The burn and mechanical+ burn treatments showed reduced potential fire behavior the first few years following prescribed burning but increased time between burn intervals results in rapid fuel accumulation for these forests (Wade and others 2000).

The SPB infestations affected stands with higher basal areas, i.e., mechanical treatment and control, with little damage observed in the mechanical+burn treatment (Phillips and Waldrop 2008). Effects from the SPB would have lessened the amount of fine fuel input typically expected from ice storms since needles from affected trees had already dropped, leaving less surface area for ice deposition. Shepard (1978) showed that a thinning from below, similar to the mechanical treatment used for this study, would reduce susceptibility of loblolly pine stands to damage from ice storms by removing high-risk trees and encouraging vigorous growth, but timing of the thinning with respect to ice storm occurrence is an important factor. The mechanical treatment applied for this study occurred 2 years prior to the first ice event; therefore, the remaining trees should have had sufficient time to recover and the stands should have been less vulnerable to ice damage (Bragg and others 2003). However, differences between effects from the beetles, ice storms, and/or mortality from treatment on fuel input could

Table 1—Fire behavior predictions (BehavePlus 4.0) for stands affected by southern pine beetles, ice storm damage, and fuel-reduction treatments in the Piedmont of South Carolina

Sample year	Treatment	Rate of spread	Flame length	Spread distance	Area	Scorch height
		km/ha	---------- *m*	----------	*ha*	*m*
2000	C	1.1	2.4	9.0	890.8	2.4
	M	2.1	3.8	17.1	2778.9	7.9
	B	1.0	2.2	7.8	691.9	1.8
	MB	2.5	4.3	20.1	3847.8	11.0
2001	C	0.7	1.8	6.0	452.8	1.2
	M	1.8	3.2	14.0	1875.9	5.2
	B	0.2	0.8	1.7	58.0	0.3
	MB	N/A	N/A	N/A	N/A	N/A
2005	C	3.0	5.0	23.6	5324.9	15.9
	M	2.4	4.1	19.2	3516.3	9.8
	B	N/A	N/A	N/A	N/A	N/A
	MB	0.6	1.6	4.9	338.5	0.9
2007	C	2.7	4.6	21.2	4264.3	12.8
	M	2.0	3.5	15.6	2327.1	6.7
	B	2.5	4.4	19.9	3740.4	11.6
	MB	2.9	4.8	23.1	5084.1	14.3

C = control; M = mechanical; B = burn; MB = mechanical+burn; N/A = not applicable.

Note: Model parameters included 80th-percentile weather conditions for the typical fire season (February to April) and a fire duration period of 8 hours.

not be determined based on the fuel measurements recorded. But examining the interactions between multiple disturbances and understanding their relative influences on fuel loadings, rather than focusing on a single event, (e.g., Lundquist 2007) could provide valuable information for land managers, e.g., fire risk assessment and hazard fuel reduction.

Southern Appalachian Mountains

Following initial treatment at the Green River site, fuel-reduction treatments increased fine fuel loadings in the mechanical treatment, but burning decreased fine fuel loadings by almost half (fig. 3A), most of which included removal of leaf litter—the primary fuel in surface fires for these forests. Damage from the 2005 ice event caused significant increases in fine fuel loadings for the mechanical treatment and control (P-values ≤ 0.0001), which were significantly greater than the burn and mechanical+burn treatments (P-values ≤ 0.0001). After 8 years, the mechanical treatment had the greatest fine fuel loadings (23.0 t/ha), whereas the burn (18.8 Mg/ha) and mechanical+burn (18.6 t/ha) treatments had significantly less fuel (P-values ≤ 0.0068). However, the burned areas experienced high overstory mortality following the 2003 prescribed burns (Waldrop and others 2008), which led to increased 1,000-hour fuel loadings

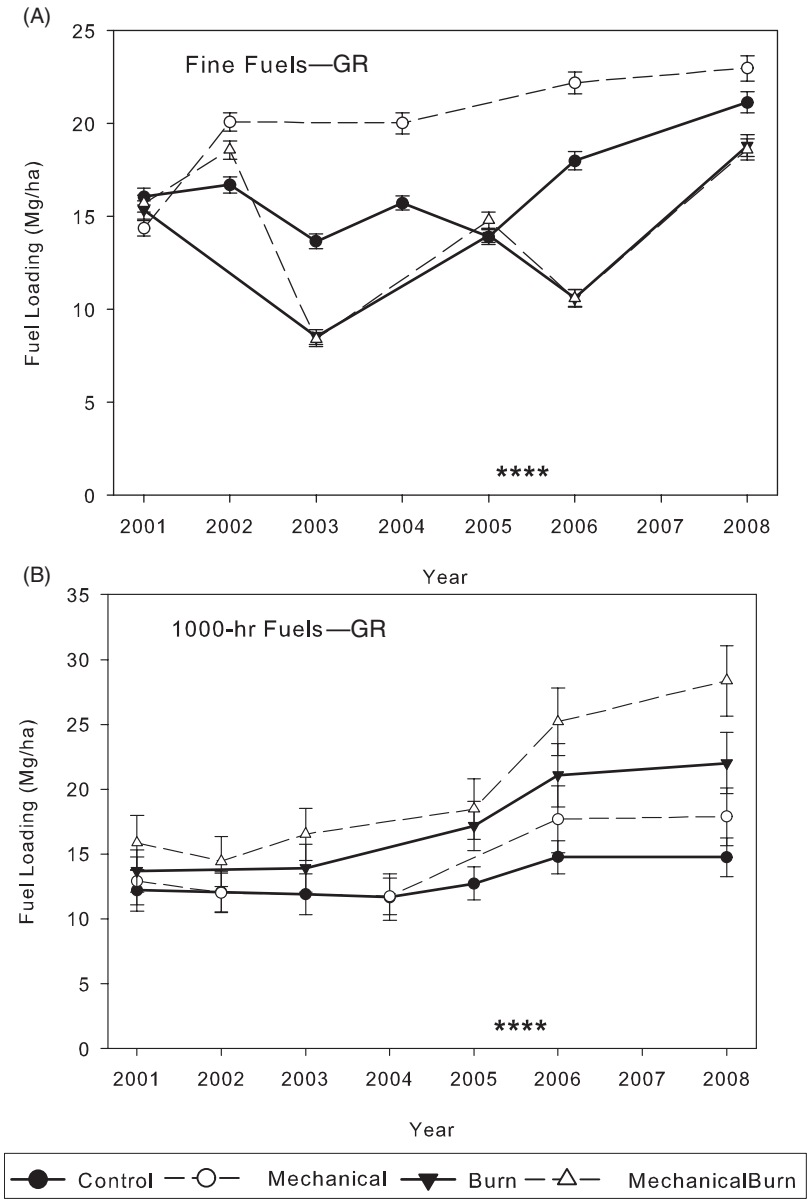

Figure 3—Loadings (Mg/ha) for (A) fine fuels (litter, 1-hour, 10-hour, and 100-hour) and (B) 1,000-hour fuel in stands affected by fuel reduction treatments and natural disturbances in the Southern Appalachians of North Carolina. An ice storm event (****) affected these stands in December 2005. Fuel reduction treatments were applied during 2001 to 2003 and 2006.

2 to 5 years after treatment (fig. 3B). This delayed mortality coincided with the ice storm causing substantial increases in large woody fuels. All treated areas and the control showed significant increases in 1,000-hour fuels following the ice event (*P*-values ≤ 0.0058) with large fuel loadings continuing to increase 2 years later for the mechanical+burn treatment. After 8 years, the mechanical+burn treatment had significantly greater 1,000-hour fuels than the mechanical treatment (*P*-value = 0.0002) and the control (*P*-value ≤ 0.0001). But it was not significantly different from the burn treatment (*P*-value = 0.1427).

Fuel-reduction treatments were intended to remove the vertical fuel component of these stands, primarily ericaceous shrubs, which should considerably affect fire behavior. Predicted fire behavior after 8 years was lower for all treated areas as compared to the control, although differences between the mechanical treatment and the control were small (table 2). Relatively few differences between the burn and mechanical+burn treatment were evident, but considering the presence of ericaceous shrubs (Waldrop and others 2008) and fewer 1,000-hour fuels, we would expect increased fire behavior within the burn treatment with respect to that in the mechanical+burn treatment.

Previous studies have reported inputs from ice damage of 5.1 m^3/ha for old-growth oak-hickory forests in Missouri (Rebertus and others 1997); 19.4 m^3/ha for mesic forests in Wisconsin (Bruderle and Stearns 1985); and 33.6 m^3/ha for old-growth hardwood forests in Quebec (Hooper and others 2001). For this study, the average volume of biomass input following ice damage was approximately 13 m^3/ha for woody fuels, which falls on the lower end of this range. However, this additional amount of fuel can appreciably affect fire behavior as mentioned above and should be accounted for when considering hazard fuel reduction. While the mechanical treatment successfully reduced the vertical fuels, it probably had no effect on ice deposition and resulting damage within the stands. Burning reduced overstory basal area which could influence future impacts from storm damage. The ecological impacts of these disturbances and their interactions with other factors are not well understood. Couple that lack of knowledge with changing land management practices, in addition to climate change, and it is evident that continued

Table 2—Fire behavior predictions (BehavePlus 4.0) for stands affected by ice storm damage and fuel-reduction treatments in the Southern Appalachians of North Carolina

Sample year	Treatment	Rate of spread	Flame length	Spread distance	Area	Scorch height
		km/ha	---------- *m*	----------	*ha*	*m*
2001	C	1.1	2.5	8.8	958.2	3.7
	M	1.2	2.7	9.6	1151.5	4.3
	B	1.1	2.6	9.2	1047.8	4.0
	MB	1.2	2.7	9.5	1124.1	4.3
2003	C	0.9	2.2	7.5	701.6	2.4
	M	N/A	N/A	N/A	N/A	N/A
	B	0.1	0.5	0.7	13.0	0.0
	MB	0.1	0.4	0.5	8.4	0.0
2006	C	3.2	5.5	25.2	7898.6	23.5
	M	3.0	5.5	24.1	7213.3	24.1
	B	0.7	1.6	6.0	548.2	0.9
	MB	0.2	0.7	1.6	56.8	0.3
2008	C	1.9	4.0	15.2	2850.6	11.0
	M	1.8	3.8	14.3	2519.1	10.1
	B	1.4	3.0	10.9	1459.7	5.5
	MB	1.4	3.0	11.4	1598.7	5.8

C = control; M = mechanical; B = burn; MB = mechanical+burn; N/A = not applicable.

Note: Model parameters included 80th-percentile weather conditions for the typical fire season (February to April) and a fire duration period of 8 hours.

research is necessary to provide complete recommendations for land management.

CONCLUSIONS

Application of fuel-reduction treatments, including mechanical removal of fuel, prescribed fire, and the combination of mechanical removal and fire, appear to affect impacts from natural disturbances on fuel loadings in the Piedmont and Southern Appalachians. The combination of multiple disturbances at the Piedmont site made it difficult to separate impacts, but the stands experiencing SPB (control, mechanical, and burn1 treatments) had significantly greater fuel loadings. In the Southern Appalachians, fuel loadings within areas that were burned on a 3-year cycle contained less fine fuels but greater amounts of large fuels than other treatments. Surprisingly at the Piedmont site, predicted fire behavior was actually greatest for stands with the least amount of fine fuel loadings (mechanical+burn), demonstrating the sensitivity of BehavePlus to relatively small changes in fuel height. Altering stand structure and fuel complexes can have significant impacts on a variety of ecosystem components; therefore, different treatments may be appropriate depending on management objectives.

ACKNOWLEDGMENTS

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LITERATURE CITED

Abell, C.A. 1934. Influence of glaze storms upon hardwood forests in the Southern Appalachians. Journal of Forestry. 32: 35–37.

Andrews, P.L.; Bevins, C.D.; Seli, R.C. 2008. BehavePlus fire modeling system. Version 4.0: user's guide. Gen. Tech. Rep. RMRS-GTR-106WWW Revised. Ogden, UT: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 116 p.

Boerner, R.E.J.; Runge, S.D.; Cho, D.; Kooser, J.G. 1988. Localized ice storm damage in an Appalachian plateau watershed. American Midland Naturalist. 119: 199–208.

Bragg, D.C.; Shelton, M.G.; Zeide, B. 2003. Impacts and management implications of ice storms on forests in the southern United States. Forest Ecology and Management. 186: 99–123.

Brown, J.K. 1974. Handbook for inventorying downed woody material. Gen. Tech. Rep. INT-16, Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 24 p.

Bruederle, L.P.; Stearns, F.W. 1985. Ice storm damage to a southern Wisconsin mesic forest. Bulletin of the Torrey Botanical Club. 112: 167–175.

Harmon, M.E. 1982. Fire history of the western most portion of the Great Smoky Mountains National Park. Bulletin of the Torrey Botanical Club. 109: 74–79.

Hooper, M.C.; Arii, K.; Lechowicz, M.J. 2001. Impact of a major ice storm on an old-growth hardwood forest. Canadian Journal of Botany. 79: 70–75.

Lafon, C.W. 2006. Forest disturbance by ice storms in *Quercus* forests of the Southern Appalachian Mountains, USA. Ecoscience. 13: 30–43.

Lafon, C.W.; Kutac, M.J. 2003. Effects of ice storms, southern pine beetle infestation, and fire on Table Mountain pine forests of southwestern Virginia. Physical Geography. 24: 502–519.

Lundquist, J.E. 2007. The relative influence of diseases and other small-scale disturbances on fuel loading in the Black Hills. Plant Disease. 91: 147–152.

National Oceanic and Atmospheric Administration. 2002a. Climatography of the United States series no. 81: monthly station normals of temperature, precipitation, and heating and cooling degree days, 1971–2000 (North Carolina–31). Asheville, NC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Climatic Data Center. 33 p.

National Oceanic and Atmospheric Administration. 2002b. Climatography of the United States series no. 81: monthly station normals of temperature, precipitation, and heating and cooling degree days, 1971–2000 (South Carolina–38). Asheville, NC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Climatic Data Center. 23 p.

National Weather Service. 2005. The ice storm of 15 December 2005 in the western Carolinas and extreme northeast Georgia. Greenville-Spartanburg, SC: National Weather Service Office. http://www.erh.noaa.gov/gsp/localdat/cases/15Dec2005IceStorm/15Dec2005IceStorm.html. [Date accessed: October 10, 2008].

Phillips, R.J.; Waldrop, T.A. 2008. Changes in vegetation structure and composition to fuel reduction treatments in the South Carolina Piedmont. Forest Ecology and Management. 255: 3107–3116.

Rebertus, A.J.; Shifley, S.R.; Richards, R.H.; Roovers, L.M. 1997. Ice storm damage to an old-growth forest in Missouri. American Midland Naturalist. 137: 48–61.

SAS Institute Inc. 2002. SAS user's guide: statistics. Version 9.00. Cary, NC: SAS Institute Inc.

Shepard, R.K., Jr. 1978. Ice storm damage to thinned loblolly pine plantation in northern Louisiana. Southern Journal of Applied Forestry. 2: 83–85.

South Carolina Forestry Commission. 2004. Ice storm: forest losses top $95 million. SCFC news release February 18, 2004. http://www.state.sc.us/forest/b021804.htm. [Date accessed: October 10, 2008].

Travis, D.J.; Meentemeyer, V. 1991. Influence of glaze ice storms on growth rates of loblolly pine *Pinus taeda* and shortleaf pine *Pinus echinata* in the Southern Appalachian Piedmont. Climate Research. 1: 199–205.

U.S. Department of Agriculture Forest Service. 2003. Forest insects and disease conditions in the United States 2002. Washington, DC. 124 p.

Wade, D.D.; Brock, B.L.; Brose, P.H. [and others]. 2000. Fire in eastern ecosystems. In: Brown, J.K.; Smith, J.K., eds. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42. Ogden, UT: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 53–96. Vol. 2.

Waldrop, T.A.; Brose, P.H. 1999. A comparison of fire intensity levels for stand replacement of Table Mountain pine (*Pinus pungens* Lamb.). Forest Ecology and Management. 113: 155–166.

Waldrop, T.A.; Brudnak, L.; Rideout-Hanzak, S. 2007. Fuels on disturbed and undisturbed site in the Southern Appalachian Mountains, USA. Canadian Journal of Forest Research. 37: 1134–1141.

Waldrop, T.A.; Glass, D.W.; Rideout, S. [and others]. 2004. An evaluation of fuel-reduction treatments across a landscape gradient in the Piedmont forests: preliminary results of the National Fire and Fire Surrogate study. In: Conner K.F., ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 54–59.

Waldrop, T.A.; Yaussy, D.A.; Phillips, R.J. [and others]. 2008. Fuel reduction treatments affect stand structure of hardwood forests in western North Carolina and southern Ohio, USA. Forest Ecology and Management. 255: 3117–3129.

NOT ALL BASAL AREA IS CREATED EQUAL: THE INFLUENCE OF SPECIES AND STAND DEVELOPMENT ON CANOPY COVER OF FOUR COMMON SOUTHERN PINES

David G. Ray[1]

Abstract—Restoring natural fire regimes and diverse ground cover to planted or old-field origin southern pine stands typically requires a substantial reduction in overstory density. While maintaining full canopy cover (CC) is consistent with maximizing fiber production, this approach does not allow sufficient light to reach the forest floor to accomplish a broader set of objectives. The relationship between stand basal area (BA) and CC has been used to regulate the overwood in shelterwood seed cuttings and is worth exploring for other purposes. Two factors potentially complicating the use of BA as a proxy for CC are: (1) the dynamic relationship between stem diameter and crown size and (2) species-level differences. Data collected as part of a regional inventory (Forest Inventory and Analysis Forest Health Monitoring plots) were used to construct regression models of crown projection area (CPA) for the four most common southern pines (*Pinus taeda*, PITA; *P. elliottii*, PIEL; *P. echinata*, PIEC; *P. palustris*, PIPA). Species, stem diameter, and live-crown ratio were all identified as important predictor variables ($P < 0.05$). Relative to a stated objective of achieving 50 percent CC there were some substantial differences in the amount of BA suggested for retention in stands of different average diameter (5- to 18-inch d.b.h, PITA 46 to 70 square feet per acre; PIEL 46 to 64; PIEC 47 to 71; PIPA 50 to 74); the differences among species were somewhat less dramatic. There was a clear tendency for relatively small stems, e.g., <10 inch d.b.h., to have larger crowns per unit stem diameter, implying a given CC could be achieved with less BA. Species shade tolerance also appeared to influence crown-stem allometry, with the more shade-tolerant species tending to provide a given amount of CC with less BA. Both factors, species, and average tree size appear worthy of consideration when attempting to meet residual canopy cover goals.

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Not all Basal Area is Created Equal: The Influence of Species and Stand Development on Canopy Cover of Four Common Southern Pines

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BACKGROUND

•Densely stocked pine plantations lack many of the attributes associated with fully functioning upland pine ecosystems (Mitchell et al. 2007).

•There is growing interest in converting existing plantations to a more natural state (Masters et al. 2007, Stanturf et al. 2004).

•Frequent low intensity fires are an integral component of upland-pine ecosystems in the Southeast (Komarek 1974).

•Using structure provided by an existing overstory, even if a less preferred species, may be preferable to starting over from scratch (Kirkman et al. 2007).

•Reducing overstory stocking (canopy cover) to levels that allow the development of diverse ground cover is an initial step in the restoration process.

•Knowledge of differences in canopy cover related to species and stage of development can be used to customize restoration treatments.

Table 1. Characteristics of the FHM sample, means and standard errors.

Species	Common	Code	Count (n)		Dbh (in)		LCR (%)		MCD (ft)	
			Nat	Plant	Nat	Plant	Nat	Plant	Nat	Plant
P. taeda	loblolly	PITA	1,640	3,379	7.7 (0.2)	6.6 (0.2)	37 (1.2)	41 (1.5)	11.6 (0.4)	10.4 (0.3)
P. palustris	longleaf	PIPA	194	13	8.6 (0.3)	7.8 (0.1)	39 (1.1)	58 (0.9)	12.3 (0.4)	12.7 (0.2)
P. echinata	shortleaf	PIEC	568	32	7.4 (0.2)	7.6 (0.2)	33 (1.3)	36 (1.1)	10.2 (0.4)	9.8 (0.5)
P. elliottii	slash	PIEL	284	473	7.4 (0.2)	6.5 (0.2)	28 (1.2)	39 (1.0)	11.1 (0.4)	10.4 (0.3)

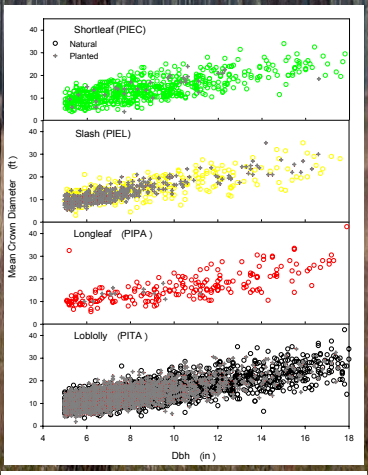

Fig 1. Relationship between stem diameter and mean crown diameter by species and stand origin.

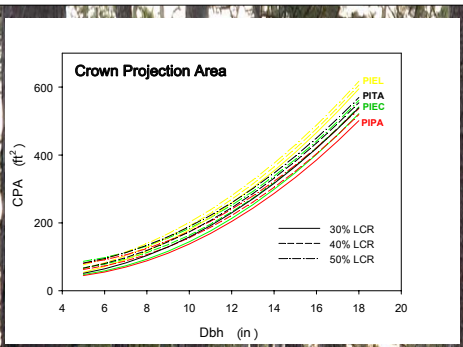

Fig 2. Species-specific crown projection area models of the form: CPA = Dbh + Dbh² + LCR.

METHODS

•Data from the FIA Forest Health Monitoring Program was used to develop species-specific models of crown projection area for four widely distributed southern pines (Bechtold 2003) (Fig 2).

