USDA FOREST SERVICE
U.S. DEPARTMENT OF AGRICULTURE

Southern Research Station | General Technical Report SRS-274 | July 2024

Proceedings of the 22nd Biennial Southern Silvicultural Research Conference

March 21–23, 2023

Fredonia Hotel and Convention Center

Nacogdoches, TX

DISCLAIMER

The use of trade or firm names in these proceedings is for reader information and does not imply endorsement by the U.S. Department of Agriculture or Stephen F. Austin State University of any product or service.

PESTICIDE PRECAUTIONARY STATEMENT

Some of the papers and extended abstracts within these proceedings include research involving pesticides. They do not contain recommendations for their use, nor do they imply that the uses discussed here have been registered. All use of pesticides must be registered by appropriate State and/or Federal agencies before they can be recommended.

CAUTION: Pesticides can be injurious to humans, domestic animals, desirable plants, and fish or other wildlife if they are not handled or applied properly. Use all herbicides selectively and carefully. Follow recommended practices for the disposal of surplus pesticides and their containers.

EDITORS' NOTE

Papers published in these proceedings were submitted by authors in electronic media. Some editing was done to improve the readability, accuracy, and consistency of contributed papers. Authors are responsible for content and accuracy of their individual papers and the quality of illustrative materials.

Cover photo courtesy of Jason Ellis.

July 2024
https://doi.org/10.2737/SRS-GTR-274

Forest Service
Research & Development
Southern Research Station
General Technical Report SRS-274

Southern Research Station
200 W.T. Weaver Blvd.
Asheville, NC 28804
https://www.research.fs.usda.gov/srs

Proceedings of the 22nd Biennial Southern Silvicultural Research Conference

March 21–23, 2023
Fredonia Hotel and Convention Center
Nacogdoches, TX

EDITED BY

Don C. Bragg
Brian P. Oswald
Nancy E. Koerth

HOSTED BY

Arthur Temple College of Forestry and Agriculture, Stephen F. Austin State University
USDA Forest Service, Southern Research Station

PREFACE

The 22nd Biennial Southern Silvicultural Research Conference (BSSRC) was held March 21–23, 2023, at the Fredonia Hotel in Nacogdoches, TX. This conference is 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 artificial and natural hardwood regeneration, fire, intermediate management and stand development, forest measurements and modeling, longleaf pine, shortleaf pine, loblolly pine, and vegetation management. In addition to an opening morning plenary session, six sessions including 68 oral and 46 poster presentations were offered on March 21 and 22. Additionally, a field tour was held on Thursday, March 23, and participants visited several sites in eastern Texas to observe management on the Boggy Slough Conservation Area, research at a carbon afforestation project of Stephen F. Austin State University's Arthur Temple College of Forestry and Agriculture, and the I.E. Fairchild State Forest. In all, nearly 190 people attended the 22nd BSSRC.

ACKNOWLEDGMENTS

With over 4 decades of programs, the BSSRC remains an important venue for sharing and disseminating silvicultural research and training the next generation of silviculturists. This conference would not have been possible without the dedication and collaboration between universities, the USDA Forest Service, State and private agencies, and forest industry partners. We gratefully acknowledge the staff of the Arthur Temple College of Forestry and Agriculture of Stephen F. Austin State University for their diligence in handling the fiscal responsibilities of the meeting and for helping with the registration process. Special thanks to the 22nd BSSRC steering committee members for their countless hours of dedication and work over the many months it took to facilitate this conference. The following persons devoted numerous hours to reviewing abstracts, establishing the program for oral and poster presentations, developing field tours and presentations, and making all the other necessary arrangements to ensure a successful conference.

PROGRAM CHAIRS AND LOCAL ARRANGEMENTS

Brian Oswald, Stephen F. Austin State University, Nacogdoches, TX

Don Bragg, USDA Forest Service, Southern Research Station, Monticello, AR

STEERING COMMITTEE MEMBERS

Gordon Holley, Louisiana Tech University, Ruston, LA

Jeremy Stovall, Stephen F. Austin State University, Nacogdoches, TX

Rebecca Kidd, Stephen F. Austin State University, Nacogdoches, TX

Kyle Cunningham, University of Arkansas Cooperative Extension, Little Rock, AR

Eric Taylor, Texas A&M Forest Service, College Station, TX

Nancy Koerth, USDA Forest Service, Southern Research Station, Nacogdoches, TX

Callie Schweitzer, USDA Forest Service, Southern Research Station, Huntsville, AL