•Analysis of covariance was used to test for differences in crown-stem allometry related to stand origin and species, with live crown ratio (LCR) as a covariate.

•The basal area (BA, ft²/ac) associated with a given level of canopy cover was determined by scaling individual tree crown projection area (CPA) up to the stand level.

Table 2. Regression parameters and standard errors for CPA models.

Species Code	n	b1 (Dbh)	b2 (Dbh2)	b3 (LCR)	R^2
PITA	5,019	-8.809 (0.547)	2.014 (0.043)	1.505 (0.048)	0.56
PIPA	207	-12.215 (3.823)	2.057 (0.243)	1.808 (0.378)	0.56
PIEC	600	-12.938 (1.59)	2.148 (0.121)	1.959 (0.181)	0.58
PIEL	757	-6.163 (1.437)	2.075 (0.112)	1.11 (0.145)	0.60

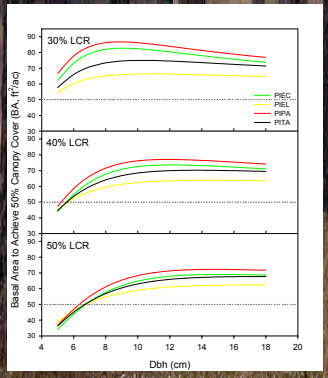

Fig 3. Estimates of stand level BA required to provide 50% CC.

KEY FINDINGS

•Crown-stem allometry was independent of stand origin (P=0.750) yet differed among species (P=0.039) (Fig 1).

•Live crown ratios were positively related to crown projection areas, i.e. trees with relatively longer crowns tended to have larger CPAs' at a given DBH (Fig 2).

•CPA was positively linked to species shade tolerance, where very shade-intolerant PITA had relatively smaller crowns than shade intolerant PIEC and PITA, which were somewhat smaller than those of shade-intermediate PIEL (Fig 2).This finding is consistent with field measurements of canopy gap fraction indicating PIPA provides less canopy cover per unit BA than PIEL (Kirkman et al. 2007).

•The amount of residual basal area required to provide 50% canopy cover increased with stem diameter, perhaps up to 10-in Dbh, indicating younger/smaller trees have larger CPAs per unit BA than older/larger ones (Fig 3).

Fig 4. Juvenile (top) and mature (bottom) pine stands containing 60 ft²/ac of BA.

CONCLUSION

•The relatively large differences in the amount of stand BA to provide 50% CC revealed in this study (34-87 ft²/ac) indicates species, tree size, and LCR, are worthy of consideration when formulating restoration prescriptions involving these southern pines. Perhaps most notably, that relatively lower BAs' should be retained in younger, i.e. smaller average diameter stands, and in particular those with lower LCRs'.

REFERENCES

Bechtold, W.A. 2003. Crown-diameter prediction models for 87species of stand-grown trees in the Eastern US. South. J. Appl. For. 27(4): 269-278.

Kirkman, L.K., et al. 2007. The perpetual forest: using undesirable species to bridge restoration. J. Appl. Ecol. 44: 604-614.

Komarek, E.V. 1974. Effects of fire on temperate forests and related ecosystems: Southeastern United States. In: Lozlowski, T.T., Ahlgren, C.E., Eds. Fire and Ecosystems. Academic Press, NY: 251-277.

Masters, R.E., et al. 2007. Red Hills Forest Stewardship Guide. Tall Timbers Research Station & Land Conservancy. 78 p.

Mitchell, et al. 2007. Silviculture that sustains: the nexus between silviculture, frequent prescribed fire, and conservation of biodiversity in longleaf pine forests of the Southeastern U.S. Can. J. For. Res., Vol. 36: 2724-2736.

Stanturf, J.A. et al. 2004. Restoration of Southern Ecosystems. In: Southern Forest Science: Past, Present and Future. Ch. 13: 123-131.

THE HEALTH OF LOBLOLLY PINE STANDS AT FORT BENNING, GA

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Abstract—Approximately two-thirds of the red-cockaded woodpecker (*Picoides borealis*) (RCW) groups at Fort Benning, GA, depend on loblolly pine (*Pinus taeda*) stands for nesting or foraging. However, loblolly pine stands are suspected to decline. Forest managers want to replace loblolly pine with longleaf pine (*P. palustris*), but they must do this gradually to continuously supply RCW habitats. Knowledge of the current decline status and causal factors is therefore needed. We analyzed recent forest inventory data (until 2006) covering 8403 ha of naturally regenerated loblolly pine (LB) and 554 ha of loblolly pine plantations (LBP). Overall, LBP stands were healthier than LB and may be a useful RCW habitat option during a transition period to a landscape with sufficient amount of RCW usable longleaf pine stands. In order to draw conclusions regarding the decline status of loblolly pine forests on a landscape such as Fort Benning, it is necessary to understand natural stand development and dynamics, and to investigate further the causes of decline.

INTRODUCTION

In much of the Southeastern United States, post-European settlement land use practices, especially fire exclusion, have resulted in the replacement of historically dominant longleaf pine (*Pinus palustris* Mill.) with loblolly pine (*P. taeda* L.). This widespread conversion has many land managers concerned, largely due to the ecological significance of longleaf pine. For example, longleaf pine is the preferred habitat for the federally endangered red-cockaded woodpecker (*Picoides borealis*) (RCW), yet on lands supporting RCW populations the lack of longleaf pine has necessitated the use of loblolly pine for foraging and nesting (U.S. Department of the Interior, Fish and Wildlife Service 2003). Fort Benning, GA, is a good example of this phenomenon. The installation has about 36 400 ha of upland pine forest, of which <4 000 ha are classified as longleaf pine, with the balance dominated by loblolly pine (U.S. Army Infantry Center 2006). Consequently, two-thirds of the 330 active RCW clusters currently are in loblolly pine stands, including an estimated 70 percent of the natural RCW cavity trees (U.S. Army Infantry Center 2006).

Forest managers at Fort Benning are currently interested in restoring longleaf pine to upland sites dominated by loblolly pine. Although this goal could be achieved by clearcutting the existing loblolly pine stands and planting longleaf pine seedlings, conversion efforts are complicated in loblolly pine stands that are currently being used for RCW habitat. In such stands, longleaf restoration must occur by gradual conversion of loblolly stands, such that mature loblolly stands are retained for RCW habitat throughout the development of newly planted longleaf stands. This approach rests on the assumption that mature loblolly pine stands will remain healthy enough to support existing RCW populations until enough mature longleaf stands are available to support the RCW population. Recent reports of loblolly decline symptoms in the Southeastern United States (e.g., Eckhardt and Menard 2008) bring this assumption into question. Further, forest managers are concerned that ongoing loblolly decline could limit available RCW habitat and slow population recovery. Knowledge of the current status and underlying cause(s) of loblolly pine decline is needed to address this concern.

Symptoms of loblolly pine decline include short chlorotic needles, sparse crowns, and reduced radial growth by stand age 40 to 50 years, with mortality generally occurring 2 to 3 years after these symptoms are observed (Hess and others 1999). Previous studies report that loblolly pine decline typically occurs on well-drained soils (Eckhardt and Menard 2008, Eckhardt and others 2007), which dominate Fort Benning's upland pine sites. Loblolly pines prefer relatively rich and moist soils (Harper 1965), whereas dry, poor uplands are considered to be "offsite" and are likely to increase stress on pines growing there. Although the mechanisms of loblolly pine decline are not fully understood, poor belowground growth and loss of root function have been implicated as main causes for decline. Further, decline may be exacerbated by a host of abiotic and biotic variables, including landscape position, e.g., slope and aspect, soil physical and chemical properties, water stress, landscape legacy effects, pathogen infection, e.g., *Leptographium* spp., and unusual climate patterns.

The primary objective of this study was to assess the health status of loblolly pine stands at Fort Benning using existing field survey data collected by land managers on the installation. The dataset included traditional inventory measures, e.g., stem densities and basal area, several forest health metrics, e.g., crown vigor class, presence of decline symptoms, insects or other disease indicators, and ground cover data, e.g., vegetation cover and bare ground

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abundance. The study was conducted in two forest types: naturally regenerated, second-growth upland loblolly pine stands (LB) and loblolly pine plantations (LBP). LB stands provide much of the current RCW habitat, and LBP may provide future RCW habitat, so both were of interest.

MATERIALS AND METHODS

Study Site

Fort Benning is located in westcentral Georgia on the geographical Fall Line (fig. 1). The installation covers two major ecological provinces: the Sandhills in the northeastern two-thirds of the installation, and the Upper Loam Hills in the southwestern one-third. The terrain is rolling and ranges in elevation from 58 to 225 m above sea level (U.S. Army Infantry Center 2006). The climate is classified as warm humid temperate with hot, humid summers and mild winters. Mean annual precipitation is 1240 mm and is evenly distributed throughout the year (National Climatic Data Center, Asheville, NC). Major soil textures are loamy sand, sandy loam, and sandy clay loam.

Field Surveys

Fort Benning's Land Management Branch conducted an extensive, installation-wide inventory in 2005, with the primary objective of providing current stand and habitat information for RCW management. We analyzed the data collected through November 2006, which included information from 8403 ha of natural loblolly pine stands (LB) and 554 ha of loblolly pine plantations (LBP) (fig. 1). Prior to the survey, individual stands were delineated using the most recent aerial imagery. A stand was defined as a contiguous group of trees sufficiently uniform in species composition, age or arrangement of age classes, and site condition to be considered a distinguishable unit. Plantations <30 years old were considered homogeneous and all other stands were considered heterogeneous; minimum stand size was 4 ha, with a few exceptions.

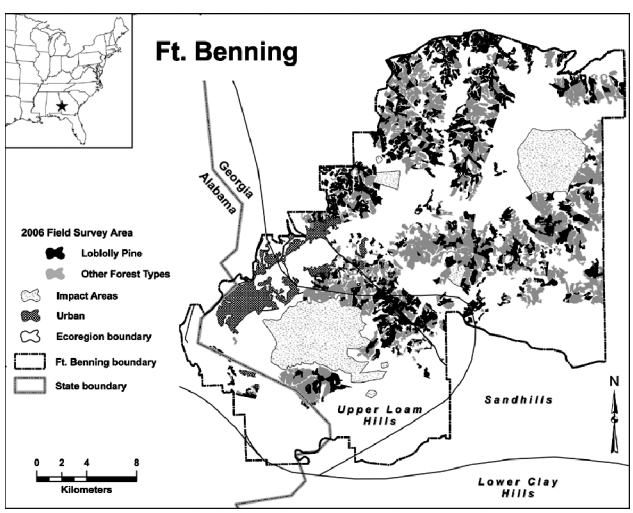

Figure 1—Geographical location of Fort Benning, GA, (inset) and the area surveyed as part of the most recent forest inventory. Loblolly pine indicates stands identified as loblolly pine and loblolly pine plantation within the survey area.

Field crews collected data from 10 sampling points in each homogeneous stand and 20 sampling points in each heterogeneous stand. In stands smaller than 4 ha, one sampling point was established per 0.8 ha in homogeneous stands and one sampling point was established per 0.4 ha in heterogeneous stands. To locate each sampling point, field crews identified a cruise route through the longitudinal axis of the stand and ran a compass line on this route. Cruise lines that tended to follow drains, ridges, trails, or other linear features were avoided. If a stand configuration was such that one line transect through the longitudinal axis did not result in enough sampling points to capture the variability of the stand, then the sampling scheme was modified in one of the following ways: (1) parallel transects were established two chains apart with sampling points 2 to 5 chains apart along each transect; (2) in a circular-shaped stand, sample transects were established in a triangular pattern; or (3) in a square-shaped stand, two perpendicular transects crossing through the center of the stand were established with sampling points established at 2- to 5-chain intervals along this route.

At each sampling point, variable radius 10-factor basal area prism plots were used to collect overstory data. Species and diameter at breast height (d.b.h.) of each tree larger than 12.5 cm (to nearest 0.25 cm) were recorded to describe stand structure and composition. Tree health was assessed by determining crown vigor class (CVC) following U.S. Forest Service, Forest Health Monitoring protocol (U.S. Forest Service 1999). Using this designation, each tree was assigned a "grade" to characterize canopy health (1 = good, 2 = fair, 3 = poor). CVC was mainly determined by crown ratio, crown dieback, and crown density, where CVC1 = crown ratio >35 percent, crown dieback <5 percent, and crown density >80 percent; CVC3 = crown ratio <35 percent, crown dieback >50 percent, and crown density <20 percent; and all other trees were classified as CVC2. Other potential health problems were recorded in an additional "insect or disease" category (ID), recorded as presence/absence of the following: (1) fusiform rust, (2) loblolly pine decline symptom (Symp), (3) annosus root rot, (4) black turpentine beetle [*Dendroctonus terebrans* (Oliv.)], and (5) other. Stand-level percentages of all pines exhibiting insect or disease conditions were used for the analysis. Hog damage (HD) and gopher tortoise burrow (GTB) presence were recorded within 400 m^2 fixed-radius plots centered on each sampling point. The ground cover was characterized by recording percent cover of herbaceous vegetation, woody vegetation, pine straw, and bare ground (including hardwood leaf litter) within 40 m^2 plots at each sampling point. Cover for each group was visually estimated in 10-percent increments, and when these four percentages were added together, their sum equaled 100 percent. Forest inventory data from previous surveys were used to determine stand age and site index (SI). If existing stand-age data were perceived to be incorrect, then dominant or codominant trees were cored to determine age. If the stand was a pine plantation, only one tree was cored.

Statistical Analysis

From the inventory data, we calculated pine basal area (BA, m^2/ha), pine stem density (SD, number of trees/ha), large pine (d.b.h. >35 cm) SD, hardwood BA, large hardwood (d.b.h. >35 cm) SD, and total BA of each stand. Pine tree health and condition data were analyzed at the stand level; mean CVC was calculated for each stand, and the percentage of all pines exhibiting Symp or other ID conditions were calculated for each stand. Differences in forest characteristics, e.g., BA, stem density, and SI, between the two forest types were tested by t-tests. Within each forest type, the effects of HD and GTB on CVC and ID were tested using t-tests, while effects on the Symp variable (percent of trees with decline symptoms) were tested using Wilcoxon rank sum. To meet the normality assumption, pine basal area (BA, m^2/ha), pine stem density (SD, number of trees/ha), hardwood BA, and CVC were transformed using a logarithmic function; SD of pine trees larger than 35 cm d.b.h. and ID were transformed by a square root function; and hardwood SD and total BA were transformed by an arcsine function. Symp data could not be transformed to follow a normal distribution, and data were therefore analyzed using Spearman rank test. All statistical analyses were performed using SAS (Version 9.01. SAS Institute Inc., Cary, NC).

RESULTS

Status of Forest Decline across Forest Types

Mean CVC and percentage of pines with ID were all significantly ($P < 0.05$) higher in naturally regenerated LB than LBP (table 1). Percentage of trees exhibiting Symp, indicated by sparse crowns and chlorotic needles, was also higher in LB than LBP, but the difference was not significant ($P > 0.05$). The majority (54 percent of area) of LB had intermediate to poor crown health, i.e., average CVC between 2 and 3, whereas roughly 25 percent (of area) of the LBP fell within this class. Results for ID were similar to Symp: 7 percent of LB (599 ha) had >50 percent ID, i.e., more than 50 percent of pine trees damaged by insect or disease, while there were no LBP with over 50 percent ID (fig. 2). At the same time, 33 percent (of area) of LB had <20 percent ID and about one-third of LB forest (2500 ha) showed between 20 and 30 percent of the stems with ID. Two-thirds (of area) of the LBP had <20 percent ID (fig. 2). Symp in LBP was always <10 percent, while 84 percent of LB had <10 percent Symp.

Table 1—Characteristics of naturally regenerated loblolly pine forests and loblolly pine plantations on Fort Benning, GA

Characteristics		Forest type, total area	
		Loblolly pine (ha)[a]	Loblolly pine plantation (ha)[a]
		8403	554
Stand condition	Stand age (year)	59 (20) a	31 (22) b
	Site index	79 (12)	84 (24)
	Stand size (ha)	13.9 (10.8) a	7.4 (11.9) b
Overstory condition	Pine basal area (m^2/ha)	10.3 (3.7) b	16.8 (6.6) a
	Pine stem density (number/ha)	179 (124) b	587 (311) a
	Pine (d.b.h. >35 cm) stem density (number/ha)	32 (15) a	6 (15) b
	Hardwood basal area (m^2/ha)	2.2 (1.8) a	0.4 (0.6) b
	Hardwood (d.b.h. >35 cm) stem density (number/ha)	4.7 (4.6) a	0.4 (1.4) b
	Total basal area (m^2/ha)	12.5 (4.0) b	17.1 (6.5) a
Health metrics	Crown vigor class	1.9 (0.3) a	1.7 (0.4) b
	Insect or disease (percent)	27.0 (14.3) a	15.9 (11.8) b
	Pine decline symptom (percent)	4.7 (8.6)	0.4 (1.1)
Ground cover	Herbaceous (percent)	25 (10)	22 (12)
	Woody plants (percent)	22 (10)	20 (11)
	Bare ground (percent)	26 (14) a	13 (11) b
	Pine straw (percent)	27 (11) b	44 (18) a

Data are presented as mean values (1 standard deviation). BA and d.b.h. indicate for basal area and diameter at breast height, respectively.

[a] Different letters within a row indicate a significant difference ($P < 0.05$) between forest types.

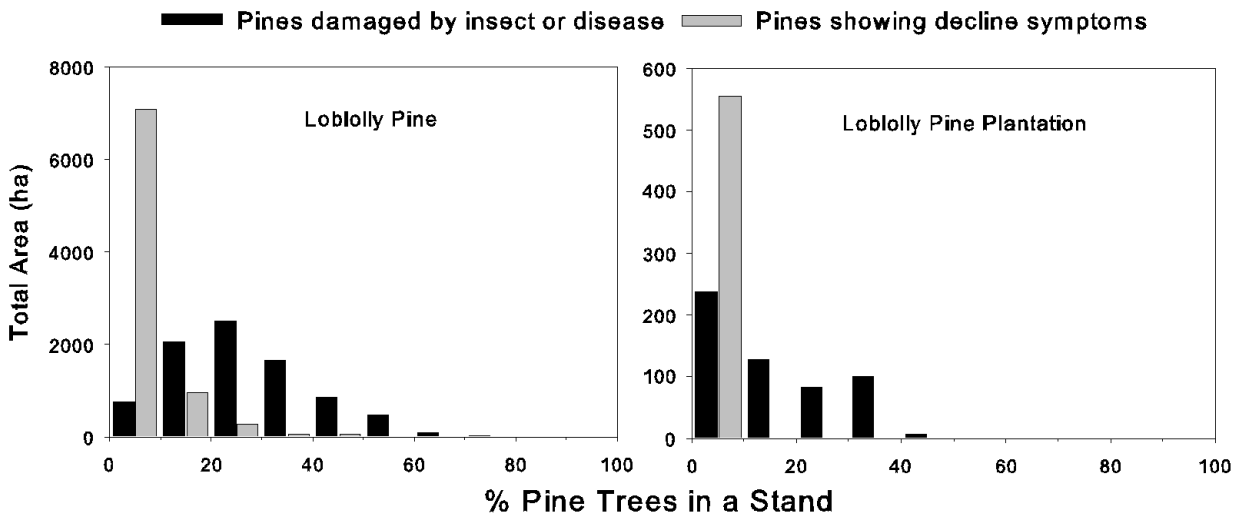

Figure 2—Total area of loblolly natural stands (LB) and loblolly plantations (LBP) surveyed by 10 percentile insect/disease and decline classes.

Relationship of Forest Health Metrics and Environmental Variables

SI was similar between LB and LBP forests. As expected, LBP stands were significantly ($P < 0.05$) younger, denser, and had smaller trees than LB stands (table 1). LBP stands had more than triple the number of pine trees per unit area and 25 percent more BA than LB stands. However, the number of large pine trees (d.b.h. >35 cm) in LBP stands was <20 percent of those in LB stands (table 1).

Although some significant relationships between the health metrics and pine SD emerged, there was no clear pattern (table 2). For example, correlation showed that stands tended to have poorer crowns (higher CVC) with higher SD ($P < 0.05$) in LB, but higher Symp percent occurred at lower SD in LBP stands ($P < 0.05$). LBP stands had significantly ($P < 0.05$) lower hardwood BA and SD compared to LB stands, but again the influence of hardwoods on the health metrics was difficult to discern.

Percent herbaceous and woody ground cover was similar between LB and LBP stands, but LBP had significantly ($P < 0.05$) more pine straw and less ($P < 0.05$) bare ground than LB stands (table 1). Results from the correlation analysis between ground cover variables and the health metrics were noisy but suggested some relationships (table 2). Within LB stands, stands were healthier (lower CVC value) with greater percent woody plant cover and less pine straw cover; pine ID increased with the increase of herbaceous and woody plant cover and the decrease of bare ground and pine straw cover; pine Symp was found with higher herbaceous cover and lower pine straw cover. Within LBP stands, stand health decreased (higher CVC value) and pine Symp increased with the increase of bare ground.

Tree health metrics generally were significantly correlated with one another. Stands were unhealthier (higher CVC value) when pine ID was higher ($r = 0.12$, $P < 0.01$ for LB; $r = 0.41$, $P < 0.01$ for LBP), and when pine Symp was higher ($r = 0.15$, $P < 0.01$ for LBP only). Incidence of decline (Symp) was positively correlated with ID ($r = 0.20$, $P < 0.01$ for LB; $r = 0.36$, $P < 0.01$ for LBP). We did not find any significant ($P < 0.05$) effect of soil surface disturbance from GTB or HD on CVC, ID, or Symp in either forest type.

DISCUSSION

Is Fort Benning at Risk of Loblolly Pine Decline?