Conner Fristoe, Weyerhaeuser, Columbus, MS

Mickey Rachel, RoyOMartin Lumber Company, Alexandria, LA

Brian Lockhart, Consulting Silviculturist, Gentry, AR

OTHER SUPPORTING PERSONNEL

Brandi Derouen, Stephen F. Austin State University, Nacogdoches, TX

Christen Miles, Stephen F. Austin State University, Nacogdoches, TX

Jon Gauntt, Stephen F. Austin State University, Nacogdoches, TX

Amy Neal, Stephen F. Austin State University, Nacogdoches, TX

M. Evan Stuber, USDA Forest Service, Southern Research Station, Crossett, AR

SESSION MODERATORS

March 21, 2023

Welcome and Plenary Session: Brian Oswald, Stephen F. Austin State University

Field Trip Sites Session: Brian Oswald, Stephen F. Austin State University

Growth and Yield and Modeling 1: Eric Taylor, Texas A&M University

Growth and Yield and Modeling 2: Eric Taylor, Texas A&M University

Fire 1: Mohammad Bataineh, University of Arkansas-Monticello

Fire 2: Callie Schweitzer, USDA Forest Service, Southern Research Station

Fire 2: Callie Schweitzer, USDA Forest Service, Southern Research Station

Hardwood 1: Brian Lockhart, Retired Silvicultural Researcher

Hardwood 2: Brian Lockhart, Retired Silvicultural Researcher

Shortleaf Pine: Don Bragg, USDA Forest Service, Southern Research Station

Poster Session: unmoderated

March 22, 2023

Restoration and Reclamation: Conner Fristoe, Weyerhaeuser Company

Loblolly Pine 1: Conner Fristoe, Weyerhaeuser Company

Loblolly Pine 2: Conner Fristoe, Weyerhaeuser Company

Hardwoods 3: Brian Lockhart, Retired Silvicultural Researcher

Intermediate Treatments/Competition Control 1: Joshua Adams, Louisiana Tech University

Intermediate Treatments/Competition Control 2: Wayne Clatterbuck, University of Tennessee

Intermediate Treatments/Competition Control 3: Andy Ezell, Mississippi State University

Management and Silvics: Andy Ezell, Mississippi State University

Ecosystem Services: Tom Dean, Louisiana State University

Longleaf Pine 1: Brian Oswald, Stephen F. Austin State University

Longleaf Pine 2: Pat Minogue, University of Florida

Longleaf Pine 3: Pat Minogue, University of Florida

Ethics Training: Brian Lockhart, Retired Silvicultural Researcher

22ND BSSRC STUDENT ORAL AND POSTER PRESENTATION CONTEST WINNERS [MAJOR PROFESSOR(S) IN PARENTHESES]

Oral Presentations

Megan Berry – The University of Tennessee (Dr. Patrick Keyser)

Jacob Murry – The University of Alabama (Dr. Justin Hart)

Ryan Bohannon – Clemson University (Dr. Geoff Wang)

Poster Presentations

Gabriel Nyen – Mississippi State University (Dr. Adam Polinko/Dr. Joshua Puhlick)

Joshua Everett – Oklahoma State University (Dr. Chris Zou)

Sloane Scott – University of Missouri (Dr. Benjamin Knapp)

IN MEMORIAM
DON C. BRAGG, PH.D.
SEPTEMBER 12, 1970–DECEMBER 30, 2023

The passing of Don C. Bragg, his wife Hope, and adult children Kenny and Beth, in a disastrous home explosion at Hope's father's home in Michigan, was terrible news to learn on New Year's Eve. That news saddened the hearts of everyone who met Don during his career.

Don and I shared a common character flaw that many of you share as well—a certain measure of disdain for authority! I think that is a key attribute for a good research scientist. It drives the scientist to ask questions, to look at data from a variety of perspectives, and meticulously find new and different ways to support and test their hypotheses.

Don published more than 100 manuscripts, most of which were in refereed outlets regionally, nationally, and internationally. Through his research, Don touched almost all of us who are involved in the ecology, silviculture, and history of Southern forests.