All health metrics suggest that LBP are healthier than natural LB stands on Fort Benning. One explanation could be related to age; LBP stands were younger than LB stands (59 and 31 years for LB and LBP, respectively), and loblolly pine decline is generally associated with stands older than 40 years (Hess and others 1999). However, we found no significant ($P < 0.05$) correlation between stand age and the health metrics within each forest type. We hypothesize that there may be a threshold age beyond which the likelihood of decline increases, e.g., 40 years, but we did not find any specific patterns from our data. Regardless of the underlying mechanism, relatively healthier LBP stands may become more valuable over time, especially if existing natural stands continue to decline or decline becomes more widespread in LB stands. Though LBP stands may eventually decline as well, their current status suggests that they may provide critical, short-term RCW habitat until younger longleaf pine plantations develop into suitable RCW habitat.

Interpreting the health status of Fort Benning's pine forest is complicated by the fact that there is no robust, universal definition of loblolly pine decline. It is generally held that forest decline refers to a continuous loss of vigor or health associated with an unclear causal factor or with complex interactions between biotic and abiotic factors, and previous studies (e.g., Eckhardt 2003, Houston 1992, Manion 1991) defined loblolly pine decline as "a gradual deterioration in

Table 2—Spearman correlation of tree health metrics (crown vigor class, insect or disease, and decline) with other variables in naturally regenerated loblolly pine forests and loblolly pine plantations

	Loblolly pine (n = 603 except n of SI = 577)			Loblolly pine plantation (n = 76 except n of SI = 41)		
Stand characteristics	Crown vigor class	Insect or disease	Decline symptom	Crown vigor class	Insect or disease	Decline symptom
Site index	–0.09	–0.00	–0.07	–0.06	0.08	0.12
Stand age (year)	–0.06	–0.04	–0.03	0.07	–0.07	0.10
Pine basal area (m^2/ha)	0.05	–0.07	0.01	–0.19	0.09	–0.20
Pine stem density (number/ha)	0.11[a]	–0.19[a]	0.02	–0.19	–0.19	–0.39[a]
Pine (d.b.h. >35 cm) stem density (number/ha)	–0.04	0.04	0.14	0.05	0.17	0.03
Total basal area (m^2/ha)	0.02	–0.14[a]	–0.02	–0.19	0.11	–0.15
Herbaceous (percent)	–0.00	0.14[a]	0.12[a]	0.02	0.06	0.15
Woody plant (percent)	–0.12[a]	0.13[a]	–0.08	–0.20	0.03	0.02
Bare ground (percent)	–0.01	–0.08[a]	0.05	0.22[a]	–0.13	0.31[a]
Pine straw (percent)	0.13[a]	–0.21[a]	–0.12[a]	–0.08	0.11	–0.21
Crown vigor class	—	0.12[a]	–0.05	—	0.41[a]	0.15[a]
Insect or disease (percent)	0.12[a]	—	0.20[a]	0.20[a]	—	0.36[a]
Decline symptom (percent)	–0.05	0.20[a]	—	0.15[a]	0.36[a]	—

SI = site index.

[a] Indicates significant ($P < 0.05$) correlations.

health and vigor of canopy-dominant trees that frequently ends in death." However, this definition does not distinguish natural mortality due to aging from decline and provides no practical threshold for making consistent judgments. Self-thinning mortality is a natural process, common to all stages of forest development, that can be influenced by many stand and site conditions. Despite results from an extensive field survey (>9000 ha of loblolly pine forests), the ambiguity surrounding what constitutes loblolly pine declines makes it difficult to draw definitive conclusions about the presence of decline on the installation.

Factors Associated with Loblolly Pine Health

The results from correlation analysis were inconsistent, making it difficult to draw definite conclusions about relationships between decline and possible causal factors from our dataset. This was especially true in the LBP stands, probably due to the narrow range in health metrics recorded and the smaller sample size of LBP stands. Only 11 stands among 76 LBP stands had pine trees classified as Symp, limiting the interpretation of LBP correlation tests. Therefore, the following discussion will focus on the naturally established LB stands.

CVC was significantly, negatively correlated ($P < 0.05$) with SI, indicating that site conditions may play a role in reduced tree health. Many of the upland pine stands on the installation are on sandy, well-drained, nutrient-poor growing sites. Loblolly pines are known to be mature at age 80 and begin to naturally lose vigor at age 150 (Harper 1965), but poor-growing conditions may accelerate natural senescence, resulting in concern about "decline." Moreover, given that loblolly pine demands more nutrients than other pines (Baker and Langdon 1990), soils on the installation may be insufficient for healthy loblolly pine growth. Symptoms of nutrient deficiency in trees are often quite similar to those reported as loblolly pine decline. For example, Smethurst and others (2007) suggested that potassium (K) deficiency was the main cause of chlorotic needles and sparse canopies of radiata pines (*P. radiata* D. Don) in Australia. Further, Hess and others (1999) reported very low K in the soils of declining loblolly pine stands in Alabama, suggesting a connection between

site nutrients and tree health. Although we have no conclusive evidence that currently reported loblolly pine decline is associated with nutrient deficit, it is likely that nutrient deficiencies contribute to a loss of pine vigor.

Patterns of ground cover vegetation are often useful indicators of site disturbance and, in particular, fire history. Understory vegetation (herbaceous and woody plant cover) in pine stands often increases after prescribed fire (e.g., Hendricks and Boring 1999) and prescribed fire combined with thinning (Wayman and North 2007). Prescribed fire and thinning could result in both pine straw cover decrease and understory cover increase, factors we found associated with increasing ID and Symp. This raises questions about the relation between ground cover and overstory tree health; perhaps, rather than a direct link between ground cover and loblolly pine vigor, current ground cover is a reflection of past management history that has influenced both ground cover and overstory health.

Forests on Fort Benning are heavily managed, including recent reintroduction of prescribed burning, and it is possible that management activities have added stress to pines. The installation burns approximately 12 000 ha/year on an approximately 3-year rotation with prescribed fire, and additional wildfires due to military munitions are common. Prescribed fire was introduced at this scale in 1994, following U.S. Fish and Wildlife Service recommendations for RCW management (U.S. Army Infantry Center 2006). Although prescribed burning improves habitat structure for RCW, fire could negatively influence belowground pine production by reducing nutrient availability (Raison and others 1985), decreasing water infiltration rates (DeBano 2000), decreasing water holding capacity (Boyer and Miller 1994), and reducing soil organic matter and soil porosity (Busse and others 2000, Landsberg 1994, Tiedemann and others 2000). At the time of reintroduction, little was known about the precautions managers should take when burning in areas with high organic matter accumulation, i.e., duff.

It is likely that the manifestation of decline symptoms is a response to a combination of stress factors, including many not accounted for within the dataset used in this study. Historical land use, e.g., landscape legacy, has lasting effects on growing conditions that play a critical role in shaping current stand health. Prior to Fort Benning's establishment as a military installation, the region was heavily farmed (U.S. Army Infantry Center 2006), resulting in massive erosion and soil degradation. Present day timber harvesting and feral hogs' behavior may each reduce belowground productivity by increasing soil compaction and physically damaging root systems. Further, effects of military training on tree growth are not fully understood. In addition to land use, climate patterns and/or climate change may affect belowground dynamics by altering temperature and precipitation patterns. For example, Fort Benning has experienced several severe droughts in the last decade, increasing moisture stress on sandy sites with intrinsically low water holding capacity. Understanding the role of climate (and global warming, in particular), as well as many of the other possible contributors to loblolly pine decline in this area, would require extensive, long-term study on a host of biotic and abiotic stress variables.

CONCLUSIONS AND FUTURE NEEDS

It was clear that naturally regenerated LB stands were less vigorous than LBP at Fort Benning and more often exhibited symptoms of ID. It may be strategic to maintain pockets of LBP to serve as a bridge for future RCW habitat. Overall, the percentage of trees with decline symptoms seems modest, making it difficult to determine if LB at the installation is really declining. To make a concrete determination regarding the extent of LB decline, we would require the (1) development of a mortality/vigor threshold to determine stand decline, e.g., stand mortality >15 percent and percent of trees in low vigor >30 percent at a given time indicates decline, and (2) an understanding of the dynamics of tree mortality and stand development at Fort Benning. We did observe a positive relationship between crown health and SI, suggesting that the growing site and associated resource availability may limit LB growth on the installation. Our results suggest that future work should be aimed at evaluating practical criteria to determine LB decline and its underlying causes, including nutrient availability and forest management practices.

ACKNOWLEDGMENTS

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LITERATURE CITED

Baker, J.B.; Langdon, O.G. 1990. Loblolly pine (*Pinus taeda* L.). In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: 1. Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service: 497–512.

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.

Busse, M.D.; Simon, S.A.; Riegel, G.M. 2000. Tree-growth and understory responses to low-severity prescribed burning in thinned *Pinus ponderosa* forest of central Oregon. Forest Science. 46: 258–268.

DeBano, L.F. 2000. The role of fire and soil heating on water repellence in wildland environments: a review. Journal of Hydrology. 231/232: 195–206.

Eckhardt, L.G. 2003. Biology and ecology of *Liptographium* species and their vectors as components of loblolly pine decline. Baton Rouge, LA: Louisiana State University, The Department of Plant Pathology and Crop Physiology. [Number of pages unknown]. Ph.D. dissertation.

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

Eckhardt, L.G.; Weber, A.M.; Menard, R.D. [and others]. 2007. Insect-fungal complex associated with loblolly pine decline in central Alabama. Forest Science. 53(1): 84–92.

Harper, V.L. 1965. Silvics of forest trees of the United States. Agric. Handb. 271. Washington, DC: U.S. Department of Agriculture Forest Service. 762 p.

Hendricks, J.J.; Boring, L.R. 1999. N_2-fixation by native herbaceous legumes in burned pine ecosystems of the Southeastern United States. Forest Ecology and Management. 113: 167–177.

Hess, N.J.; Otrosina, W.J.; Jones, J.P. [and others]. 1999. Reassessment of loblolly pine decline on the Oakmulgee District, Talladega National Forest, Alabama. Rep. 99-2-03. Pineville, LA: U.S. Department of Agriculture Forest Service, Forest Health Protection. [Number of pages unknown].

Houston, D.R. 1992. A host-saprogen model for forest dieback-decline diseases. In: Manion, P.D.; Lachance, D. eds. Forest decline concepts. St. Paul, MN: American Phytopathological Society Press: 3–25.

Landsberg, J.D. 1994. A review of prescribed fire and tree growth response in the genus *Pinus*. Proceedings, 13th conference on fire and forest meteorology. Society of American Foresters. 326 p.

Manion, P.D. 1991. Tree disease concepts. 2d ed. Englewood Cliffs, NJ: Prentice Hall, Inc. 402 p.

Raison, R.J.; Khanna, P.K.; Woods, P.V. 1985. Transfer of elements to the atmosphere during low-intensity prescribed fires in three Australian subalpine eucalypt forests. Canadian Journal of Forest Research. 15: 657–664.

Smethurst, P.; Knowles, A.; Churchill, K. [and others]. 2007. Soil and foliar chemistry associated with potassium deficiency in *Pinus radiate*. Canadian Journal of Forest Research. 37: 1093–1105.

Tiedemann, A.R.; Klemmedson, J.O.; Bull, E.L. 2000. Solution of forest health problems with prescribed fire: are forest productivity and wildlife at risk? Forest Ecology and Management. 127: 1–18.

U.S. Army Infantry Center. 2006. Integrated natural resources management plan, Fort Benning Army Installation, Incremental Revision 2006. Fort Benning, GA: Directorate of Public Works, Environmental Division. 1,037 p.

U.S. Department of Agriculture Forest Service. 1999. Forest Health Monitoring 1998 field method's guide. Research Triangle Park, NC: U.S. Department of Agriculture Forest Service, National Forest Health Monitoring Program. [Number of pages unknown].

U.S. Department of the Interior, Fish and Wildlife Service. 2003. Draft red-cockaded woodpecker recovery plan. Atlanta. [Number of pages unknown].

Wayman, R.B.; North, M. 2007. Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. Forest Ecology and Management. 239 (1-3): 32–44.

HYDROLOGIC INFLUENCE ON SEDIMENT TRANSPORT OF LOW-GRADIENT, FORESTED HEADWATER STREAMS IN CENTRAL LOUISIANA

Philip Saksa, Yi Jun Xu, and Richard Stich[1]

Abstract—Extensive research has been conducted on headwater streams in regions with high topographic variation. However, relatively few studies have examined low-gradient headwater stream systems, such as those existing in much of the southeastern Coastal Plain. The focus of this study is to investigate spatial and temporal variation of headwater stream hydrology in a low-gradient forested watershed and determine their effect on the transport of suspended and dissolved sediment. Stream discharge and sediment loads were monitored from December 2005 to April 2007 throughout a central Louisiana third-order watershed, with stream channel slopes of <1 percent. The study found that headwater streamflow in this low-gradient forested watershed was highly variable, from intermittent/no-flow conditions in the late summer, to high-volume overbank conditions in the winter. Transitioning from the headwater streams to the watershed outlet, stream hydrologic response and streamflow variability decreased. Suspended and dissolved solid concentrations during baseflow showed minimal seasonal variation, and loading was mainly controlled by discharge levels. Sediment yield from the watershed was low, due in part to the below normal precipitation and subsequent low storm runoff. As most of the land use in the watershed is commercial forest management, the low runoff decreases erosion susceptibility from harvesting activities. However, caution must also be taken and full implementation of forestry best management practices is recommended as harvest sites can become quickly saturated following precipitation events, creating the potential for unchecked surface runoff and sediment delivery to streams.

INTRODUCTION

Headwater streams comprise over 77 percent of all streams in the United States, encompassing almost half of the total stream length (Leopold and others 1964). Contributing an estimated 70 percent, their contribution and importance to hydrological processes and water quality in all watersheds are considerable. Although forested headwaters have been intensively studied for over a century, few studies exist on the unique processes of low-gradient meandering streams in the southeastern Coastal Plain in the United States. This gap in research and knowledge is especially important, as forests cover approximately 55 percent of the land cover in the Southeast (Flather and others 1990), occupying a large portion of headwater areas. The region has low average land slope and very low-channel slopes, creating extensively meandering streams with very low velocities and seasonally inundated backwaters.

Complexities created by the low-gradient topography and locally elevated ground water located on the Coastal Plain headwaters can make the quantification of sediment yield difficult. Assessment of sediment delivery to streams in this region is critical, however, as it is the primary pollutant to streams from forest-dominated land (Patric and others 1984). During high-flow periods, overbank flooding and reconnection of backwater channels and oxbows complicates in-channel sources and sinks of sediments. Additionally, locally minor variations in topography and large woody debris create complex channel velocities, affecting individual site sedimentation characteristics (Hupp 2000).

In their review of several coastal forested watershed studies, Amatya and others (2005) commented on the limited number of hydrology and water budget studies in these complex and complicated areas and expound on the need for long-term ecohydrologic monitoring to more fully determine the effects of forest management on water quality. In this study, we established a relatively long-term experiment in a low-gradient, forested watershed on the Louisiana Coastal Plain region to determine timber harvest effects on surface hydrology and water quality. In this paper, we analyze data collected from the first 2 years and discuss hydrologic effects on sediment concentrations and loading in this headwater region.

METHODS

Site Description

Located in northcentral Louisiana (fig. 1), the Flat Creek watershed has a drainage area of 369 km^2 and is characterized by relatively flat, low sloping forest land and pasture. Elevation ranges from 24 m at the southern outlet to a high of 91 m in the northern uplands. The long-term (1971 to 2000) average annual temperature in the area was 17.9 °C, ranging from 7.2 °C in January to 27.5 °C in July, and the long-term average annual precipitation was 1508 mm with a low of 90.7 mm in September and a high of 157.7 mm in December (National Climatic Data Center 2002). Soils in the area mainly consist of the poorly drained Guyton (silt loam) series along the Flat and Turkey Creek flood plains, with moderately well drained Sacul-Savannah (fine sandy loam) soils in the upland areas.

Analyzing a LandSat-5 TM image from May 16, 2006, shows evergreen forests dominating land cover with 51.4 percent, followed by deciduous forests at 32.6 percent, recovering harvested areas (1 to 3 years) at 7.0 percent, recently

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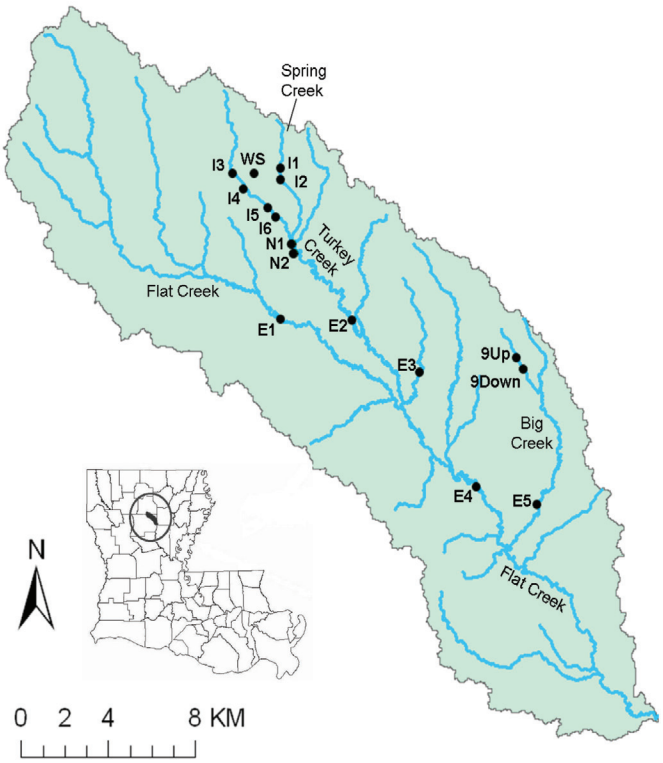

Figure 1—Geographical location of Flat Creek watershed in Louisiana.

harvested areas (<1 year) at 5.0 percent, pasture at 3.9 percent, and surface water making up the final 0.1 percent. A number of county and local dirt roads exist throughout the area. These roads may be significant sources of stream sediment, particularly where they cross or are adjacent to streambeds (Jones and others 2000). Beaver dams are also prevalent throughout the stream network and have been found to be particularly frequent along Turkey Creek.

Field Measurements and Sampling

Automatic samplers were installed at six sites (I1 to I6, fig. 1) along two first-order streams to collect stormwater samples and record stream level. The samplers recorded stream level in continuous 15-minute intervals from December 2005 to April 2007 and were also used to collect storm event water samples to determine effects on stream sediment loading. Programmed to start sampling at a 0.5-foot rise in stream level, the samplers collected 400 mL samples hourly for a period of 20 hours (20 times 400 mL = 8 L composite sample), which were then reduced to 1000 mL unfiltered and 500 mL filtered samples. Baseflow and stormflow 500 mL samples were filtered with a 47-µm glass fiber filter (GF/F) (Whatman International Ltd., Maidstone, UK). The 500-mL and 1000-mL samples were analyzed for total dissolved solids (TDS) and total suspended solids (TSS), respectively, by the Louisiana State University AgCenter Chemistry Laboratory (Baton Rouge, LA). Samples were processed in accordance with U.S. Environmental Protection Agency (USEPA) procedures, with a holding time of 7 days and storage at 4 °C. The test for suspended solid concentration had a detection limit of 5.0 mg/L; samples less than this level were estimated at 2.5 mg/L.

In addition, monthly baseflow water samples were collected at these and five other locations (E1–E5, fig. 1) distributed across the watershed. In monthly sampling, streamflow velocity was measured using an Acoustic Doppler Velocimeter (FlowTracker, SonTek/YSI, Inc., San Diego, CA). The stream level and velocity data were used to develop a stage-discharge rating curve for the monitoring sites. A weather station (4-channel HOBO Micro Station, Onset Computer Corporation, Bourne, MA) was installed near stream sampling locations (WS, fig. 1) to obtain relevant climatic parameters of air temperature, precipitation, wind, and solar radiation.

Development of Stage-Discharge and Sediment Rating Curves

A stage-discharge rating curve was developed for each stream sample site using stream level and velocity measurements. The curve was fitted through a natural log transformation as given below:

$$\ln(Q(t)) = b_0 + b_1 \ln(L(t)) + \varepsilon(t)$$

where
Q = discharge (m^3/second)
$L(t)$ = stream level (m)

The stage gages and water level loggers installed at the extensive sites were used to similarly develop a discharge rating curve for each sample location. Relationships were initially determined between the extensive level stage-gage records and other intensive monitoring locations where daily level data was available for the study period. The water level loggers installed in January 2007 were used to relate discharges between all other extensive sites and an associated intensive site, where daily discharge information was available, to determine the extensive site daily discharges previous to the logger installation. Discharge at site E1 did not show a good relationship with any intensive site and subsequently could not be calculated. This may have been due to the spatial variation of precipitation inputs or differences in individual site characteristics.

A log-linear regression model was developed to estimate TSS and TDS loadings at all sites:

$$\ln(S_i(t)) = b_0 + b_1 \ln(Q_{day}(t)) + \varepsilon(t)$$

where
Q_{day} = daily discharge (m^3)
$S(t)$ = daily loading (kg)
i = the type of solid
$\varepsilon(t)$ = an error term assumed to be normally distributed.

The regression was performed using SAS Statistical Software (SAS Institute Inc., 1996). The fitted parameters used to estimate discharge and solids loadings are summarized in

table 2. Stage-discharge relationships for I5 and I6, impacted heavily by beaver and debris dams, were unsuccessful and resulted in an inability to determine TSS and TDS loading relationships.

RESULTS AND DISCUSSION

Hydrologic Conditions

Precipitation during the study period from December 2005 to April 2007 was below the long-term average observed from 1971 to 2000 (National Climatic Data Center 2002). Only 3 months (Feb., Oct., and Dec. 2006) showed higher precipitation than the long-term average (fig. 2). Precipitation in March to September 2006 was low, representing 54 percent of the long-term average amount for the same period. The largest storm event occurred on Oct. 15 and 16 where 185 mm of rain fell, 11 percent of the entire 17 months observed.