He was also an excellent editor for scientific writing and had just as much impact on many of us there as he did in his research. Whether he edited (in some cases, meticulously!) your manuscripts for Station proceedings, or your submissions to the *Journal of Forestry* (where he served as an editor from 2013 to 2017), or even if you wrote one of the dozens of manuscripts he reviewed anonymously, Don touched many of our careers.

He was also gifted in technology transfer, especially with boots-on-the-ground. If you've taken a tour of the Crossett Experimental Forest during the past 2 decades, you've witnessed first hand Don's thoughts on the development of Southern yellow pine stands in the West Gulf region. You've also been touched by Don if you've heard his views on the successes and failures of active forest management during one of the dozens of Forest Service or Society of American Foresters field tours in which he participated regionally and nationally over the past 20 years.

My heart aches that we've lost Don in the prime of his career. But the body of work Don leaves behind is extraordinarily rich. Scientists stand on the shoulders of scientific giants before us. Don Bragg now joins the cadre of giants upon whom we will all rely upon in the future. I'll be eternally grateful for the pleasure of knowing him.

James M. Guldin, Ph.D.
USDA Forest Service, Southern Research Station (retired)
Springfield, Missouri

IN MEMORIAM
SCOTT D. ROBERTS, PH.D.
OCTOBER 22, 1957–APRIL 27, 2023

Up until his retirement from Mississippi State University in 2018, Scott was a consistent participant in the Biennial Southern Silvicultural Research Conference. I counted 16 papers in the associated proceedings that had his name in the list of authors. This does not include posters presented by his graduate students. Scott contributed more to this conference than his presentations from the podium or poster board, however. Scott could talk to anybody about anything with name recall that even a politician would envy. His nonchalant attitude (honed by his "leisure ethic") was a product of his self-confidence and his intelligence. He laughed and smiled a lot. He laughed at irony and as way to acknowledge disagreements. Most of all, however, he laughed because he was having good times with friends. The silvicultural community of the South missed him when he moved to Utah, but there was always news about what he was doing in retirement. His tragic death ends his story but does not erase his contributions to our discipline nor our memory of him.

Tom Dean, Ph.D.
Louisiana State University

The Scott David Roberts Memorial Scholarship fund has been established by the family and friends of Scott to support forestry majors concentrating in forest management within the College of Forest Resources at Mississippi State University.

CONTENTS

FIELD TRIP SITES

Silviculture at the Boggy Slough Conservation Area: Research and Management to Promote Conservation
Steven B. Jack and Robert S. Sanders 2

GROWTH AND YIELD MODELING

An Integrated System Approach to Model Tree and Stand Growth
Quang V. Cao 6

A Stand-Level Growth and Yield Model System for Shortleaf Pine Plantations in the Western Half of the Southeastern United States: an Initial Attempt
Curtis L. VanderSchaaf 10

An Individual-Tree Measure of Growing Stock and Stand Density
Thomas J. Dean 20

Biomass Estimates for Natural-Origin Hardwood Saplings of Southeastern Arkansas
Mohammad M. Bataineh 26

A Growth And Yield System of Differential Equations for Slash Pine Plantations Including Response to Silvicultural Treatments
Laura Ramirez, Cristian R. Montes, and Bronson P. Bullock 33

Diameter Estimation Based on LiDAR Point Clouds at Stand Level of Loblolly Pine Plantations
Clinton Talmage, Yuhui Weng, Yanli Zhang, and Jason Grogan 44

Quantifying Forest Aboveground Biomass Using Remote Sensing and Ground Measurements for Mixed Hardwood Forests in East Tennessee
Sakar Jha, Sheng-I Yang, D. Stuart Hale, David Shoch, Trisha Johnson, Donald G. Hodges, and Kevin P. Hoyt 51

FIRE

Predicting Fire Intensity Using Vegetation, Fuel, and Weather Variables
John Craycroft and Callie Schweitzer 59

Prescribed Fire Barriers and Solutions for Managing Sustainable Forests: Forum Discussion
Callie Schweitzer, Dan Dey, and Andrea Brandon 68

Linking Fire Behavior to Effects: Use of Pyrometers and Thermocouples
Mohammad M. Bataineh, Brant Portner, Matthew Pelkki, and Robert Ficklin **72**