Streamflow during the study period was similarly variable. Discharges generally peaked in February 2006 and December 2006/January 2007 due to a combination of high precipitation and wet antecedent conditions during those months (fig. 3). All sites experienced intermittent, no-flow conditions in the late summer months of 2006 due to low precipitation. The large storm in October 2006 came after this dry period and returned all streams to a connected, actively flowing status. Discharge is most likely underestimated for this month as streams extensively overflowed their banks, flooding the riparian zone and were beyond the extents of the developed stage-discharge relationships. Although bank overflow occurred several times during the course of the study, it was not as extreme or long lasting.

Seasonal and Spatial Variations in TSS and TDS Concentrations

TSS concentrations generally showed expected responses to streamflow conditions. Highest levels were observed following initial increases of streamflow after long dry periods, as in December 2005 and November 2006, and particularly in

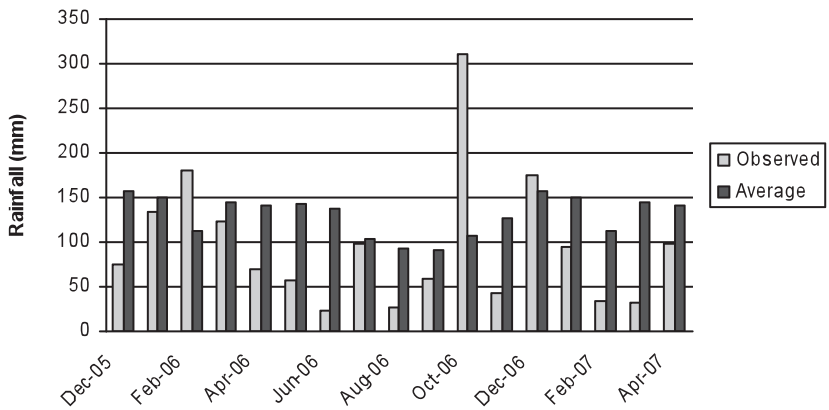

Figure 2—Monthly observed and average (1971–2001) precipitation for the 17-month study.

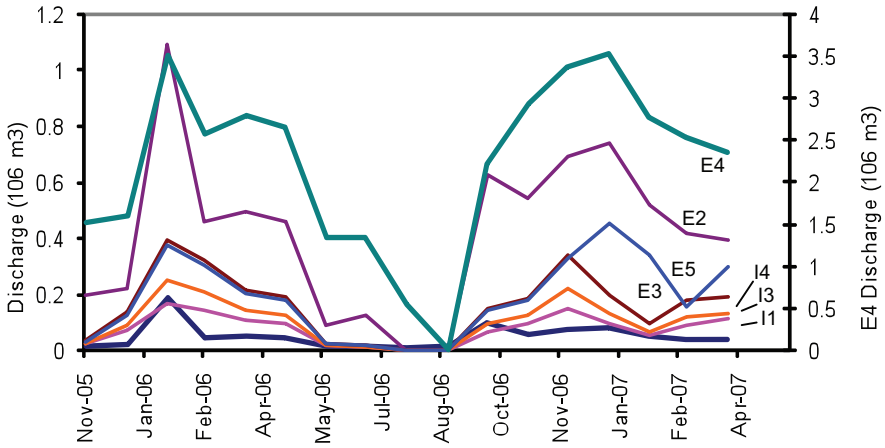

Figure 3—Total monthly discharge for all study sites where flow could be determined. E4 discharge is on the right ordinate due to a much higher magnitude of streamflow.

streams draining the largest areas (table 1). Generally, higher values were seen in the wet winter months, with lower values in the dry summer period. Average monthly values across all sites ranged from 5.7 mg/L in July 2006 to 38 mg/L in December 2005.

Mean monthly TDS concentrations from all sites ranged from 79.8 to 148.3 mg/L in December 2006 and December 2005, respectively (table 2). Generally, increasing values were observed as the streams returned from dry, no-flow conditions, to the higher winter discharges as in November and December 2006. This could possibly be due to higher levels of the local water table, increasing the influence of ground water on streamflow.

Although no clear spatial variation existed in suspended solid concentration, maximum values were highest in the streams with the largest drainage areas—E2 and E4 (fig. 4). Also, no clear spatial variation of TDS concentrations, due to position in the watershed or stream network, was found (fig. 4).

However, geomorphic characteristics may have played a role in concentration of TDS in a stream. Stream sites I1 and E2, that were straight and narrow with high velocities and had a high position upon the landscape, had the lowest overall TDS concentrations. Other sites with wider and deeper streams, and slower velocities, were at times characterized as pools due to extremely low flow, but contained a relatively large volume of water. Sites characterized as pools may provide greater time for the streamwater to interact with the adjacent soil and ground water in the hyporheic (adjacent riparian) zone, increasing TDS levels. Sharp TDS peaks in July 2006 during extremely low-flow conditions, and just before a majority of the streams became intermittent, suggest the same phenomena. Although site E4 also has higher velocity than most other sites, its wetted area was the largest and the

Table 1—Stream total suspended solid concentrations determined from monthly baseflow water sampling over the study period

	Total suspended solids						
Sites	I1	E3	I3	I4	E5	E2	E4
	mg/L						
Dec-05	12.0	19.0	9.0	15.0	37.0	70.0	105.0
Jan-06	2.5	2.5	12.1	16.1	12.1	14.1	10.1
Feb-06	8.1	25.5	16.2	11.2	19.1	11.1	21.2
Mar-06	9.2	14.3	9.2	5.1	9.2	5.1	2.5
Apr-06	25.5	22.9	23.5	28.3	25.5	20.1	20.0
May-06	8.1	9.1	5.2	2.5	10.2	19.2	2.5
Jun-06	14.4	14.2	13.3	20.5	2.5	—	19.3
Jul-06	6.2	2.5	2.5	9.1	5.1	5.0	9.3
Aug-06	6.1	34.1	—	9.0	2.5	—	2.5
Sep-06	—	—	—	—	6.1	—	—
Oct-06	19.2	—	—	—	—	—	—
Nov-06	25.9	19.0	26.9	26.1	28.3	43.6	39.8
Dec-06	2.5	25.9	23.1	24.3	25.5	19.4	26.7
Jan-07	8.1	14.1	14.1	13.1	23.3	6.2	19.6
Feb-07	17.4	17.2	10.1	10.2	15.2	7.1	6.1
Mar-07	25.2	28.6	31.6	18.3	28.2	18.3	30.6
Apr-07	15.2	17.0	19.6	15.1	19.3	22.2	34.4
Mean	12.8	17.7	15.5	14.9	16.8	20.1	23.3
±SD	±7.9	±8.9	±8.5	±7.5	±10.5	±18.2	±25.6

— = no flow during monthly sampling; SD = standard deviation.

Table 2—Stream total dissolved solid concentrations determined from monthly water baseflow sampling

	Total suspended solids						
Sites	I1	E3	I3	I4	E5	E2	E4
	mg/L						
Dec-05	74.0	186.0	145.0	169.0	121.0	90.0	253.0
Jan-06	91.1	113.5	140.9	135.9	109.9	93.9	140.9
Feb-06	92.9	77.5	102.8	108.8	123.9	83.9	79.8
Mar-06	87.4	100.7	126.8	141.9	130.8	96.9	106.5
Apr-06	82.5	111.1	115.5	108.7	116.5	100.9	109.0
May-06	126.9	124.9	148.8	105.5	141.8	117.8	99.5
Jun-06	72.8	118.8	114.7	116.5	126.5	—	154.7
Jul-06	70.5	64.2	118.5	104.9	92.4	56.7	116.7
Aug-06	106.9	130.9	—	117.0	162.5	—	173.5
Sep-06	—	—	—	—	128.9	—	—
Oct-06	107.8	—	—	—	—	—	—
Nov-06	86.1	101.0	117.1	108.9	79.7	71.4	59.6
Dec-06	98.5	61.6	82.9	101.7	62.7	65.2	86.3
Jan-07	108.9	99.9	96.9	64.3	99.7	94.8	96.4
Feb-07	89.6	114.8	123.9	97.8	132.8	99.9	124.9
Mar-07	120.8	100.4	123.4	145.7	139.8	86.7	101.4
Apr-07	156.8	104	133.4	138.9	150.7	100.8	120.6
Mean	98.3	107.3	120.8	117.7	120.0	89.2	121.5
±SD	±22.7	±29.7	±18.4	±25.1	±26.1	±16.6	±46.5

— = values represent no flow during monthly sampling; SD = standard deviation.

site was located in an area of extensive backwaters, providing a similar interaction with soils as the pooled sites.

Difference in TSS and TDS Concentrations between Baseflow and Storm Events

Stormflow can result in the highest rates of suspended solids loading due to increased erosion and the large volume of discharge water. Storm events in a forested catchment on Penang Hill, Malaysia, accounted for only 12.7 percent of the streamflow throughout the year, but were responsible for 60 percent of the TSS load (Ismail 2000). Storm sample TSS concentrations for this study ranged from <5.0 mg/L (I4, I5) to 109 mg/L (I1). Average storm samples of suspended solids consistently produced two to five times higher concentrations than average monthly baseflow samples for all sites (fig. 5). Sites I5 and I6, impacted most by beaver and debris dams, show the least differences between the two types of sampling. Increasing drainage area and stream size likely also contributed to the settling of sediments before reaching the most downstream sites, with dams simply increasing the magnitude of these effects.

Storm sample TDS concentrations across all sites ranged from 54.3 mg/L (I4) to 188.8 mg/L (I3). Mean concentrations generally increased with increasing drainage area and may be due to having a lower position in the watershed and more influenced by baseflow levels with higher TDS. Unlike the suspended sediments, TDS concentrations during the baseflow and storm events did not differ significantly except for at one site, I5 (fig. 5). Increased TDS concentrations in storm samples at I5 may have been influenced by runoff from a paved road and bridge located directly upstream of the site. No other site was located near a paved road.

Sediment Loading and Fluxes

Looking at the TSS loading over the study period (fig. 6), the level of streamflow shows a greater influence on loading than variations in concentration. Streamflow conditions, influenced

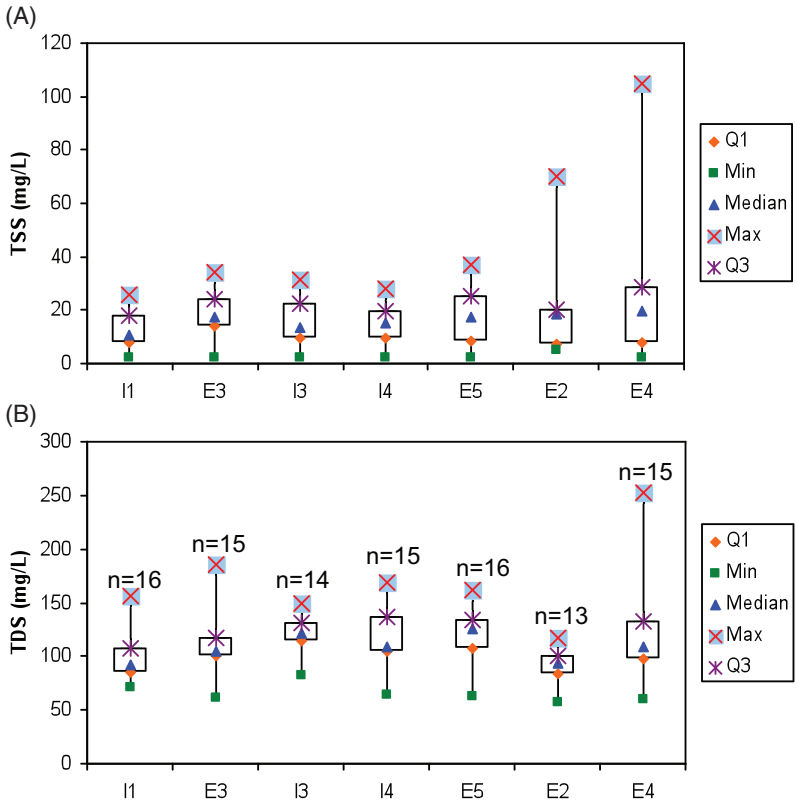

Figure 4—Box and whisker plots of monthly baseflow (A) total suspended solid and (B) total dissolved solid concentrations. Boxes show values in the middle 50 percent, bounded by first and third quartiles (Q1 and Q3) and sites are arranged from lowest to highest drainage area. Sample numbers (*n*) apply to both plots and variations are due to dry periods where no surface flow existed at the site and no sample was collected.

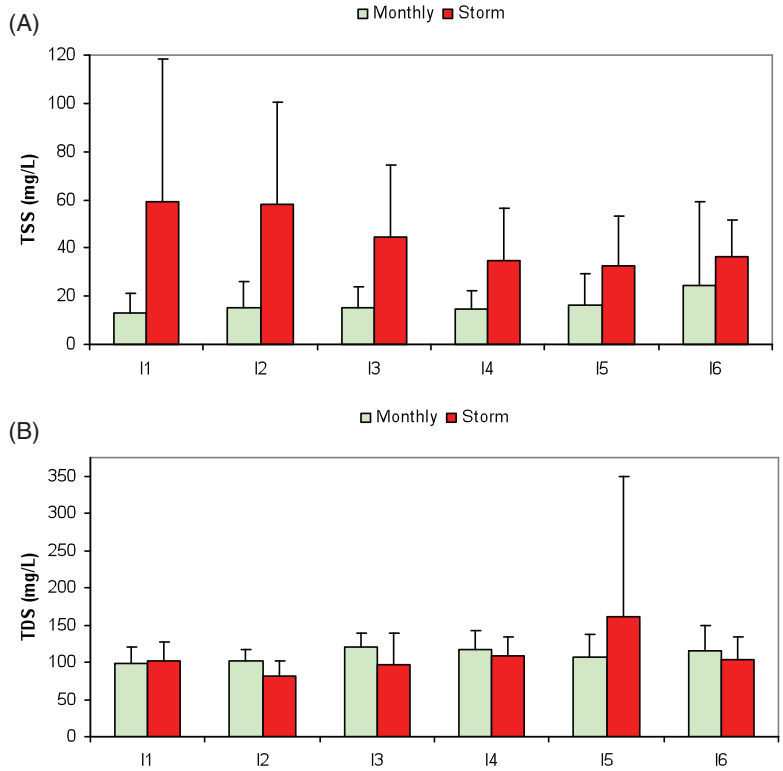

Figure 5—Average monthly baseflow and storm sample concentrations of (A) total suspended solid and (B) total dissolved solids with standard deviation bars.

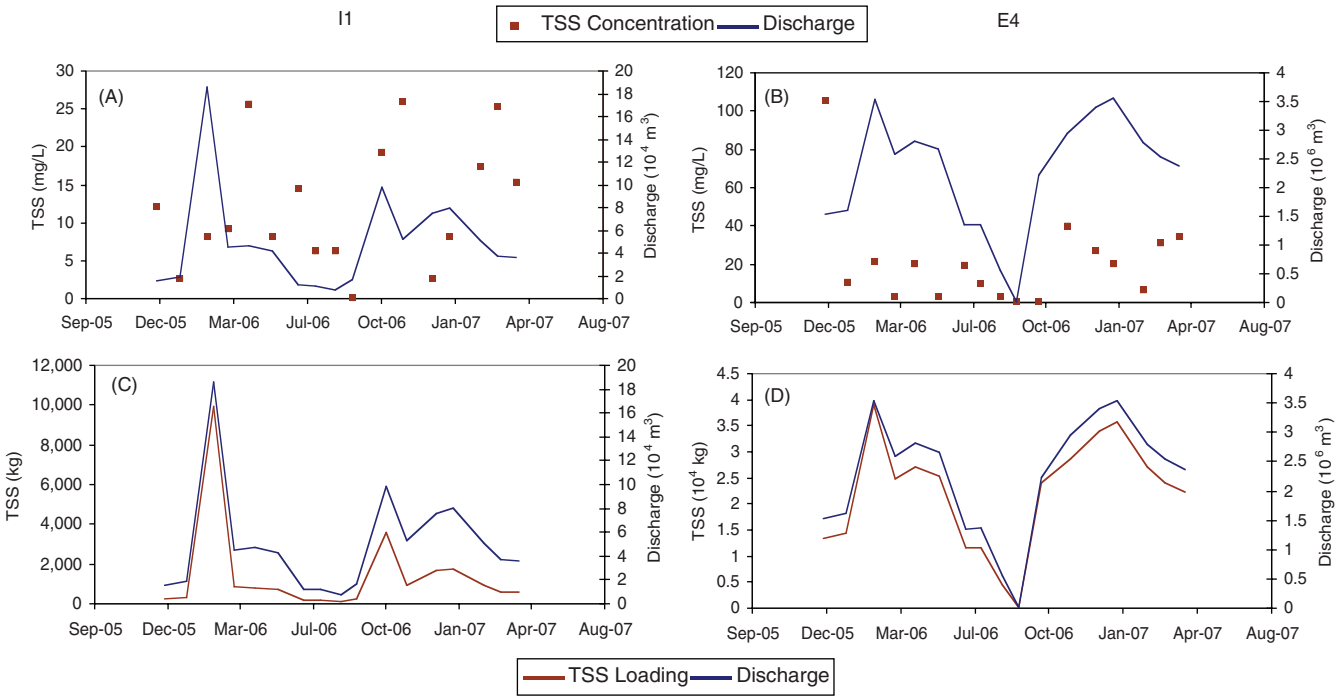

Figure 6—Comparisons of monthly discharge with monthly (A) and (B) total suspended solid (TSS) concentrations and (C) and (D) loadings for sites I1 and E4, the smallest and largest drainage areas, over the 17-month study period.

by characteristics such as antecedent moisture conditions and rainfall intensity/duration, would then have a lesser effect on TSS loading. Seasonal patterns of loading rates closely follow discharge, with high levels in the wet winter months and low levels in the drier summer. Site E5 appears to have higher rates of loading than all other sites except E4 in the winter.

As with suspended solids, monthly average TDS loads for all sites clearly followed the pattern of streamflow (fig. 7). Over the 17-month study period, total mass export of dissolved solids from the watershed was about 10 times higher than that of suspended solids. As the watershed is considered impaired by the USEPA for high TDS concentrations, further research on the complex hydrological processes present in the watershed, is needed to better determine the source of dissolved solids present in the stream.

Mean monthly TSS flux from the effective watershed outlet at site E4 was 0.8 kg/ha, increasing to 4.5 kg/ha at site I1. Site E2, the lowest monitoring location on Turkey Creek before draining into Flat Creek, had the lowest flux at 0.7 kg/ha/month. The higher discharge of Flat Creek at site E4 also carries a higher sediment flux than the input from Turkey Creek, even though it drains a larger area. Although these average fluxes cover two wet seasons and one dry season in the 17 months analyzed, precipitation was also 26 percent below normal for the study period, so fluxes may not be far from mean monthly value from 1 year with normal precipitation.

Patric and others (1984) compared sediment yields across the United States, and average annual yields for the eastern region were much greater than in Flat Creek, with 0.074 ton per acre (166 kg/ha) and 0.158 ton per acre (354 kg/ha) in watersheds less than and greater than 2 square miles (5.2 km^2), respectively. E2 (8.3 kg/ha/year) and E4 (9.0 kg/ha/year) were even lower than the lowest reported range of 0.01 ton per acre per year (22.4 kg/ha/year). Western regions in the study showed similar sediment yields, with only Pacific Coast forests showing significantly higher values (0.02 to 49.90 kg/ha/year). Due to the watershed hydrologic and geomorphic characteristics, forested land in the southeastern Coastal Plain appears to have among the lowest sediment yields in the United States.

CONCLUSIONS

Headwater streamflow in this low-gradient forested watershed was highly variable, from intermittent/no-flow conditions in the late summer, to high-volume overbank conditions in the winter season. Transitioning from the headwater streams to the watershed outlet, stream hydrologic response and streamflow variability decreased. Headwater response to storm events was quick, while hydrographs of increasing drainage area had longer lag times and more gradual falling limb recessions. The flat slopes, low-permeable soils, and beaver/debris dams reduced peak discharges, later releasing the stored water to extend streamflow during dry periods. These effects were compounded, and are most prevalently shown, at the watershed outlet. These physical watershed characteristics impacting the stream hydrology are the major influence on sediment loading in Flat Creek.

Suspended and dissolved solid concentrations during baseflow showed little seasonal variation. Mass loadings were

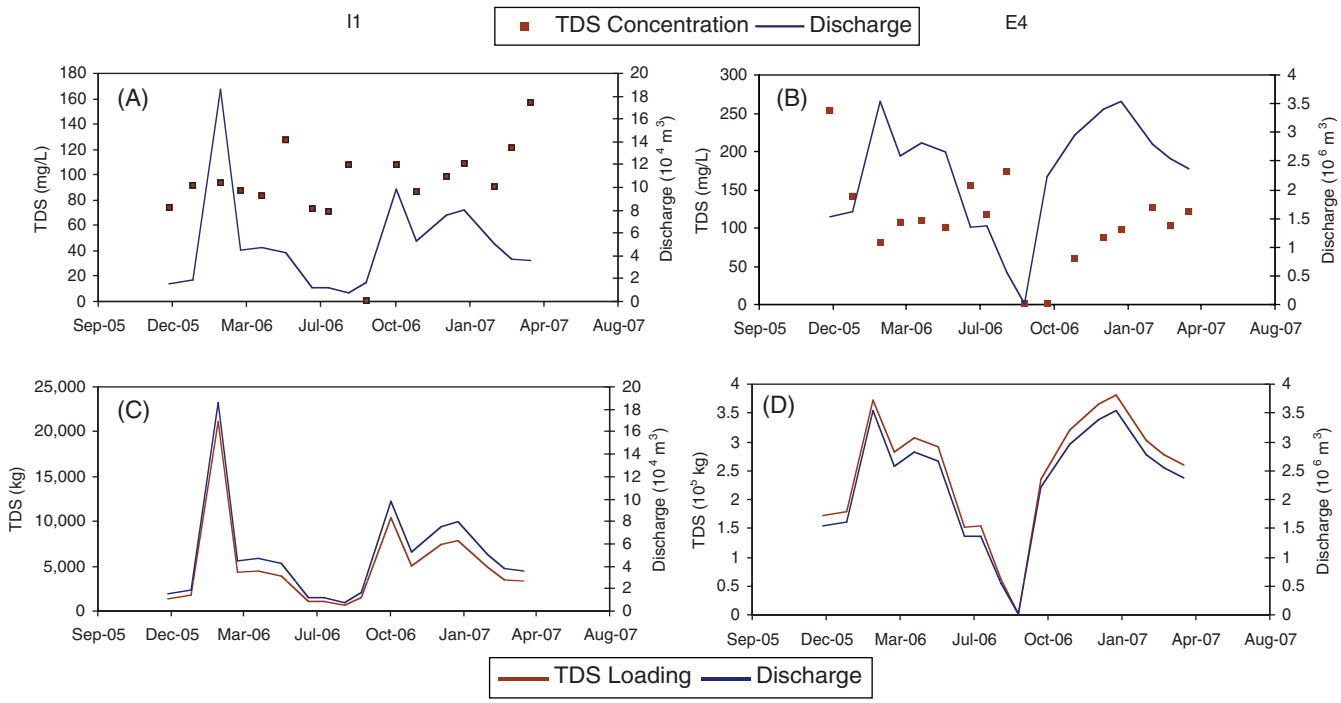

Figure 7—Comparisons of monthly discharge with monthly (A) and (B) total dissolved solid (TDS) concentrations and (C) and (D) loadings for sites I1 and E4, the smallest and largest drainage areas, over the 17-month study period.

influenced more by the discharge regime than fluctuations in concentration. Sediment yield from the watershed was low, indicating that sediment transport in low-gradient headwaters is highly retentive. As most of the land use in Flat Creek is commercial pine plantation, the attenuated runoff decreases erosion susceptibility from harvesting activities. However, caution must also be taken and forestry best management practices implemented as harvest sites can become quickly saturated following precipitation events, creating the potential for direct surface runoff and sediment delivery to streams.

ACKNOWLEDGMENTS

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LITERATURE CITED

Amatya, D.M.; Trettin, C.C.; Skaggs, R.W. [and others]. 2005. Five hydrologic studies conducted by or in cooperation with the Center for Forested Wetlands Research. Res. Pap. SRS-40. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 22 p.

Flather, C.H.; Joyce, L.A.; King, R.M. 1990. Linking multiple resource analyses to land use and timber management: application and error considerations. In: LaBau, V.J.; Cunia, T., eds. State-of-the-art methodology of forest inventory: a symposium proceedings. Gen. Tech. Rep. PNW-GTR-263. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station: 478–486.

Hupp, C.R. 2000. Hydrology, geomorphology and vegetation of Coastal Plain rivers in the South-eastern USA. Hydrological Processes. 14(16–17): 2991–3010.

Ismail, W.R. 2000. The hydrology and sediment yield of the Sungai Air Terjun catchment, Penang Hill, Malaysia. Hydrological Sciences Journal. 45(6): 897–910.

Jones, J.A.; Swanson, F.J.; Wemple, B.C.; Snyder, K.U. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology. 14: 76–85.

Leopold, L.B.; Wolman, M.G.; Miller, J.P. 1964. Fluvial processes in geomorphology. San Francisco: W.H. Freeman and Company: 142.

National Climatic Data Center. 2002. Climatography of the United States no. 81: 16 Louisiana. Asheville, NC: National Oceanic and Atmospheric Administration, National Climatic Data Center. http://cdo.ncdc.noaa.gov/climatenormals/clim81/LAnorm.pdf. [Date accessed: July 28, 2011].

Patric, J.H.; Evans, J.O.; Helvey, J.D. 1984. Summary of sediment yield data from forested land in the United States. Journal of Forestry. 82(2): 101–104.

SAS Institute Inc. 1996. SAS/STAT user's guide. Version 6.12. 4th ed. Cary, NC: SAS Institute Inc.

FOREST LANDOWNER ATTITUDES TOWARD SHORTLEAF PINE RESTORATION: RESULTS OF NINE MISSOURI FOCUS GROUPS

Heather Scroggins, David Gwaze, and Michele Baumer[1]

Abstract—Shortleaf pine (*Pinus echinata* Mill.) once occurred on 6.6 million acres in the State of Missouri, but by the 1970s only 400,000 acres had shortleaf pine. Since 1935 seeds and seedlings have been sold to the public in the State, as well as planted on public lands, for habitat improvement, timber production, and increasing biodiversity. In Missouri, as in many other States, the majority of forest land (approximately 85 percent) is privately owned. In essence, if shortleaf pine restoration efforts are to succeed, they must do so on private land. In 2007 and 2008 a series of nine focus groups was conducted in the historic shortleaf pine range of Missouri. The focus groups ranged from approximately 90 to 120 minutes in length, and had anywhere from 6 to 14 participants. Motivations for growing and managing shortleaf pine were varied, and included ease of production, aesthetics, and wildlife habitat goals, as well as a more general restoration ethic. Economic incentives included sales of timber, increased property values, possible improvements in the growth of more valued species like walnuts, and decreased heating and cooling costs. Many focus group participants alluded to the suitability and hardiness of shortleaf pine as a solution to various problematic land characteristics. It would appear that educational efforts and materials should be better targeted, highlighting planting methods, ease of growth, innate suitability for local habitats, and wildlife benefits. In addition, onsite technical assistance to landowners should be continued or expanded if possible, and increased field days or farm tours should be considered.

INTRODUCTION

Shortleaf pine (*Pinus echinata* Mill.) once occurred on 6.6 million acres in the State of Missouri, but by the 1970s only 400,000 acres had shortleaf pine. Extensive logging from 1880 to 1920, frequent wildfires, and overgrazing have all been suggested as causes of shortleaf population decline. Since 1935 seeds and seedlings have been sold to the public in the State, as well as planted on public lands, for habitat improvement, timber production, and increasing biodiversity.

In Missouri, as in many other States, the majority of forest land (approximately 85 percent), is privately owned. In essence, if shortleaf pine restoration efforts are to succeed, they must do so on private land. To that end, the objectives of this study were to (1) gain an understanding of forest landowners' motivations for managing trees, particularly shortleaf pine; (2) appreciate the challenges and needs of forest landowners in the historic shortleaf pine range; and (3) understand how the Missouri Department of Conservation (MDC) can assist forest landowners.

METHODS

In 2007 and 2008 a series of nine focus groups was conducted in the historic shortleaf pine range of Missouri. The majority of the focus group participants were recruited from the George O. White State Forest Nursery customer database. An attempt was made to prescreen potential participants based on whether they had purchased any shortleaf pine seedlings or seed, and further checks were made while issuing invitations over the telephone. While this resulted in a somewhat imperfect split, to the extent practical, groups were formed based on the presence or absence of shortleaf pine interest. Four focus groups were held with landowners who had done some management for shortleaf pine or who had shortleaf pine naturally occurring on their land. An additional focus group was held with landowners who had a strong commercial interest in pine. The remaining four focus groups were with landowners who did not manage for shortleaf pine.

The focus groups ranged from approximately 90 to 120 minutes in length and had anywhere from 6 to 14 participants. Participants were mailed a check for $50 to compensate them for their time and travel expenses to the focus group location. All nine focus groups were audiotaped, fully transcribed, and thematically analyzed for content. While the focus group protocols were similar, the makeup of the groups meant that some questions would vary. However, participants in each group were shown a large photo of a native shortleaf pine forest, and each protocol began with general questions about trees and their management, then asked about motivations, and finally challenges related to tree management.

FINDINGS

Motivations

Motivations for growing and managing shortleaf pine were varied, and included ease of production, aesthetics and wildlife habitat goals, as well as a more general restoration ethic. Other than some regional differences in the prevalence of economic importance, motivations were similar across all groups. Some of the more commonly stated motivations are highlighted here.

Economic incentives included sales of timber, increased property values, possible improvements in the growth of more valued species like walnuts, and decreased heating and cooling costs. Some mention was also made of potential carbon credit sales at some future date.

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Many focus group participants alluded to the suitability and hardiness of shortleaf pine as a solution to various problematic land characteristics. Somewhat related to this idea of innate suitability of pine was a tendency to go back to the idea of a preference for shortleaf pine because it was "supposed to be" or it "used to be" in the area or even on the piece of property in question. In all of the focus groups there was discussion of providing shortleaf pine to future generations. Many participants were quick to point out that there was no economic benefit for them, but there might be at some future time for their children or grandchildren. Other participants had more purely ecological or restoration reasons for bequeathing their pine stands, closely related to the idea of "used to be/supposed to be" as previously discussed.

There were many different recreation-related reasons offered for shortleaf pine management. Not surprisingly, a majority revolved around the hunting of species such as deer, turkey, and quail. In addition, people mentioned nonconsumptive benefits of those same species, as well as improved hiking, horse riding, and other benefits. Aesthetic reasons for growing and managing shortleaf pine were stated by a majority of focus group participants, and ranged from the attractiveness of the tree itself, to its evergreen nature, to the smell, and sound, and beyond. For a multitude of reasons, participants found shortleaf pine attractive, and many were emotionally attached to it for the same reasons.

Challenges and Needs

For the most part focus group participants did not have serious problems when it came to managing for shortleaf pine. While certain problems, such as rainfall, were out of MDC control, there were requests for education and labor assistance, as well as some minor issues with the State forest nursery, that can be addressed. Suggestions on labor and educational assistance that would be useful were varied, ranging from simple planting instructions, to field days and help with tax preparation. In some groups a small number of participants had experienced issues with the State forest nursery, including the availability of seeds and timing of receiving seedlings.

Most participants who were currently growing shortleaf pine said that stronger markets would not necessarily affect their future management plans, which was not surprising given their current high level of interest despite lagging markets. One individual also commented that he did not like the damage caused to his land by logging contractors, making him hesitant about harvesting for profit. As might be expected, the group composed only of people who derived at least some income from shortleaf pine placed more emphasis on markets. A reliable market for saw logs was seen by many members of that group as the biggest obstacle to increasing their involvement in shortleaf pine management.

Those who did not grow shortleaf pine tended to indicate that they were unfamiliar with the requirements of shortleaf pine, as well as what, if any, benefits it provided for wildlife. These participants in particular voiced a need for educational brochures and training about shortleaf pine. They also indicated that technical assistance and equipment loan or rental programs might encourage them to consider becoming involved in shortleaf pine restoration on their land.

There were some suggestions on how MDC could help landowners who were already managing for shortleaf pine, as well as encourage others to grow it. For the most part suggestions centered around the available quantities of seed and seedlings, as well as when those were available to the public. One participant in the economic group did request help from MDC in developing a marketable use for off-fall. Participants also broached the idea of demonstration areas or farm tours. Several participants had benefited from onsite assistance from MDC staff and commented favorably on the assistance given to private landowners.

MANAGEMENT IMPLICATIONS

Many participants have a strong affinity for shortleaf pine and are strongly motivated to grow it for widely diverse reasons. While most do not face insurmountable challenges, an increase in the availability and variety of educational materials may enhance efforts to encourage shortleaf pine restoration on private lands. It would appear that educational efforts and materials should be better targeted, highlighting planting methods, ease of growth, innate suitability for local habitats, and wildlife benefits. In addition, onsite technical assistance to landowners should be continued or expanded if possible, and increased field days or farm tours should be considered.

LAND CLASSIFICATION OF THE STANDING STONE STATE FOREST AND STATE PARK ON THE EASTERN HIGHLAND RIM IN TENNESSEE: THE INTERACTION OF GEOLOGY, TOPOGRAPHY, AND SOILS

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Abstract—This paper summarizes the application of a land classification system developed by the senior author to the Standing Stone State Forest and State Park (SSSF&SP) on the Eastern Highland Rim. Landtypes are the most detailed level in the hierarchical system and represent distinct units of the landscape (mapped at a scale of 1:24,000) as defined by climate, geology, soils, topography, and vegetation. The area is highly dissected with local relief of about 1,000 feet. Mississippian and Ordovician strata are essentially level bedded; defining elevations were assigned. Suites of soils are common to the nine strata, and a group of landtypes was defined for each geologic strata/soils combination. Each of the 19 landtypes is described in terms of 9 elements. Additional information includes species suitability, site productivity, and operability for management activities. The maps aid the delineation of stands, streamside management zones, and "conservation" and other special use areas; the location of rare, threatened and endangered (RT&E) species; the design of harvests; and the modeling of future forest conditions. The landtypes are an integral element in modeling wildlife habitat, in siting game food plots, and planning other wildlife management activities. The maps are excellent training devices and extremely useful in explaining management plans to legislators and the public.

INTRODUCTION

The Tennessee Division of Forestry (TDF) has adopted a land classification system developed by the senior author (Smalley 1991b) as the basic theme of information for the management of its 15 State forests (SF). At least one SF occurs in each of eight physiographic provinces—Southern Appalachian Mountains, Ridge and Valley, Cumberland Plateau, Eastern Highland Rim, Nashville Basin, Western Highland Rim, Upper Coastal Plain, and Mississippi Embayment. In this paper we describe how the system was applied to the Standing Stone State Forest and State Park (SSSF&SP) located on the Eastern Highland Rim.

THE LAND CLASSIFICATION SYSTEM

Initially, the land classification system was developed for the 29 million acres of the Cumberland Plateau and Highland Rim/Pennyroyal Physiographic Provinces in parts of Alabama, Georgia, Tennessee, Kentucky, and Virginia (Smalley 1986, Smalley and others 1996). The system was adapted from Wertz and Arnold's (1975) Land System Inventory. The system can best be described as a process of successive stratifications of the landscape. Stratifications are based on the interactions and controlling influences of ecosystem components—physiography, climate, geology, soils, topography, and vegetation. Macroclimate does not vary much across both physiographic provinces, but microclimate varies because of local relief. Since the current species composition and structure of rim and plateau forests was more a function of repeated disturbances than an indication of succession and site potential, vegetation was relegated to a minor role in the development of the land classification system (Delcourt 1979). Application of the system to other physiographic provinces represents an extension of the original concept (Smalley 1991a).

EASTERN HIGHLAND RIM

The Eastern Highland Rim (Pennyroyal in Kentucky) (EHR) region covers about 11,440 square miles extending from Louisville, KY, through Tennessee, to Russellville, AL (Smalley 1983). In Tennessee the EHR includes the upland surrounding the Nashville Basin on the east and the knobby transition from the rim to the basin (Edwards and others 1974, Fenneman 1938, Springer and Elder 1980). It is bounded on the east by the ragged western escarpment of the Mid-Cumberland Plateau (Smalley 1982). The division between the EHR and the Western Highland Rim (Smalley 1980) is somewhat arbitrary, defined mostly on the basis of soils.

Compared with the National Hierarchical Framework of Ecological Units (Avers and others 1993, Bailey and others 1994, Cleland and others 1997), the EHR is equivalent to the Eastern Karst Plain Subsection (223Eb) of the Interior Low Plateau-Highland Rim Section (223E) of the Central Interior Broadleaf Forest Province (223) (Cleland and others 2007).

SPECIFIC LOCATION

The SSSF&SP is in Overton and Clay Counties (36°27' N, 85°27' W) along the western edge of the Eastern Highland Rim. It falls into two subregions—the Highland Rim Plateau and the Transition to the Nashville Basin (Smalley 1983) (fig. 1). Two land type associations are represented—LTA-A Strongly Dissected Plateau and LTA-E Tennessee Knobs.

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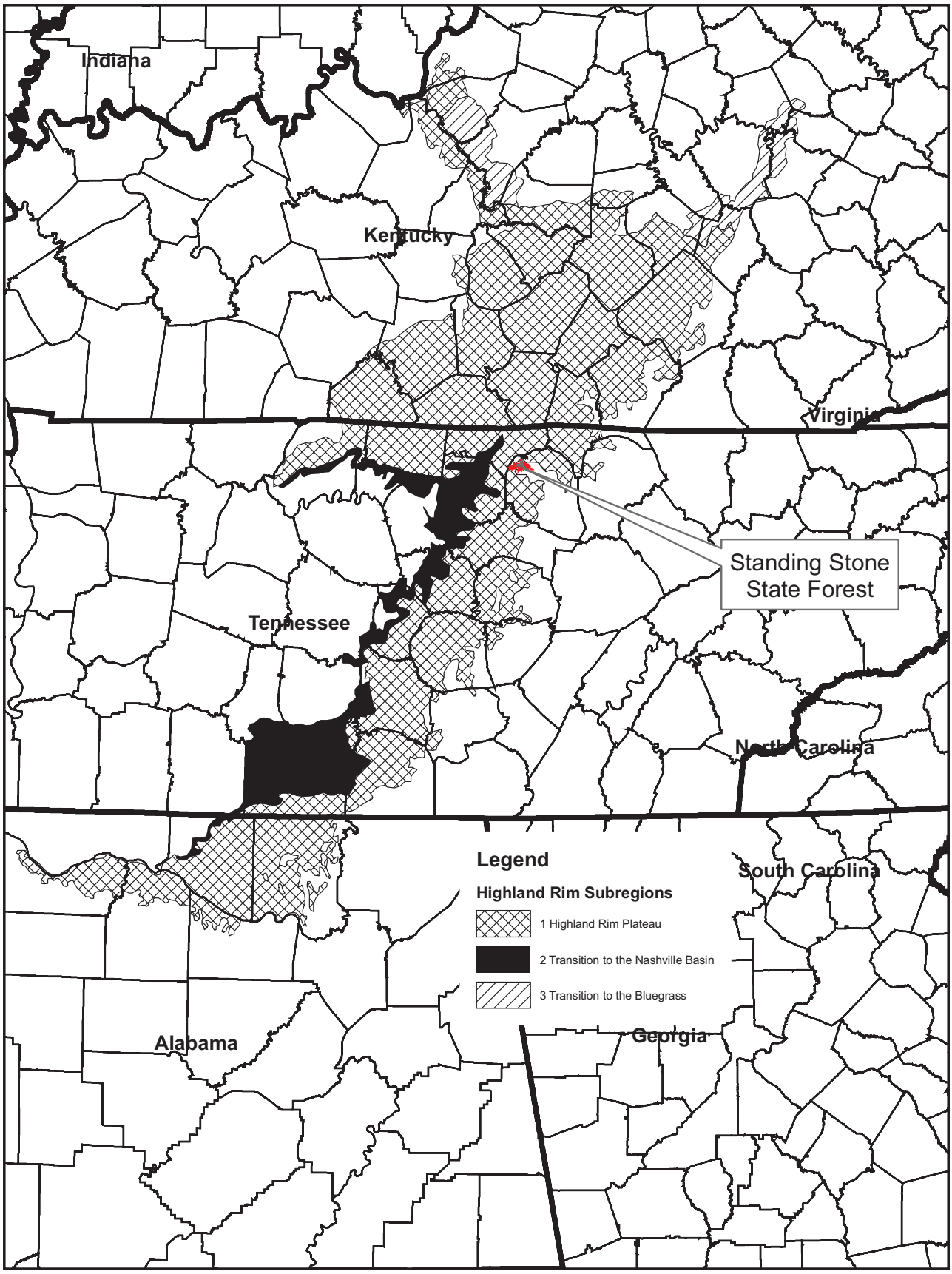

Figure 1—Location of Standing Stone State Forest and State Park in relation to the subregions of the Eastern Highland Rim.

The SSSF&SP lies between the communities of Hilham on the south, Timothy on the north, and Allons on the east. Livingston, the county seat of Overton County, is about 5 miles to the southeast and Celina, the county seat of Clay County, is about 6 miles to the northwest.

THE FOREST

The SSSF&SP consists of acreage purchased by the Resettlement Administration of the Federal Government beginning in 1935. In 1939 the U.S. Department of Agriculture leased the acquired land to the State of Tennessee, Department of Conservation, Division of Parks. The land was deeded to the State in 1955. In 1961, by agreement, the administration of 8,490 acres was transferred to the Division of Forestry (Standing Stone State Forest) and 855 acres (Standing Stone State Park) was retained by the Division of Parks. The cleared lands were eroded due to extensive row cropping and poor farming practices. The forests had been extensively logged (mostly high-grading) and burned. The park area was developed by the Civilian Conservation Corps in the 1930s and early 1940s. Recent surveys show the forest is 8,445 acres in extent, and the park is 865 acres for a total of 9,310 acres. Two in-holdings total 136 acres. The SSSF&SP occur on two U.S. Geological Survey quadrangle maps: Hilham and Livingston. The gross area mapped was 22,247 acres; 19,627 (88 percent) occurs on the Hilham quad and 2,620 acres (12 percent) occurs on the Livingston quad.

CHARACTERISTICS OF THE FOREST AND PARK

Geology

Stratigraphy was obtained from the geology maps for the Hilham and Livingston quads (scale 1: 24,000) (Wilson 1968, Wilson and Colvin 1968). These strata are essentially level-bedded, and defining elevations can be assigned (fig. 2). Most of the strata are of Mississippian age (estimated ≥325 million years BP). The Pennington Formation and Bangor Limestone (mostly limestones with some shale) occupy the highest parts of the landscape (east side of SSSF&SP) on Reynolds Mountain above an elevation of 1,440 feet. The nearly flat-to-rolling terrain in and around Allons on the Livingston quad and the higher ridges (mountains) on the Hilham quad are capped with the Hartselle Formation (primarily sandstone), locally known as Brotherton Bench. Elevation is between 1,300 and 1,440 feet. Below 1,300 feet is the Monteagle Limestone, and below 1,110 feet is the St. Louis Limestone and Warsaw Formation. Topography over these three strata is undulating to rolling combinations of ridges and slopes. The Fort Payne Formation occurs between 1,100 and 900 feet. Mill Creek (Hilham quad) has carved into the Leipers and Catheys Formations of Ordovician age as far upstream as the dam forming Standing Stone Lake. These two formations consist of calcarenite, some phosphate pellets, and fine-grained limestone (argillaceous, nodular, and shaly). These Ordovician rocks are visible only in the bed of Mill Creek; elsewhere they are covered with alluvium.