Influences of Overstory Pyrophytic and Mesophytic Trees and Understory Solar Irradiance on Leaf Litter and Woody Debris Fuel Moisture Retention
Luiza G. Lazzaro, Heather D. Alexander, Jeffrey B. Cannon, and Michael J. Aspinwall **80**

HARDWOODS

Advancing Established Natural White Oak (*Quercus alba*) Reproduction Through a Midstory Removal
Canaan J. Dugger and Wayne K. Clatterbuck **84**

Timing of Dormant Season Inundation is Crucial to Flood Effects on Bottomland Red Oaks
Benjamin A. Babst, Richard D. Sample, and James Cook **93**

The Response of Willow Oak (*Quercus phellos*) Seedling Roots to Fall Environmental Cues and Winter Flooding
Benjamin A. Babst, Jonathan M. Kressuk, Tucker Collins, Emile S. Gardiner, and Mohammad M. Bataineh . **96**

Factors Affecting Natural Oak Regeneration in Bottomland Hardwood Forests in Northern Missouri: Light Levels, Acorn Losses, or Lack of Acorns?
Isaac Hayford and Benjamin O. Knapp **99**

Leaf Area Index Across Eastern Cottonwood and Hybrid Poplars and its Relationship With Productivity and Water Use Efficiency Under Coppicing and Noncoppicing Management in the Southeastern United States
Jiaxin Wang and Heidi J. Renninger **108**

Planted White Oak (*Quercus alba*) Response to Spacing Density and Orientation in Gap Openings in the Blue Ridge Mountains of North Carolina
Stacy L. Clark, Tara Keyser, Scott E. Schlarbaum, and Arnold Saxton **110**

Multiphased Shelterwood on the Daniel Boone National Forest, Kentucky, Tests Herculean Effort to Regenerate Oak
Callie Schweitzer, Jared Calvert, Stacy Clark, and Jacob Royse **114**

A Look Back on 26 Years of Hardwood Development After Afforestation on Flood-Prone Agricultural Lands in the Louisiana Delta
Joshua P. Adams and William B. Patterson **119**

Regenerative Effects of Patch-Cut Harvests at Natchez Trace State Forest in West Tennessee After More Than 25 Years
Wayne K. Clatterbuck and Joshua K. Biggerstaff **122**

SHORTLEAF PINE

Site Preparation for Shortleaf Pine Cluster Planting and Restoration of Shortleaf Pine-Hardwood Mixtures: First Year Results
David Clabo and Michael Hinson **128**

Sustainable Pathways for Shortleaf Pine Restoration in Uncertain Climates
Casey Iwamoto, Courtney Siegert, Joshua Granger, Krishna Poudel, Adam Polinko, and Zachary Freedman **135**

Growth Response to Competition and Prescribed Fire in Planted Shortleaf Pine
Hope M. Fillingim, Benjamin O. Knapp, Daniel C. Dey, Michael C. Stambaugh, John M. Kabrick, and Grant P. Elliott **138**

RESTORATION AND RECLAMATION

Some Silvicultural Implications of Updating the Southern Region's Old-Growth Guidance
Don C. Bragg **147**

Reconstructing Stand Dynamics of a Shortleaf Pine-Oak Woodland in East Texas: a Case Study
Benjamin O. Knapp and Steven B. Jack **156**

Evaluating the Trajectory of Vegetation Succession on Clearcut and Surface-Mined Sites
Ian Kennedy, Melody Mount, Wayne Clatterbuck, David Buckley, and Jennifer Franklin **163**

LOBLOLLY PINE

Exploring the Linkage Between Physiology and Shade-Related Mortality in Loblolly Pine Branches
Adelodun R. Majekobaje and Thomas J. Dean **170**

Characterizing Spatial Dependencies of Competing Vegetation in Thinned Loblolly Pine Plantations
John B. Young, Bronson P. Bullock, and Cristian R. Montes **176**

Climate-Smart Forestry and its Implications for Southern Silviculture
G. Geoff Wang **186**

INTERMEDIATE TREATMENTS/COMPETITION CONTROL

Aminocyclopyrachlor Cut Stem Applications Compared to Standard Herbicides for Eucalyptus Control
Patrick J. Minogue and Kimberly A. Lorentz **197**