Topography and Drainage

The highly dissected nature of the area results in a local relief of slightly over 1,000 feet (fig. 2). Elevation of Reynolds Mountain just east of the forest boundary is 1,620 feet. Goodpasture, Wilson, and Cooper Mountains exceed 1,400 feet. Landforms range from broad ridges with rolling sideslopes to very steep lower sideslopes. Bottoms are fairly broad. The area is drained by Mill Creek and Right Fork and their tributaries. These streams empty into the Cumberland River (Cordell Hull Lake—pool level is 504 feet) near Butler's Landing in Clay County about 56 miles upstream from Cordell Hull Lock and Dam. Standing Stone Lake, an impoundment on Mill Creek has a pool level of 726 feet. The bottom along Mill Creek below the dam has an elevation of about 600 feet. Slopes, particularly those over the Fort Payne formation, are very steep. Sinkholes, some quite large and deep, are common.

Soils

The Tennessee General Soil Map (Scale 1: 750,000) (Springer and Elder 1980) shows the SSSF&SP to be in general soil association H21 (Bouldin-Rock outcrop-Ramsey) which is equivalent to land type association C-Strongly Dissected Margins and Sides of the Cumberland Plateau. This anomaly happens because soils formed from the Hartselle Formation are similar to soils occurring on the Cumberland Plateau. In reality the ragged western escarpment of the Cumberland Plateau lies several miles to the east. As stated earlier, the SSSF&SP is situated on the Eastern Highland Rim at the transition to the Nashville Basin.

Five soil associations are common to this geologic diverse area (McCowan 2005, Krantz and McCowan 2006). The Christian-Sengtown association consists of deep and very deep, rolling-to-steep well-drained soils formed in residuum from cherty limestone. Minor soils are Talbott, Minvale, and Waynesboro. The Nella-Talbott-and similar soils association consists of very deep and moderately deep, rolling-to-very steep well-drained soils formed in colluvium and residuum from limestone. Minor soils are Bouldin and Etowah. The Gilpin-Shelocta-and similar soils association consists of moderately deep and deep, rolling-to-very steep well-drained soils formed in residuum from cherty limestone and siltstone. Minor soils are Bouldin, Ramsey, and Lily. The Lonewood-Clarkrange association consists of deep and very deep, undulating-to-rolling, well-drained, and moderately well-drained soil formed in loess and residuum from sandstone. Minor soils are Lily and Ramsey. The Garmon-Newbern association consists of moderately deep-to-shallow, steep to very steep, well-drained, and somewhat excessively drained soils formed in residuum from calcareous shale. Minor soils are Humphreys, Ocana, and Christian. These soils represent four orders—Alfisols, Inceptisols, Mollisols, and Ultisols. The taxonomic classification of these soils is shown in table 1.

Climate

Surface weather data were obtained from the National Oceanic and Atmospheric Administration's (NOAA) National Climatic Data Center for weather station 405332, Livingston, TN, from 1971 to 2000 (National Climatic Data Center 2004). Average annual precipitation is about

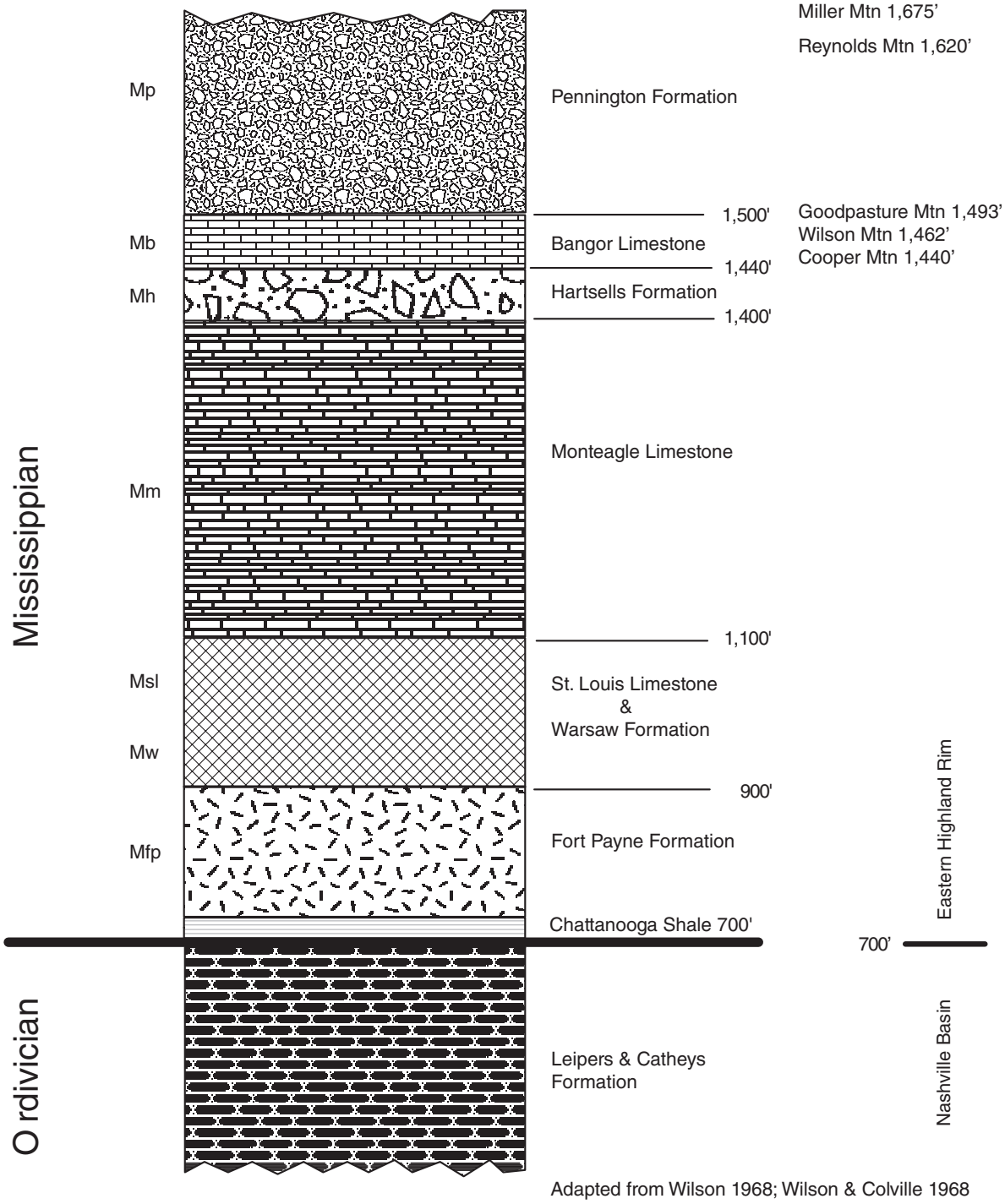

Figure 2_Stratigraphy of the Eastern Highland Rim in the vicinity of Standing Stone State Forest and State Park.

52.59 inches of which 34.41 inches fall in the April through October growing season. The average snowfall is 9.3 inches. The average winter temperature is 40.4 °F and the average daily minimum is 28.8 °F. The record low is –25 °F recorded on January 21, 1985. The summer average daily temperature is 73.1 °F; the average daily maximum is 85.1 °F.

Humidity, sunshine, and wind data are reported in the Clay County soil survey (Krantz and McCowan 2006). Average relative humidity in midafternoon is 57 percent. Humidity is higher at night and averages about 84 percent at dawn. Cloud-free days occur 64 percent of the time in summer and 43 percent in the winter. The prevailing wind is from the south. Average wind speed is highest, about 10 miles per hour from December to April.

Table 1—Taxonomic classification of soils common to Standing Stone State Forest and State Park

Order/Suborder/great group/series	Family/taxonomic class
Alfisols	
Udalfs	
Hapludalfs	
Talbott	Fine, mixed, semiactive, thermic Typic Hapludalfs
Paleudalfs	
Sengtown	Fine, mixed, semiactive, thermic Typic Paleudalfs
Inceptisols	
Aquepts	
Endoaquepts	
Melvin	Fine-silty, mixed, active, nonacid, mesic Fluvaquentic, Emdoaquepts
Udepts	
Eutrudepts	
Sullivan	Fine-loamy, siliceous, active, thermic Dystric Fluventic Eutrudepts
Ocana	Fine-loamy, mixed, active, thermic Dystric Eutrudepts
Hamblen	Fine-loamy, siliceous, semiactive, thermic Fluvaquentic Eutrudepts
Garmon	Fine-loamy, mixed, semiactive, mesic Dystric Eutrudepts
Newbern	Loamy, mixed, active, mesic Lithic Eutrudepts
Ochrepts	
Dystrudepts	
Ramsey	Loamy, siliceous, subactive, mesic Lithic Dystrudepts
Mollisols	
Rendolls	
Rendolls	
Gladeville	Clayey-skeletal, mixed, active, thermic Lithic Haprendolls
Udolls	
Hapludolls	
Barfield	Clayey, mixed, active, thermic Lithic Hapludolls
Ultisols	
Udults	
Fragiudults	
Clarkrange	Fine-silty, siliceous, semiactive, mesic, Typic Fragiudults
Hapludults	
Alticrest	Coarse-loamy, siliceous, semiactive, mesic Typic Hapludults
Lily	Fine-loamy, siliceous, semiactive, mesic Typic Hapludults
Lonewood	Fine-loamy, siliceous, semiactive, mesic Typic Hapludults
Christian	Fine, mixed, semiactive, mesic Typic Hapludults
Sequoia	Clayey, mixed, active, mesic Typic Hapludults
Paleudults	
Bouldin	Loamy-skeletal, siliceous, subactive, mesic, Typic Paleudults
Nella	Fine-loamy, siliceous, semiactive, thermic Typic Paleudults
Minvale	Fine-loamy, siliceous, subactive, thermic Typic Paleudults
Waynesboro	Fine, kaolinitic, semiactive, thermic Typic Paleudults

Source: Soil Survey Staff (2008). Official soil series descriptions. http://ortho.ftw.nrcs.usda.gov/ogi-bin/osdname.cgi. [Date accessed: September 15, 2008, verified November 20, 2008].

Vegetation

The Eastern Highland Rim forests are part of the broad oak-hickory forests (western mesophytic/oak-hickory forests) region described by Bryant and others (1993). Delcourt and Delcourt (2004) concluded that oak-hickory forests are prevalent on the more xeric sites with more mesic species, similar to those in the mixed mesophytic forests further east on the Cumberland Plateau escarpment and in the Cumberland Mountains, and occur on cool slopes and in the gorges along the escarpment into the Nashville (central) Basin in Tennessee. Species distribution on the SSSF&SP is governed by slope, aspect, soil depth, soil moisture, and geologic substrate. Considerable acreage is occupied by 50- to 70-year-old forests resulting from old-field succession.

LAND TYPES

Nineteen land types were identified and mapped on the extended area of SSSF&SP (table 2). Because of the close association of geology and soils, a group of land types was defined for each geologic strata/soils combination. Land types were numbered sequentially according to elevation—highest to the lowest.

Table 2—Land types of the Eastern Highland Rim occurring on Standing Stone State Forest and State Park

Land type number	Land type name	Acres	Percent of the total acres
	Land types over the Pennington Formation and the Bangor Limestone		
1	Broad limestone ridges	22	0.1
2	North-facing shaly limestone slopes	68	0.3
3	South-facing shaly limestone slopes	43	0.2
	Land types over the Hartselles Formation		
4	Undulating sandstone uplands	811	3.6
5	Narrow to moderately broad sandstone ridges	228	1.0
6	Shallow soils and sandstone outcrops	44	0.2
7	Depressions, flats, and sinkholes	39	0.2
	Land types over the Monteagle Limestone		
8	Narrow to moderately broad limestone ridges and spurs	1,286	5.8
9	North-facing limestone slopes	2,042	9.2
10	South-facing limestone slopes	994	4.5
11	Depressions and sinkholes	281	1.3
	Land types over the St. Louis Limestone and Warsaw Formation		
11	Depressions and sinkholes (see above)		
12	Narrow to broad limestone ridges and knobs	6,987	31.4
13	North-facing limestone slopes	1,675	7.5
14	South-facing limestone slopes	1,409	6.3
	Land types over the Ft. Payne Formation		
15	North-facing shaly slopes	2,632	11.8
16	South-facing shaly slopes	2,732	12.3
	Stream and creek bottoms		
17	Stream and creek bottoms with good drainage	837	3.8
18	Stream and creek bottoms with poor drainage	25	0.1
	Miscellaneous landforms		
19	Water—lakes, ponds, and streams	90	0.4
	Total	22,245	100.0

Land types 1, 2, and 3 are common to the Pennington Formation and Bangor Limestone. Land types 4, 5, 6, and 7 are common to the Hartselle Formation. Land types 8, 9, 10, and 11 are common to the Monteagle Limestone. Land types 11, 12, 13, and 14 are common to the St. Louis Limestone and Warsaw Formation. Land types 15 and 16 are common to the Fort Payne Chert. Land types 17 and 18 are creek and stream bottoms with good and poor drainage, respectively. All bodies of water (ponds, lakes, and streams) are assigned LT-19. Descriptions of the two most extensive land types are shown in table 3.

APPLICATIONS OF THE LAND CLASSIFICATION SYSTEM

Earlier research on the Cumberland Plateau showed that the land classification system divided the Prentice Cooper State Forest landscape into distinct ecological units with relative discreet plant communities (Arnold and others 1996). Additionally, the system grouped soils on the Catoosa Wildlife Management Area into landforms units having relative homogeneous chemical and physical properties (Hammer and others 1987). The utility of the system for all six SFs on the Cumberland Plateau has been reported (Smalley and others 2006). Plant community-landform relations have been studied at two locations on the Western Highland Rim (Clatterbuck 1996, Wheat and Dimmick 1987).

Cleland and others (2007) listed ecosystem mapping, resource assessments, environmental analyses, watershed analyses, desired future conditions, resource management, and monitoring as uses of the National Hierarchical Framework of Ecological Units system. These uses also apply to Smalley's system. Currently, TDF is focusing on ecosystem delineation, resource assessment, desired future conditions, and resource management and monitoring (Smalley and others 2006). Much more data need to be obtained before meaningful environmental and watershed analyses can be made.

Current Uses

Each SF is divided into compartments consisting of groups of stands averaging approximately 1,000 ± acres. Compartment plans are written to meet multiple use, broad landscape scale goals. The individual stand silviculture prescriptions are developed to be congruent with the overall compartment and forest level goals.

Stand delineation—Stands (silvicultural management units) are delineated at the same scale as the landtype maps (1:24,000). They have similar forest type and productivity and may range in size from 5 to 40 acres with the average being 23 acres. Stand delineation is the result of a combination of considerations. While the primary objective is to create

Table 3—Example of the information found in the land type descriptions

Description of Land type 12: narrow to moderately broad limestone ridges and knobs over the St. Louis Limestone and Warsaw Formation
Geographic setting: Deep to very deep, gravely silt loam to gravely clay soils on gently sloping to strongly sloping narrow to moderately broad ridges and knobs over the St. Louis Limestone and Warsaw Formation. Slope generally ranges from 5 to 12 percent on the moderately broad ridges but may be as much as 20 percent on the narrow ridges and knobs. Elevation ranges from 900 to 1,100 feet. Land types 8, 9, and 10 over the Monteagle Limestone occur above LT-12. Land types 15 and 16 over the Fort Payne formation occur below LT-12. This land type is dotted with artificial ponds, sinks, and depressions. There is no comparable land type in the guide for the Eastern Highland Rim (Smalley 1983). Land type 12 is the most extensive land type. Seventy-seven units were mapped; 75 (6,684 acres) on the Hilham quad and 2 (303 acres) on the Livingston quad. Altogether 6,987 acres were mapped constituting 31.4 percent of the total area.
Dominant soils: Christian, Sengtown, and Waynesboro. Sengtown and Waynesboro soils are very deep (>60 inches) and Christian soils are deep (40 to 60 inches). These soils developed in residuum or old alluvium from cherty limestone with some possible influence from shale and sandstone. Christian and Sengtown soils have mixed mineralogy; Waynesboro soils are kaolinitic. Sengtown and Waynesboro soils have a thermic temperature regime; Christian soils have a mesic temperature regime. Coarse fragment content ranges up to 35 percent, but usually is <15 percent.
Bedrock: shaly limestone, limestone
Depth to bedrock: ≥40 inches
Texture: usually gravely silt loam near the surface, but may lack the gravel in places; grades to gravelly loam and clay in the subsoil.
Soil drainage: well drained, moderately permeable
Relative soil water supply: medium
Soil fertility: moderate
Forest type: mixed oaks, hickories, maples, yellow-poplar, American beech, eastern redcedar, Virginia and shortleaf pines. An extensive acreage south of the forest has been cleared and is currently in pasture or hay.

Legend

Com p 03 Stands

Landtype ID, Landtype Description

1 Broad Limestone Ridges over the Pennington Formation
2 North-facing Shaly Limestone Slopes over the Pennington Formation
3 South-facing Shaly Limestone Slopes over the Pennington Formation
4 Undulating Sandstone Uplands over Hartselles Formation
5 Narrow to Moderate Broad Sandstone Ridges over Hartselles Formation
6 Shallow Soils and Sandstone Outcrops over the Hartselles Formation
7 Depressions, Flats, and Sinkholes over the Hartselles Formation
8 Narrow to Moderately Broad Limestone Ridges & Spurs over Monteagle Limestone
9 North-facing Limestone Slopes over the Monteagle Limestone
10 South-facing Limestone Slopes over the Monteagle Limestone
11 Depressions and Sinkholes in the Monteagle Limestone, St. Louis Limestone
12 Narrow to Moderately Broad Limestone Ridges & Knobs over St. Louis Limestone
13 North-facing Limestone Slopes over the St. Louis Limestone
14 South-facing Limestone Slopes over the St. Louis Limestone
15 North-facing Shaly Slopes over the Fort Payne Formation
16 South-facing Shaly Slopes over the Fort Payne Formation
17 Stream and Creek Bottoms with Good Drainage
18 Stream and Creek Bottoms with Poor Drainage
19 Water Lakes, Ponds, and Streams

Figure 3—Landtypes and delineated stands in compartment 03 on Standing Stone State Forest.

management units of uniform characteristics, many times other needs result in stand boundaries being drawn along roads to facilitate stand access or along streams to reduce stream crossings. Consequently, ridge land types (LTs-1 and 2) and upland hollows (LTs-14 and 15) may be split. Conversely, some individual units of a land type may cover 50+ acres. Stand size for various reasons is typically <40 acres and is primarily restricted to meet compartment goals, allow silviculture prescription on a stand by stand basis, accommodate physical boundaries, and meet certain standards for forest certification. Therefore, several stands may be defined within a single land type unit. Figure 3 shows delineated stands and land types. Note the close agreement between stand and land type boundaries.

An immediate benefit of the land type maps has been to reduce the time required to delineate stand boundaries. Heretofore, stand delineation required several weeks of work. With the availability of land type maps, the task has been reduced to a few days.

Management type determination is characterized by a single forest type, often an association of two or more species where hardwoods are dominant. Because of past abuses, the current forest type may not be the desired management type. The ancillary information about desired species and estimated productivity for each land type enables forest managers to formulate appropriate silvicultural strategies to achieve desired forest conditions.

Future Uses

Predictability of future forest attributes is invaluable to forest managers for making decisions that meet stated objectives and communicate management strategies. Land type information and models have been used to assess current and future forest ecological conditions (Druckenbrod and Dale 2004, Druckenbod and others 2006). Currently, the SF system is embarking on utilizing the U.S. Forest Service's Forest Vegetation Simulator (FVS) (Dixon 2002) and the Landscape Management System (LMS) (McCarter and others 1996, 1997, 1998) to predict future stand, compartment, and forest level conditions in both tabular and graphical forms. Land type information, primarily species composition and productivity, can be utilized in these models to show stand attributes over time. Graphical depiction will help communicate long-term forest management strategies and visual management issues to interest groups, such as, the general public, forest managers, forested landowners, SF users/visitors, State executive managers, and legislative representatives.

Figure 4 is a comparison of species composition and height growth of LT- 6 (low productivity) and LT-15 (high productivity) at ages 5 and 40 years after harvest using the LMS/FVS model. The species composition used for regeneration of each land type was derived from those listed in the description of land types. Site indices used in the LMS/FVS model were derived from site index estimates of these land types located on SSSF&SP. LT-6 is described as shallow soils and sandstone outcrops over the Hartselle Formation. Land type15 is described as north-facing shaly slopes over the Fort Payne Formation. Land type 6 supports poor site mixed oaks and some Virginia pine (*Pinus virginiana* Mill.). Whereas, LT-15 is a good north-facing slope that supports yellow-poplar (*Liriodendron tulipifera* L.), mixed oaks, and trends toward mixed mesophytic species. Height growth is depicted by the forest profile graphics in figure 4. The stadia lines on each side are 70 feet tall. These results illustrate the differences in initial species composition found on different land types, the change in species composition over time, and height growth for these land types.

Just by knowing the land type for each stand, one can roughly model each delineated stand in a forest and follow the stands through time. Treatments can also be applied to graphically examine visual aspects and/or management strategies. Figure 5 is a portion of Standing Stone depicting the visual character of several stands in a landscape view using LMS.

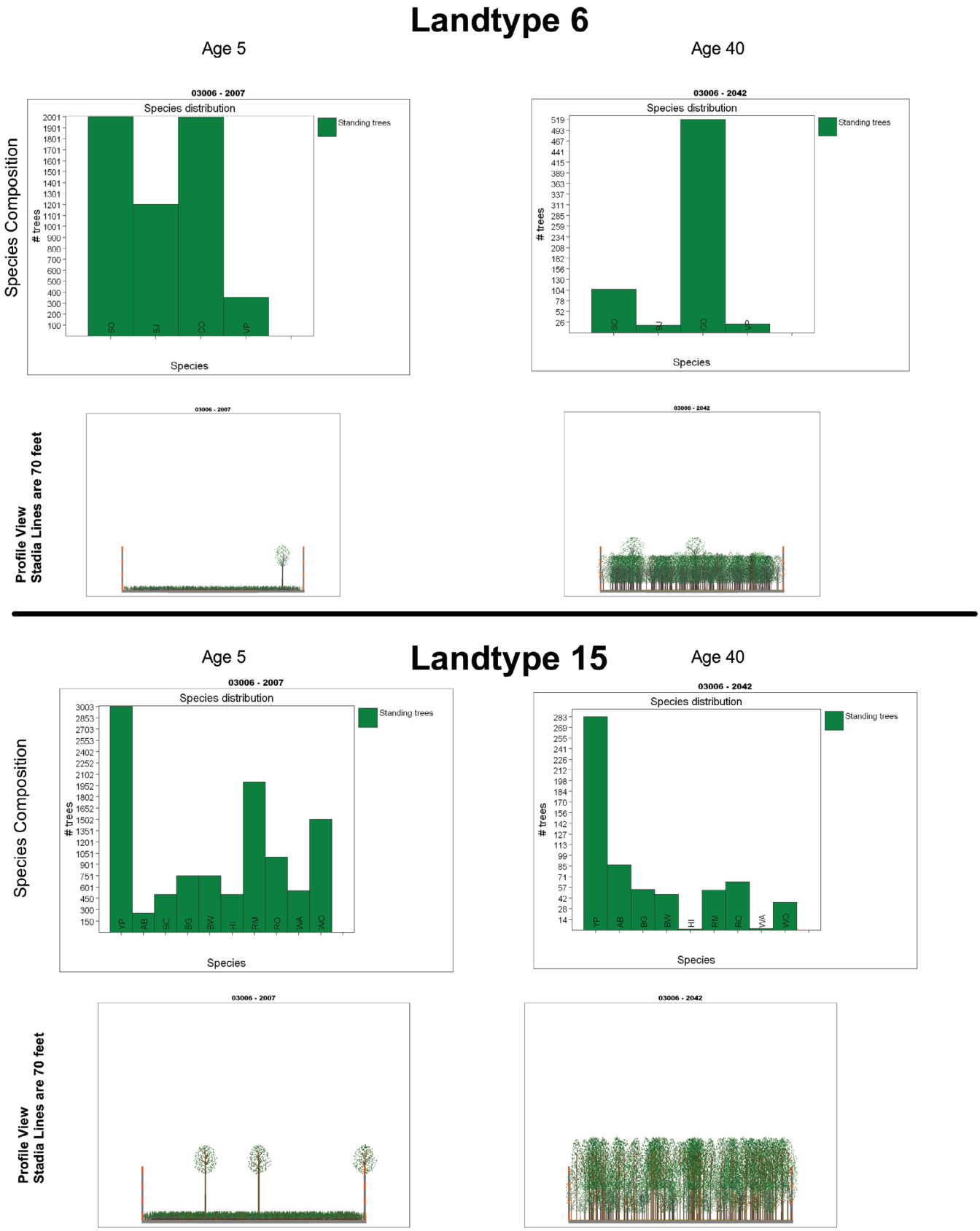

Figure 4—Comparison of species composition and height growth of LT-6 (poor site) and LT-15 (good site) at ages 5 and 40 years following harvest.

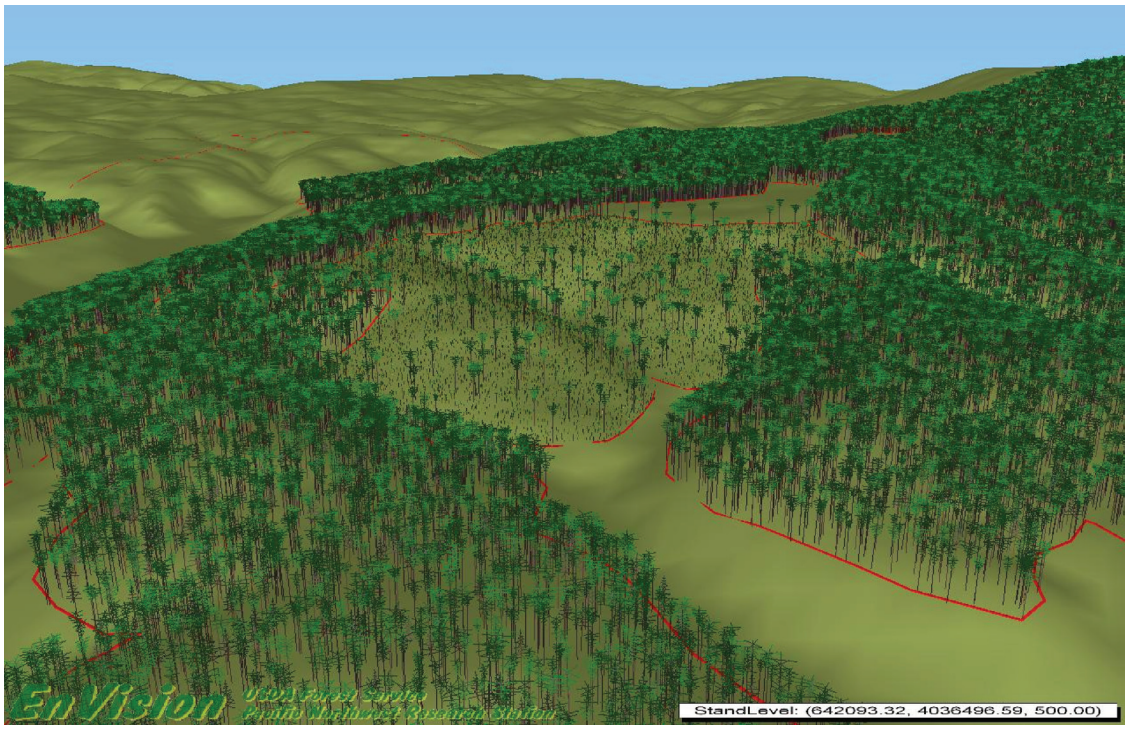

Figure 5—Graphical landscape view of a portion of Standing Stone State Forest using LMS to model stands using landtype information.

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LITERATURE CITED

Arnold, D.H.; Smalley, G.W.; Buckner, E.R. 1996. Landtype-forest community relationships: a case study on the Mid-Cumberland Plateau. Environmental Monitoring and Assessment. 39: 339–352.

Avers, P.E.; Cleland, D.; McNab, W.H. [and others]. 1993. National hierarchical framework of ecological units. Washington, DC: U.S. Department of Agriculture, Forest Service. 20 p.

Bailey, R.G.; Avers, P.E.; King, T. [and others]. 1994. Ecoregions and subregions of the United States. Washington, DC: U.S. Geological Survey. Scale 1: 750,000; colored. [Accompanied by a supplemental table of map unit descriptions compiled and edited by W.H. McNab, R.G. Bailey. Prepared for U.S. Department of Agriculture, Forest Service].

Bryant, W.S.; McComb, W.C.; Fralish, J.S. 1993. Oak-hickory forests (western mesophytic/oak-hickory forests). In: Martin, W.H.; Boyce, S.G.; Echternacht, A.C., eds. Biodiversity of the Southeastern United States: upland terrestrial communities. New York: Wiley: 143–201.

Clatterbuck, W.K. 1996. A community classification system for forest evaluation: development, validation, and extrapolation. Environmental Monitoring and Assessment. 39: 299–321.

Cleland, D.T.; Avers, P.E.; McNab, H.W. [and others]. 1997. National hierarchial framework of ecological units. In: Boyce, M.S.; Haney, A., eds. Ecosystem management: applications for sustainable forest and wildliife resources. New Haven and London, CT: Yale University Press: 181–200.

Cleland, D.T.; Freeouf, J.A.; Keys, J.E. [and others]. 2007. Ecological subregions: sections and subsections for the coterminous United States. Gen. Tech. Rep. WO-76D. [Map on CD–ROM]. [A.M. Sloan, cartographer]. Washington, DC: U.S. Department of Agriculture Forest Service. Scale 1: 3,500,000; colored. [Also available on CD–ROM consisting of GIS coverage in Environmental Systems Research Institute, Inc. (ESRI) format].

Delcourt, H.R. 1979. Late quaternary vegetation history of the Eastern Highland Rim and adjacent Cumberland Plateau of Tennessee. Ecological Monographs. 49: 255–280.

Delcourt, P.A.; Delcourt, H.R. 2004. Prehistoric Native Americans and ecological change: human ecosystems in Eastern North America since the Pleistocene. Cambridge, United Kingdom: Cambridge University Press. 216 p.

Dixon, G.E., comp. 2002. Essential FVS: a user's guide to the forest vegetation simulator. Internal Rep. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Forest Management Service Center. 193 p.

Druckenbrod, D.L.; Dale, V.H. 2004. Sustaining the landscape: a method for comparing current and desired future conditions of forest ecosystems in the North Cumberland Plateau and Mountains. ORNL/TM-2004/314. Oak Ridge, TN: Oak Ridge National Laboratory. 37 p.

Druckenbrod, D.L.; Dale, V.H.; Olsen, L.M. 2006. Comparing current and desired ecological conditions at a landscape scale in the Cumberland Plateau and Mountains, USA. Journal of Land Use Science. 2-4: 169–189. Vol. 1.

Edwards, M.J.; Elder, J.A.; Springer, M.E. 1974. The soils of the Nashville Basin. Bul. 499. [Place of publication unknown]: U.S. Department of Agriculture, Soil Conservation Service. In cooperation with University of Tennessee Agricultural Experiment Station. 125 p.

Fenneman, N.M. 1938. Physiography of Eastern United States. New York: McGraw-Hill. 418 p.

Hammer, R.D.; O'Brien, R.G.; Lewis, R.J. 1987. Temporal and spatial soil variability on three forested landtypes on the Mid-Cumberland Plateau. Soil Science Society of America Journal. 51: 1320–1326.

Krantz, G.; McCowan, C. 2006. Soil survey of Clay County, Tennessee. Washington, DC: U.S. Department of Agriculture, Natural Resources Conservation Service. 298 p. + maps.

McCarter, J.B.; Wilson, J.S.; Baker, P.J. [and others]. 1996/1997. Landscape management user's manual, Version 1.5-1.6. Landscape management project. [Place of publication unknown]: University of Washington, College of Forest Resources. 97 p.

McCarter, J.B.; Wilson, J.S.; Baker, P.J. [and others]. 1998. Landscape management through integration of existing tools and emerging technologies. Journal of Forestry. 96: 17–23.

McCowan, C. 2005. Soil survey of Overton County, Tennessee. Washington, DC: U.S. Department of Agriculture, Natural Resources Conservation Service. 424 p. + maps.

National Oceanic and Atmospheric Administration, National Climatic Data Center. 2008. World wide Web. http://cdo.ncdc.noaa.gov/climatenormals/clim20/tn/405332.pdf. [Date accessed: December].

Smalley, G.W. 1980. Classification and evaluation of forest sites on the western Highland Rim and Pennyroyal. Gen. Tech. Rep. SO-30. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 120 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.

Smalley, G.W. 1983. Classification and evaluation of forest sites on the eastern Highland Rim and Pennyroyal. Gen. Tech. Rep. SO-43. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 123 p.

Smalley, G.W. 1986. Site classification and evaluation for the Interior Uplands: forest sites of the Cumberland Plateau and Highland Rim/Pennyroyal. Tech. Publ. R8-TP9. Atlanta: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station and Southern Region. 518 p.

Smalley, G.W. 1991a. Classification and evaluation of forest sites on the Natchez Trace State Forest, State Resort Park, and wildlife management area in west Tennessee. Gen. Tech. Rep. SO-85. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 73 p.

Smalley, G.W. 1991b. No more plots; go with what you know: developing a forest land classification system for the Interior Uplands. In Mengle, D.; Tew, T., eds. Ecological land classification: applications to identify the productive potential of southern forests, Proceedings of a symposium. Gen. Tech. Rep. SE-68. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 48–58.

Smalley, G.W.; Sharber, L.B.; Gregory, J.C. 1996. Ecological land classification as a basic theme for the management of wildlands in Tennessee: a start. Environmental Monitoring and Assessment. 39: 579–588.

Smalley, G.W.; Todd, S.D.; Tarkington, K.W., III. 2006. Description of a land classification system and its application to the management of Tennessee's State forests. In: Conner, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 65–70.

Springer, M.E.; Elder, J.A. 1980. Soils of Tennessee. Agric. Exp. Stn. Bull. 596. Knoxville, TN: University of Tennessee Agricultural Experiment Station. 66 p + map.

Wertz, W.A.; Arnold, J.F. 1975. Land stratification for land-use planning. In: Bernier, G.; Winget, C.H., eds. Forest soils and forest land management. Proceedings, fourth North

American forest soils conference. Quebec, Canada: les Presses de l'Universite Laval: 617-629.

Wheat, R.M., Jr.; Dimmick, R.W. 1987. Forest communities and their relationships with landtypes on the western Highland Rim of Tennessee. In: Hay, R.L.; Woods, F.W.; DeSelm, H., eds. Proceedings sixth central hardwood forest conference. Knoxville, TN: University of Tennessee: 377–383.

Wilson, C.W., Jr. 1968. Geology map of the Livingston quadrangle, Tennessee. GM 330-NE. Scale 1: 24,000. [Place of publication unknown]: Tennessee Department of Conservation, Division of Geology.

Wilson, C.W., Jr.; Colville, 1968. Geology map of the Hilman quadrangle, Tennessee. GM 330-NW. Scale 1: 24,000. [Place of publication unknown]: Tennessee Department of Conservation, Division of Geology.

GROWTH RING RESPONSE IN SHORTLEAF PINE FOLLOWING GLAZE ICING CONDITIONS IN WESTERN ARKANSAS AND EASTERN OKLAHOMA

Douglas J. Stevenson, Thomas B. Lynch, and James M. Guldin[1]

INTRODUCTION

Width reduction in growth rings in shortleaf pine (*Pinus echinata* Mill.) following glaze ice conditions produces a characteristic pattern dependent on live-crown ratio and extent of crown loss. Ring widths of 133 trees for 3 years preceding and 7 years following the December 2000 ice storm (Bragg and others 2002) in western Arkansas and eastern Oklahoma were cross-dated to detect missing rings, then measured under a 32X microscope. Data from undamaged trees was detrended using a logarithmic decay function to identify the climate signal, which was then incorporated into the model. Ring width was affected by live-crown ratio, proportion of crown lost, presence of branch damage, and tree basal area (Aubrey and others 2007, Smolnik and others 2006). Stand basal area, tree height, and diameter were not significant. Three trees produced no growth ring in 2001; no tree survived loss of more than 83 percent of its crown. Ring widths of undamaged trees declined slightly from an average of 0.064 inch in 2001 to an average of 0.058 inch in 2007. Branch-damaged trees had ring widths inversely proportional to live-crown ratio and averaged 0.055 inch throughout the 7 years following the storm. After 7 years, radial growth rates in trees with <55 percent crown loss are increasing, while those with >55 percent crown loss are decreasing. Diameter growth initially accelerated following ice damage; after 2 to 4 years diameter growth began to decline on trees with >55 percent crown loss.

RING-WIDTH MODEL FOR SHORTLEAF PINE GROWTH FOLLOWING ICE STORM DAMAGE

$$
\begin{aligned}
W = {} & \text{Climate} * \text{TreeBA}^{b0} * (b_1 \\
& + \text{Con} * \text{TreeBA} * \text{LCR} * (1 - \exp(b_2 * t)) \\
& + b_3 * \text{Branch} / \text{LCR} \\
& + b_4 * \text{Stem} * (\text{LCR} - b_5 * \text{CrownLoss}) / \\
& (1 + \exp(\text{CrownLoss} - 1 + b_6 * t)) \\
& + b_7 * \text{Stem} * (1 - \text{CrownLoss}) * \\
& \exp(t * (\text{CrownLoss} - 1) + b_8 * t)) \qquad (1)
\end{aligned}
$$

where

W = width of annual growth ring (inches)
Climate = average width of corresponding detrended growth ring from undamaged trees
TreeBA = tree basal area (square feet)
LCR = live crown ratio
CrownLoss = proportion of live crown lost
t = years since ice storm
Con, Branch, and Stem = dummy variables denoting an undamaged tree, a branch-damaged tree and a stem-damaged tree, respectively
$b0$, b_1, b_2, b_3, b_4, b_5, b_6, b_7, and b_8 = regression coefficients

Analysis of variance

Source	df	SS	MS	F(9,1215)	*P*
Model	9	4.1791	0.4643	762.50	<0.0001
Error	1215	0.7399	0.000609		
Uncorrected total SS	1224	4.9190			

Fit index = 0.8496
s = 0.02468

Coefficient	Estimate	Standard error
b_0	0.3517	0.0263
b_1	0.9885	0.0166
b_2	0.0427	0.0135
b_3	–0.0416	0.00997
b_4	2.4195	1.0019
b_5	0.7002	0.3452
b_6	0.7697	0.2264
b_7	–0.6720	0.2914
b_8	0.4702	0.0656

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CONCLUSIONS

There is an initial increase in diameter growth following ice storm damage. After 2 to 4 years growth levels off. Loss of more than 55 percent of the top part of the trunk results in progressive reduction in radial growth. Loss of more than 83 percent of the live crown is fatal.

ACKNOWLEDGMENTS

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LITERATURE CITED

Aubrey, Doug P.; Coleman, Mark D.; Coyle, David R. 2007. Ice damage in loblolly pine: understanding the factors that influence susceptibility. Forest Science. 53(5): 580–589.

Bragg, Don C.; Shelton, Michael G.; Heitzman, Eric. 2002. Silvicultural lessons from the December 2000 ice storms. In: Walkingstick, Tamara; Kluender, Richard; Riley, Tom, eds. Proceedings of the 2002 Arkansas forestry symposium. Little Rock, AR: [Publisher unknown]: 1–6.

Smolnik, Matthew Hessl, Amy; Colbert, J.J. 2006. Species-specific effects of a 1994 ice storm on radial tree growth in Delaware. Journal of the Torrey Botanical Society. 133(4): 577–584.

REINEKE'S STAND DENSITY INDEX: A QUANTITATIVE AND NON-UNITLESS MEASURE OF STAND DENSITY

Curtis L. VanderSchaaf[1]

Abstract—When used as a measure of relative density, Reineke's stand density index (SDI) can be made unitless by relating the current SDI to a standard density but when used as a quantitative measure of stand density SDI is not unitless. Reineke's SDI relates the current stand density to an equivalent number of trees per unit area in a stand with a quadratic mean diameter (*Dq*) of either 10 inches (English units) or 25.4 cm (metric units). Thus, when used as a quantitative measure, SDI is in fact unit dependent on two levels, one is whether English or metric units are used, and the second is whether the unit area is 1 acre (ha) or two, three, etc. Foresters should express SDI as either number of trees per acre or /ha to clearly indicate the unit. When viewing SDI as a quantitative measure, it is legitimate to use the slope of the linear portion of an individual stand's size-density trajectory.

INTRODUCTION

Measures of stand density help managers identify levels of competition and site utilization and to determine necessary management scenarios to meet objectives. Reineke (1933) developed a stand density index (SDI) that relates the current stand density to an equivalent density in a stand with a quadratic mean diameter (*Dq*) of 10 inches. Reineke's SDI can be expressed as:

$$\text{SDI} = N(Dq/10)^b \qquad (1)$$

where
- SDI = Reineke's stand-density index
- N = trees per acre
- Dq = quadratic mean diameter (inches)
- b = exponent of Reineke's equation, often reported to equal –1.605

Reineke's SDI can be expressed on the ln-ln scale as:

$$\ln(\text{SDI}) = \ln N + b \ln Dq - b \ln 10 \qquad (2)$$

where
- ln = natural logarithm
- b = is the slope of the relationship between ln$N$ and lnDq in fully stocked stands, equivalent in magnitude to the exponent of equation (1), and other variables as previously defined.

This measure of stand density has been widely used in the development of density management diagrams and as a measure of stand density reported in scientific articles as well as in forest inventories. Papers have presented a variety of summary measures with accompanying units but failed to include units when presenting SDI (e.g., Cochran and Barrett 1999, Curtis and Marshall 2002, Williams 1994). For example, Curtis and Marshall (2002) included a figure presenting basal area (their figure 7) with accompanying units but the figure presenting SDI (their figure 14) did not include an accompanying unit. In this current paper, it is shown that SDI is not unitless and that units should accompany any report of SDI.

Reineke's SDI can be used either as a relative measure or a quantitative measure of stand density (Avery and Burkhart 2002, p. 321–324). Stocking refers to using a quantitative measure to relate the current stand density to some optimum stand density thought to best meet management objectives. When viewing Reineke's SDI strictly as a relative measure, the slope in equation (2) must be estimated exclusively using stands that are fully stocked (Clutter and others 1983, p. 72), or that are at the maximum level of ln$N$ for a specific lnDq for a particular species. Thus, the optimum stand densities are those of fully stocked stands and the measure of stocking is comparing Reineke's SDI of any stand to the fully stocked stands. Measures of stocking are often quantified using a ratio producing a unitless measure ranging from zero to 1, when using SDI this is referred to as relative SDI or "percentage stocking" (Reineke 1933). The terms "relative" and "relate" have caused confusion in the understanding of SDI though. Many foresters believe SDI is exclusively a relative measure to the maximum N per acre for a given Dq in even-aged stands of a certain species (Clutter and others 1983, p. 72 and 73).