A Comparison of Garlon® 4 Ultra, Garlon® XRT, Vista® XRT, Vastlan®, Chopper GEN2®, Freelexx® Oust® Extra, and TerraVue® in Site Preparation Applications
A. Brady Self, J. Eric Ezell, and Andrew W. Ezell. **204**

A Comparison of Glufosinate Products for the Control of Natural Pines
John Ezell, Andrew Self, and Andrew Ezell **207**

Seasonal Light Response Curves of *Ligustrum sinense*: A Test of Extended Leaf Phenology
Trisha M. Markus, Lauren S. Pile Knapp, David R. Coyle, and G. Geoff Wang **210**

Efficacy of Mid-Rotation Brush Control Applications Using Imazapyr and Glufosinate
Andrew Ezell, Andrew Self, and John Ezell **219**

Evaluating Dormant-Season Herbicide Applications for the Control of Chinese Privet (*Ligustrum sinense*)
Karen Vaughn, Lauren S. Pile Knapp, and G. Geoff Wang. **221**

MANAGEMENT AND SILVICS

Private Forest Landowners' Perspectives on Feral Hog Management and Control in Arkansas, Louisiana, and East Texas
Alyssa Mineau, Nana Tian, Jianbang Gan, and Gordon Holley **232**

Plant Succession Following Dozer Line Installation Through a Shale Woodland Glade on the Ouachita National Forest in Arkansas
Virginia L. McDaniel, Gabriel L. De Jong, and Christopher M. Schalk **239**

Eastern and Western Loblolly Pine Sources in the Western Gulf Coastal Plain Region—Root System Observations
Mary Anne Sayer, Michael C. Tyree, Michael A. Blazier, Joshua P. Adams, and Brett T. Wolfe **257**

ECOSYSTEM SERVICES

Does Restoring Woodland Structure with Loblolly or Slash Pine Produce Desired Ecosystem Functions?
John L. Willis, Don C. Bragg, Jeffery B. Cannon, Kamal J.K. Gandhi, Kathryn R. Kidd, Adam D. Polinko, Joshua J. Puhlick, Dan Saenz, Mary Anne Sayer, Christopher M. Schalk, Andrew B. Self, Courtney M. Siegert, and J. Morgan Varner. **268**

Assessment of Tradeoffs Among Ecosystem Services of Wetland Reserve Easements
Tyler W. Corbin, Mohammad M. Bataineh, Benjamin Babst, and I-Kuai Hung **271**

Evaluating Stem and Branch Characteristics of 5-Year-Old Sonderegger Pine Saplings
Kelsey D. Shoemaker, D. Paul Jackson, and Joshua P. Adams **281**

LONGLEAF PINE

Disturbance and Stand Structure in a Longleaf Pine Woodland
David L. Phillips, J. Davis Goode, and Justin L. Hart **288**

Does Longleaf Pine Cone Production Correlate With Seed Production, Size, and Germination Rate?
Rachel E. Nation, Heather D. Alexander, John L. Willis, Michael Aspinwall, and Jeffery B. Cannon **292**

Effects of Fertilization and Endophyte Application on Growth of Planted Longleaf Pine
Baylor Doughty, Adam Polinko, and Courtney Siegert **294**

Silvicultural Strategies for Converting Longleaf Pine Plantations to Multiaged Longleaf Pine Stands
Joshua J. Puhlick, Adam D. Polinko, and Gabriel F. Nyen **300**

Hurricanes Stimulate Cone Production in Longleaf Pine Woodlands
Jeffery B. Cannon, Brandon T. Rutledge, Joshua J. Puhlick, John L. Willis, and Dale G. Brockway **306**

Silvicultural Strategies for Promoting Longleaf Pine Recruitment in Multiaged Stands
Joshua J. Puhlick, Benjamin O. Knapp, Ian Goldberg, Brandon T. Rutledge, R. Scott Taylor, and John L. Willis **311**

Prescribed Burning Has Little Effect on Longleaf Pine Wood Quality
Edward O. Andrews and John L. Willis **317**

POSTER SESSION

Evaluating Bias in Tree-Level Measurements Obtained With a Backpack LiDAR Unit
Michael K. Crosby and Joshua P. Adams **320**

Correcting the Erroneous Geographic Coordinates of the Southwide Pine Seed Source Study Source Areas
Don C. Bragg and Shawna L. Reid **323**

Developing Successful Management Strategies for Invasive Plant Species at Townsend Bombing Range, Georgia
G. Ryan Bohannon, Trisha M. Markus, Yingzhou Tang, Gary S. Herndon, and G. Geoff Wang **332**

Effectiveness of Shelterwood and Strip Clearcuts for Regenerating Bottomland Oaks in Northeast Texas
Cason Brewster, Robert Heinemann, Randal Holeman, Cassandra Meek, Keith Anderson, Josh Everett, Omkar Joshi, and Rodney Will **337**

Reforestation Tax Incentive Impacts on Financial Returns of Loblolly Pine Plantations for Nonindustrial Private Forest Landowners in Mississippi
Curtis L. VanderSchaaf and Yanshu Li **341**

Impacts of Diesel and Insurance Costs on Owner-Operator Log Truck Drivers in the Western Gulf
Curtis L. VanderSchaaf. **351**

Field Trip Sites

SILVICULTURE AT THE BOGGY SLOUGH CONSERVATION AREA: RESEARCH AND MANAGEMENT TO PROMOTE CONSERVATION

Steven B. Jack and Robert S. Sanders

EXTENDED ABSTRACT

The Boggy Slough Conservation Area (BSCA), located west of Lufkin, TX, is a 19,055-acre property in Houston and Trinity Counties. The property has a mix of upland and bottomland forested ecosystems and a long history of management for conservation values. Since the early 20th century the property has been under private ownership, with the Temple family and their forest products corporation as owners for most of this time. The T.L.L. Temple Foundation purchased the property in 2013 to honor the Temple family, the corporate history, and their combined legacy in east Texas, and to demonstrate and promote forest conservation in the region. In 2015, the entire property was placed under a conservation easement to prevent future development and preserve its conservation values. The property is divided into North Boggy Slough (approximately 7,000 acres) and South Boggy Slough (approximately 12,000 acres), with separate hunting leases managed for each area.

BSCA has a variety of forested habitats (fig. 1). The majority of the property is classified as uplands, some of which are bisected by intermittent drainages in rolling topography. Upland forest types compose approximately 73 percent of the forest at BSCA. Upland pine plantations are predominantly loblolly pine (*Pinus taeda*), ranging in age from 15 to >40 years old, though younger plantations (≤10 years old) were planted to shortleaf (*P. echinata*) and longleaf (*P. palustris*) pine on appropriate sites. Natural upland pine stands, many of which have not been actively managed in many years, have canopies dominated primarily by loblolly and shortleaf pines with scattered longleaf pine present in some stands. Upland hardwood-mixed pine stands have variable composition, with the hardwood component [e.g., southern red oak (*Quercus falcata*), post oak (*Q. stellata*), and sweetgum (*Liquidambar styraciflua*)] increasing at lower topographic positions and higher soil moisture. The bottomland hardwood forests are not pristine due to the history of harvests in the early 20th century, but they are relatively intact compared to other bottomland forests in the region. Common overstory species in the bottomland forests include water oak (*Q. nigra*), willow oak (*Q. phellos*), overcup oak (*Q. lyrata*), green ash (*Fraxinus pennsylvanica*), and American sycamore (*Platanus occidentalis*). Forty-eight different plant associations,

Steven B. Jack, Executive Director, Boggy Slough Conservation Area, T.L.L. Temple Foundation, Lufkin, TX 75901

Robert S. Sanders, Director of Forest and Wildlife Management, Boggy Slough Conservation Area, Lufkin, TX 75901