When using SDI as a quantitative measure of stand density, this measure is relative not because it is compared to some optimum stand density and is thus unitless or strictly a relative measure, but because it relates the current stand density to an equivalent N per acre of a stand with a Dq of 10 inches (fig. 1). Thus, in some sense the standard becomes a stand with a Dq of 10 inches, but in fact any value of Dq could be used as the standard. A better word than "relates" when using SDI as a quantitative measure of stand density would be "equates." Nonetheless, this relation should not be confused as to imply that SDI is unitless. In fact, Reineke (1933) clearly specified that his SDI could be used as a quantitative measure, "It is deemed more desirable, however, to use the

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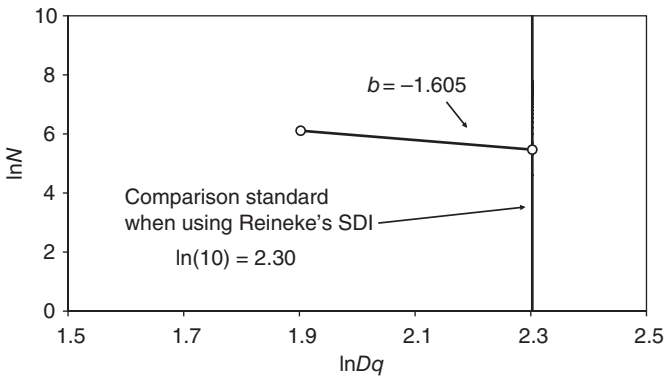

Figure 1—For Reineke's stand-density index, a slope of –1.605 is often used to relate (equate) the current stand density, e.g., 450 trees per acre, *Dq* = 6.7 inches, to an equivalent trees per acre (exp[5.47] = 237 trees per acre) for a stand with a *Dq* equal to the standard measure of 10 inches.

number of trees as the index. This is a quantitative, not a relative, measure and permits a better visualization of stand conditions."

What is advantageous about viewing Reineke's SDI as a quantitative measure is that the slope of equation (2) can be estimated for each individual stand. This approach eliminates having to estimate the slope exclusively using stands thought to be "fully stocked," or that are at the maximum level of ln*N* for a specific ln*Dq* for a particular species. Thus, the maximum size-density relationships (MSDR) of an individual stand become the optimum and we can consider these MSDRs of an individual stand as "fully stocked." One only needs to determine what observations are within the linear portion of self-thinning for an individual stand [VanderSchaaf and Burkhart (2008)—defined as the MSDR dynamic thinning line] and then estimate the slope between ln*N* and ln*Dq*, defined as the MSDR dynamic thinning line slope. Alternatively, when viewing Reineke's SDI as a quantitative measure, an estimate of the population average MSDR dynamic thinning line slope (VanderSchaaf and Burkhart 2007) can be used in Reineke's equation. Remember, from a quantitative perspective, we are only interested in equating the current stand density to a stand with a *Dq* of 10 inches. Estimating the slope of an individual stand to calculate SDI is not as foreign as it may appear. It has been well accepted that the MSDRs of each individual species can serve as defining "fully stocked" whereby we can estimate species-specific slopes in equation (2) allowing us to compare SDI among a variety of species. Using an individual stand is no different, we are simply estimating the slope of equation (2) at a much finer level.

The use of trees per acre implies SDI is not unitless. Equation (1) can also be expressed as:

$$\text{SDI} = N(Dq/25.4)^b \qquad (3)$$

where
SDI = Reineke's stand-density index
N = trees/ha
Dq = quadratic mean diameter (cm)
b = exponent of Reineke's equation, often reported to equal 1.605

It is obvious that whether one uses *N* and *Dq* in metric or English units a different value of SDI will be obtained. However, the actual on-the-ground stand density is still the same, only the units of measure have been changed. When using metric units, an SDI of 500 implies a much different level of competition and site occupancy as compared to when using English units. As opposed to equating the current stand density to an equivalent stand density with a *Dq* of 10 inches, equation (3) determines what trees/ha would equate the current stand density to a stand with a *Dq* of 25.4 cm. This is the first level where SDI is not unitless, the units of measure will impact the meaning of the stand-density index measure.

The second level where SDI is not unitless is whether *N* represents the number of trees per 1 acre (ha) or the number of trees per 2; 3; 1,000; etc., acres (ha). Rather than equating the current stand density to an equivalent number of trees with a *Dq* of 10 inches on a per-acre basis, we could also equate stand densities by using the number of trees per 2 acres, per 3 acres, etc. An SDI of 500 would represent a much different level of competition and site occupancy if *N* was on a per-acre (ha) basis or if *N* was on a 1,000-acre (ha) basis. This is the second level where SDI is not unitless.

In conclusion, SDI should not be considered unitless, a fact that Reineke himself pointed out and others have clearly noted (Husch and others 2003, p. 179). Additionally, SDI should not exclusively be considered a relative measure, it can also be interpreted as a quantitative measure. In the future, foresters should report SDI as trees per acre or trees/ha to clearly indicate the unit being used and, as many foresters already do, specify if Reineke's relative SDI is being reported.

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LITERATURE CITED
Avery, T.E.; Burkhart, H.E. 2002. Forest measurements. 5th ed. New York: McGraw-Hill Higher Education. 456 p.

Clutter, J.L.; Fortson, J.C.; Pienaar, L.V. [and others]. 1983. Timber management: a quantitative approach. Malabar, FL: Krieger Publishing Co. 333 p.

Cochran, P.H.; Barrett, J.W. 1999. Thirty-five-year growth of ponderosa pine saplings in response to thinning and understory removal. Res. Pap. PNW-RP-512. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 28 p.

Curtis, R.O.; Marshall, D.D. 2002. Levels-of-growing-stock cooperative study in Douglas-fir: report no. 14-Stampede Creek: 30-year results. Res. Pap. PNW-RP-543. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 77 p.

Husch, B.; Beers, T.W.; Kershaw, J.A., Jr. 2003. Forest mensuration. 4th ed. Hoboken, NJ: John Wiley. 443 p.

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

VanderSchaaf, C.L.; Burkhart, H.E. 2007. Comparison of methods to estimate Reineke's maximum size-density relationship species boundary line slope. Forest Science. 53: 435–442.

VanderSchaaf, C.L.; Burkhart, H.E. 2008. Using segmented regression to estimate stages and phases of stand development. Forest Science. 54: 167–175.

Williams, R.A. 1994. Stand density management diagram for loblolly pine plantations in north Louisiana. Southern Journal of Applied Forestry. 18: 40–45.

FUEL LOADING IN THE SOUTHERN APPALACHIAN MOUNTAINS MAY BE A FUNCTION OF SITE QUALITY AND DECOMPOSITION RATES

Thomas A. Waldrop, Lucy Brudnak, and Sandra Rideout-Hanzak[1]

Abstract—Fuel distribution in the Southern Appalachian Mountain region was measured on 1,008 study plots that were stratified by topographic position (aspect and slope position). Few fuel differences occurred among topographic positions indicating that fuel accumulation is no greater on highly productive sites than on less productive sites. Litter was slightly higher on undisturbed upper slopes than on lower slopes but woody fuels showed no significant differences. Rhododendron (*Rhododendron* spp.) and mountain laurel (*Kalmia latifolia*) were less common than expected occurring on 25 and 42 percent of sampled plots, respectively. The lack of significant differences among topographic positions for woody fuels suggests that varying inputs associated with site productivity are balanced by varying decomposition rates.

INTRODUCTION

Fire, both lightning and human caused, has played a significant role in the evolution of plant communities in the Southern Appalachian Mountains (Van Lear and Waldrop 1989). Fire exclusion policies on public lands likely reduced the diversity of these mountains and may have altered fuel loads. The dynamic nature of forest structure resulting from the succession of fire-dependent pine-hardwood communities to hardwood-dominated stands, as well as an abundant ingrowth of flammable understory species such as mountain laurel (*Kalmia latifolia*) makes it necessary to measure and update fuel load estimates frequently (Harrod and others 2000, Vose and others 1999). Fuel loads are a particular concern in this region because the numbers of retirement communities and single homes multiply each year.

Prediction of fuel loading in the Southern Appalachian Mountains can be as complex as the mountains themselves, because fuels may be closely associated with site quality and forest cover type. Studies by Iverson and others (2003), Kolaks and others (2004), and Waldrop and others (2004) suggest that fuel loads are controlled by the varying inputs associated with different species and productivity levels across the landscape while Abbott and Crossley (1982) discussed the impacts of varying decomposition rates at different site types. At any given time since disturbance, loading of fuels is a function of inputs from dying or broken vegetation minus losses from decay.

Although some data exist, there is limited documentation of fuels across the diverse topography of the region. In addition, past work has not covered the range of inherent variability. Thus our specific objective was to determine fuel loading by type across a range of combinations of aspect and slope positions in the Southern Appalachian Mountains.

METHODS

We designed this study to provide an exhaustive dataset of fuel loading in the Southern Appalachian Mountains because of the limited documentation of these fuels, the diverse topography of the region, and our perception that variability would be high among combinations of aspect and slope position. Therefore, we selected study sites in four States representing much of the range in elevation and topography of the region. We sampled one study area of 10 square miles in each State: South Carolina, Georgia, North Carolina, and Tennessee. Study sites included: the Sumter National Forest in northwestern South Carolina, the Chattahoochee National Forest in northeastern Georgia, the Nantahala National Forest in western North Carolina, and the Great Smoky Mountains National Park in southeastern Tennessee.

Plot locations were generated randomly within each 10-square-mile study area and were stratified by slope position and aspect using ArcView GIS software. We defined topographic position as a combination of slope position and aspect and assumed that tree productivity and, thus, fuel loading would be greater on more productive sites. Fifty plots each were located on middle slopes and lower slopes on northeast (325° to 125°) and southwest (145° to 305°) aspect. An additional 50 plots were located on ridgetops, the driest of all sites, for a total of 250 plots in each of the 4 study areas (1,000 total). Additional plots were included when necessary to give adequate representation of all slope position/aspect combinations. The resulting dataset had measurements from 1,008 plots.

Dead and down woody fuels were surveyed using Brown's (1974) planar intersect method along three 50-foot transects arranged with a common starting point and with the outer two transects 45° apart. Orientation of the middle transect in each set was determined randomly. We recorded numbers of 1- and 10-hour fuels (zero to 0.25 inch in diameter and 0.25 to 1.0 inch in diameter, respectively) crossing the transect plane along the first 6 feet of each transect. Along the first 12 feet of each transect, we recorded the number of 100-hour fuels (1.0 to 3.0 inches in diameter) crossing the transect plane. All fuels >3.0 inches in diameter, at the point where they crossed

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the transect plane, were classified as 1,000-hour fuels and were counted along the entire length of each transect. One thousand-hour fuels were recorded by diameter, type (hardwood or softwood), and decay class (solid or rotten). These counts were converted to weights using Brown's (1974) equations and specific gravity estimates for southern species by decay class developed by Anderson (1982). At the 12-, 25-, and 40-foot marks along each of the three transects we measured litter depth (O_i and O_e layers) and height of dead and down woody fuel. Litter depth was converted to weight using equations for pine and hardwood litter developed by Waldrop and others (2004) and Phillips and others (2006).

The center transect became the midline of a 50- by 44-foot vegetation plot with each side of the plot extending 22 feet from it. All trees taller than 4.5 feet were recorded within the entire plot, identified by species, and assigned to a 2-inch diameter class. On one-half of each plot, we visually estimated percent cover of ericaceous shrubs, primarily rhododendron (*Rhododendron* spp.) and mountain laurel.

Fuel loads were analyzed by analysis of variance using topographic position as the independent variable. Dependent variables included weights of litter, 1-, 10-, 100-, and 1,000-hour fuels plus cover of live fuels, predominantly mountain laurel and rhododendron. Mean separation was by linear contrast. All differences were considered significant at $\alpha = 0.05$.

RESULTS

Composition of major species groups on the 1,008 plots measured in this study was similar across topographic positions (table 1). Total basal area averaged 128 square feet per acre across all plots; it was greatest on lower slopes and decreased toward the ridges. Oak (*Quercus* spp,) was the dominant species group followed pines (*Pinus* spp.). The most common oak species included chestnut oak (*Q. prinus*), scarlet oak (*Q. coccinea*), and northern red oak (*Q. rubra*). Chestnut oak and scarlet oak were most common on dry sites and northern red oak was most common on moist sites. White pine (*P. strobus*) was the most common pine throughout all study sites and was more common on northeastern slopes than on southwestern slopes.

Downed woody fuels showed few differences in fuel loading across aspect/slope position plots (table 2). The only observed differences occurred in the litter layer. The litter on the 1,008 sample plots tended to be heaviest along the ridges and decreased going downhill on both southwest and northeast slopes, suggesting that decomposition exceeded leaf litter inputs on the wetter sites. Even though this difference among site types was significant, the relative differences were small. There was approximately 8 percent less litter on northeast lower slopes (1.68 tons per acre) than on ridges (1.83 tons per acre). There were no significant differences among slope/aspect combinations for loading of 1-, 10-, 100-, and 1,000-hour fuels or average fuel bed depth. These data should be considered preliminary because analyses of impacts such as disturbance or cover type are not yet complete. However, these findings closely agree with those of Kolaks and others (2003) indicating that down woody fuels are uniformly distributed across slopes and aspects.

Another component of fuels in eastern hardwood systems that must be considered is live fuel cover, particularly from ericaceous shrubs. Waldrop and Brose (1999) and Phillips and others (2006) indicate a strong relationship of fire intensity to a cover of mountain laurel. In this study, both mountain laurel and rhododendron were missing from most measured plots but occurred in thick clumps where they were found. Mountain laurel was found at all aspect/slope position combinations but was significantly more abundant on southwest upper slopes (table 2). Wildfires that might

Table 1—Basal area by major species or species groups and topographic position on 1,008 study plots in the Southern Appalachian Mountains of South Carolina, Georgia, North Carolina, and Tennessee

Species or group	Northeast lower	Northeast upper	Ridge	Southwest upper	Southwest lower	All plots
	square feet per acre					
Maples	14.8	11.8	16.6	10.4	10.9	12.9
Hickories	3.9	3.5	4.8	7.4	4.4	4.8
Yellow-poplar	6.5	10.9	16.6	13.5	10.5	12.1
Pines	29.6	32.2	15.3	19.6	35.7	26.5
Oaks	44.9	43.6	32.7	34.9	42.3	39.7
Hemlock	7.4	3.5	10.0	7.4	3.5	6.4
Understory	14.8	9.2	12.6	18.7	11.3	13.3
Other overstory	11.8	12.2	10.9	11.3	14.8	12.2
Total basal area	133.7	126.9	119.5	123.3	133.4	127.9

Table 2—Fuel characteristics by slope position and aspect in the Southern Appalachian Mountains of Tennessee, North Carolina, Georgia, and South Carolina

Slope/ Aspect	Litter	1-hour	10-hour	100-hour	1,000-hour	Fuel height	Mountain laurel	Rhododendron
	tons per acre					*inches*	*percent*	
Northeast Lower	1.68 a	0.32	0.91	3.8	24.0	4.3	10.6 a	37.0 c
Northeast Upper	1.82 b	0.30	0.91	3.5	18.0	4.6	13.6 a	19.7 b
Ridge	1.83 b	0.29	1.04	4.2	16.2	4.6	13.1 a	6.1 a
Southwest Upper	1.75 ab	0.30	0.97	3.7	17.3	4.3	21.0 b	6.8 a
Southwest Lower	1.70 a	0.29	0.92	3.4	18.3	4.1	15.6 a	15.4 b

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

occur could reach dangerous intensities if they burned uphill on dry southwest slopes and ran into thickets of mountain laurel. Rhododendron was also present at all slope/aspect combinations, but it was more common at lower slope and northeast-facing plots.

DISCUSSION AND CONCLUSIONS

This paper provides a preliminary analysis of an extensive dataset. An important component of fuel loading, disturbance, has yet to be considered. Wildfires, insects, diseases, wind, and ice temporarily increase the rate of fuel input by breaking limbs and felling trees.

An objective of this study was to determine if fuel loading varied by topographic position. We assumed that different species composition and productivity levels associated with slope position and aspect would create different fuel loads. For many fuel variables, there was no difference in loading across topographic positions. Litter weights varied significantly among slope/aspect positions but weights of 1-, 10-, 100-, and 1,000-hour fuels and fuel height did not vary. This result was surprising because of the large sample size used for analysis (1,008 plots). The result gives support to the conclusions of Kolaks and others (2003) and Waldrop (1996) who described the dynamics of fuel inputs and outputs of Southern Appalachian ecosystems. Both studies suggested that the differences in fuel inputs, associated with site quality at different topographic positions, were balanced by differing decomposition rates. Productive sites tend to have higher decomposition rates (Abbot and Crossley 1982), thus removing the higher fuel inputs sooner. The balance between inputs and decomposition deserves additional study in the Southern Appalachian region.

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LITERATURE CITED

Abbott, D.T.; Crossley, D.A., Jr. 1982. Woody litter decomposition following clear-cutting. Ecology. 63(1): 35–42.

Anderson, H.L. 1982. Aids to determining fuel models for estimating fire behavior. Gen. Tech. Rep. INT-122. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station. 22 p.

Brown, J.K. 1974. Handbook for inventorying downed woody material. Gen. Tech. Rep. INT-16. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station. 20 p.

Harrod, J.C.; Harmon, M.E.; White, P.S. 2000. Post-fire succession and 20th century reduction in fire frequency on xeric southern Appalachian sites. Journal of Vegetation Science. 11: 465–472.

Iverson, L.R.; Yaussy, D.A.; Rebbeck, J. [and others]. 2003. Spatial and temporal distribution of fire temperatures from prescribed fires in the mixed oak forests of southern Ohio. In: Van Sambeek, Jerry W.; Dawson, Jeffrey O.; Ponder,

Felix, Jr. [and others], eds. Proceedings, 13th central hardwood forest conference. Gen. Tech. Rep. NC-234. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Research Station: 293–294.

Kolaks, J.; Cutter, B.E.; Loewenstein, E.F. [and others].. 2003. Fuel loading in the central hardwoods. Pap. 1A-3. In: Proceedings on the 2nd international wildland fire ecology and fire management congress (CD–ROM). Orlando, FL: American Meterology Society. 6 p.

Kolaks, J.; Cutter, B.E.; Loewenstein, E.F. [and others]. 2004. The effect of thinning and prescribed fire on fuel loading in the central hardwood region of Missouri. In: Yaussy, Daniel A.; Hix, David M.; Long, Robert P.; Goebel, P. Charles, eds. Proceedings, 14th central hardwood forest conference. Gen. Tech. Rep. NE-316. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station: 168–178.

Phillips, Ross J.; Waldrop, Thomas A.; Simon, D.M. 2006. Assessment of the FARSITE model for predicting fire behavior in the Southern Appalachian Mountains. In: Conner, Kristina 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: 521–525.

Van Lear, D.H.; Waldrop, T.A. 1989. History, uses, and effects of fire in the Southern Appalachians. Gen. Tech. Rep. SE-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 20 p.

Vose, J.M.; Swank, W.T.; Clinton, B.D. [and others]. 1999. Using stand replacement fires to restore Southern Appalachian pine-hardwood ecosystems: effects on mass, carbon, and nutrient pools. Forest Ecology and Management. 114: 215–226.

Waldrop, T.A. 1996. Dynamics of coarse woody debris—a simulation study for two southeastern forest ecosystems. In: Crossley, D.A., Jr.; McMinn, James A., eds. Coarse woody debris in the South—implications to management and biodiversity. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 18–24. Chapter 7.

Waldrop, T.A.; Brose, P.H. 1999. A comparison of fire intensity levels for stand replacement of Table Mountain pine (*Pinus pungens* Lamb.). Forest Ecology and Management. 113(1999): 155–166.

Waldrop, Thomas A.; Glass, Dallas W.; Rideout, Sandra [and others]. 2004. An evaluation of fuel reduction treatments across a landscape gradient in Piedmont forests: preliminary results of the National Fire and Fire Surrogate Study.In: Conner, Kristina F., ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 54–57.

EFFECTS OF PLANTING DENSITY AND GENOTYPE ON LOBLOLLY PINE STANDS GROWING IN THE MOUNTAINS OF SOUTHEASTERN OKLAHOMA

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Abstract—We determined the effects of planting density (4- by 4-, 6- by 6-, 8- by 8-, and 10- by 10-foot spacing) on stand-level height, diameter at breast height, stem volume, basal area, and periodic annual increment for two loblolly pine (*Pinus taeda* L.) seed sources. Seed sources for the 25-year-old stands were a North Carolina seed source (NCC 8-01) and a regionally local Oklahoma/Arkansas seed source (O/A mix 4213). The research site was a droughty, mountain soil in southeastern Oklahoma outside the native range of loblolly pine. Except for subsoiling at planting, no other stand-level treatments were applied. The experimental design was a split-plot with whole-plot factor planting density (n = 2) and the split-plot factor genotype (n = 8). While similar between genotypes, as stand density increased, basal area increased (188 to 241 square feet per acre), stand volume increased (4,230 to 5,030 cubic feet per acre), and average tree diameter decreased (10.1 to 6.3 inches). Average tree heights decreased with stand density (61 to 55 feet), and the North Carolina genotype was taller than the Oklahoma/Arkansas genotype (60 vs. 56 feet). Between ages 22 and 25, periodic annual increment decreased with stand density, and density-dependent mortality occurred in the two highest planting densities. These results indicate that over 25 years, the North Carolina genotypes performed as well or better than the regional seed source. In addition, sites outside the native range of loblolly pine can support relatively high basal areas and can successfully grow commercial loblolly pine stands with little outside inputs.

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Guldin, James M., ed. 2013. 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. 585 p.

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