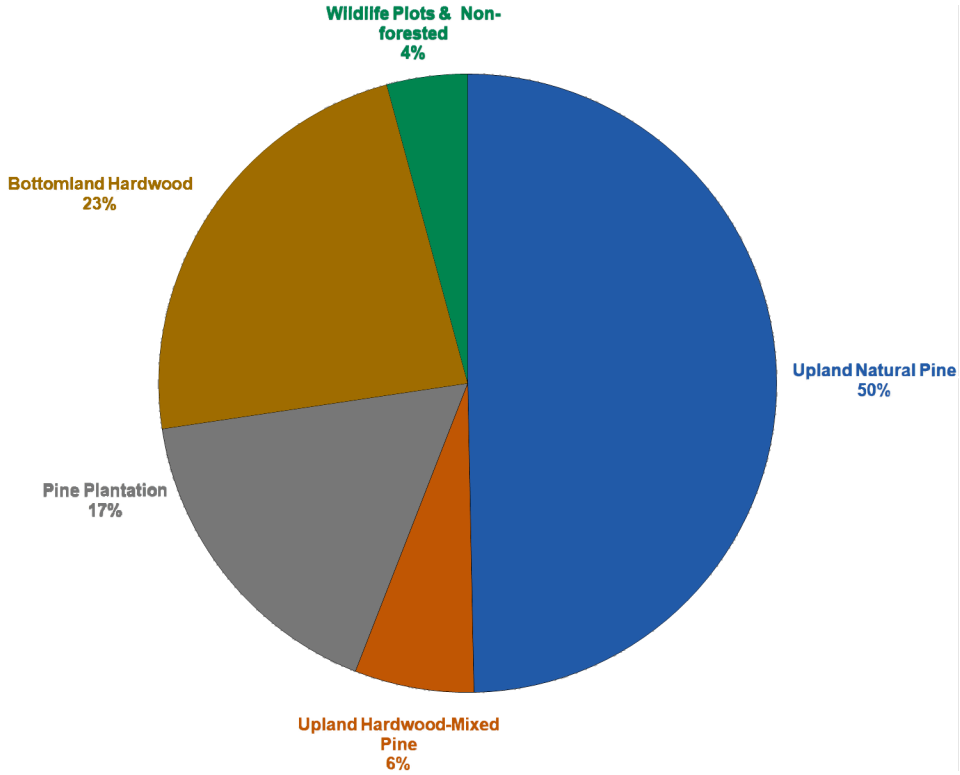

Figure 1—Percent composition of different forest types at the Boggy Slough Conservation Area in east Texas.

which are strongly related to soil characteristics, have been documented at BSCA, and several plant and animal species of conservation concern are found at the property.

The mission of BSCA is to "serve as a model of excellence for East Texas through ecological research and outreach to promote conservation, management, and stewardship of natural resources" (https://www.boggysloughconservation.org/). Within this mission, bottomland forest management focuses on regenerating desired species and controlling invasive species (e.g., Chinese tallow, *Triadica sibifera*). Active canopy manipulations are minimized in the bottomlands due to restrictions in the conservation easement. Upland forests are managed to restore and maintain an open pine or pine-hardwood canopy, to promote native ground cover species, and to favor shortleaf and longleaf pine on appropriate sites due to their fire tolerance. Primary silvicultural treatments in the uplands include selection-based harvests, frequent prescribed fire, and targeted use of herbicides (table 1). Through these management actions in upland forests, the open canopy forest condition is now more prevalent on the property (compared to conditions in 2013 when the property was purchased), and wildlife community responses to the changing habitat have been documented.

Research at BSCA is conducted through collaborations. Typical collaborations are initiated with academic and agency scientists and often utilize graduate student projects. The BSCA signature research program is achieved through the Buddy Temple Visiting Scholar and Graduate Fellow program. Under this program, regionally important research topics are identified, and eminent scholars are recruited to address the

Table 1—Acres harvested and burned with prescribed fire at the Boggy Slough Conservation Area, 2014–2023

	Acres harvested			Acres burned		
Year	South Boggy	North Boggy	Total	South Boggy	North Boggy	Total
2014	134	0	134	3,635	1,566	5,201
2015	94	202	296	807	834	1,641
2016	308	235	543	4,172	2,875	7,047
2017	905	448	1,353	3,407	1,874	5,281
2018	326	137	463	4,092	1,611	5,703
2019	430	166	596	3,192	212	3,404
2020	243	234	477	1,111	1,359	2,470
2021	212	354	566	3,652	2,822	6,474
2022	340	97	437	3,190	2,054	5,244
2023	123	268	391	3,477	2,361	5,838
Total	3,115	2,141	5,256	30,375	17,568	48,303

topics. Recent Buddy Temple Visiting Scholars have examined stand dynamics and regeneration in mixedwood forests and the effects of forest management on trophic structure of wildlife communities.

The property is frequently used for outreach events by BSCA and our partner organizations to show conservation-oriented alternatives to the dominant forest management regimes in east Texas. Sites at various stages in the restoration process provide case studies to highlight in outreach events, while monitoring of vegetation and wildlife responses to the management activities provides quantitative information to share with target outreach audiences.

While still a relatively new program, activities at the BSCA are bringing attention to conservation-oriented practices for managing forest resources in east Texas.

Growth and Yield Modeling

AN INTEGRATED SYSTEM APPROACH TO MODEL TREE AND STAND GROWTH

Quang V. Cao

ABSTRACT

An integrated system contains a unified mathematical structure for modeling tree and stand growth that can be applied at any level of resolution. This paper outlines the recent progress that has made this concept a reality: 1) the integrated system is based on a whole-stand growth model; 2) given a tree list, an individual-tree growth model can be derived from the stand model; 3) a diameter distribution (e.g., Weibull) can be partitioned into individual trees, which can be grown according to the tree growth model from 2) and 4) a stand table can be converted into a tree list, which can be used to derive a tree growth model by use of the technique from 2). This integrated system can provide growth predictions for the current stand. The inputs can be a plot summary, a tree list, a stand table, or a diameter distribution. The outputs are constrained to have numerical consistency across all levels of resolution.

INTRODUCTION

Contemporary forest management relies on the application of growth and yield models to forecast future stand growth for various scenarios. These models are classified based on the level of detail (or resolution) of their outputs into four broad categories (Burkhart and Tomé 2012). Whole-stand models are relatively simple models that provide information for the entire stand. The predicted stand attributes can be stand survival (Stankova 2016), basal area per unit area (Naing 2020), or both (Zhang and others 2022). Size-class models deal with diameter classes; these models can be stand-table projection models that project a stand table (containing the number of trees in each diameter class) into the future (Allen and others 2011), or diameter-distribution models that use a probability density function (PDF) to model the frequency of tree diameters (Carretero and Alvarez 2013). Individual-tree models deliver detailed information for each tree, in terms of tree survival (Karlsson and Norell 2005), tree diameter growth (Subedi and Sharma 2011), or both (Sun and others 2019). The user sometimes has difficulty in deciding between a model that provides reliable estimates and another that gives sufficient details. Daniels and Burkhart (1988) promoted the idea of creating an integrated system that can provide consistent growth and yield estimates at various levels of resolution, because it is based on a unified mathematical structure.

The concept of an integrated system has recently become a reality. This paper presents such a system, one that can handle inputs in a variety of formats and still give consistent results.

INPUT: STAND SUMMARY

A stand-level growth model that projects current stand attributes into the future is the heart of the integrated system. Either an existing model is used or a new model is constructed for this purpose. The stand growth model predicts number of trees per hectare ($\hat{N}_{2i}$) and quadratic mean diameter ($\hat{Q}_{2i}$) for plot i at time 2 (future) from stand summary at time 1 (present) in terms of age (A_{1i}) dominant height (H_{1i}), number of trees per hectare (N_{1i}), and quadratic mean diameter (Q_{1i}). The following example is a stand growth model from Cao (2022a):

$$\hat{N}_{2i} = N_{1i} - exp(a_0 + a_1 RS_{1i} + a_2 H_{1i} + a_3 N_{1i}/A_{1i} + a_4/A_{1i}) \quad (1)$$

$$\hat{Q}_{2i} = Q_{1i} + exp(a_5 + a_6 N_{1i}/A_{1i} + a_7/A_{1i}) \quad (2)$$

$$\hat{B}_{2i} = K\, \hat{N}_{2i}\, \hat{Q}_{2i}^2 \quad (3)$$

Quang V. Cao, Professor, School of Renewable Natural Resources, Louisiana State University Agricultural Center, Baton Rouge, LA 70803

where

$\hat{B}_{2i}$ = predicted stand basal area (m^2/ha) at the end of a 5-year growth period

A_{1i} = stand age in years for plot i at the beginning of the growth period

N_{1i} = number of trees per hectare at age A_{1i}

H_{1i} = dominant height (m) at age A_{1i}

$RS_{1i} = \frac{\sqrt{10000/N_{1i}}}{H_{1i}}$ = relative spacing at age A_{1i}

$K = \pi/40{,}000$

INPUT: TREE LIST

The earliest attempts to link tree models to stand models was the disaggregation approach (Ritchie and Hann 1997), in which information obtained from the tree model is used to disaggregate stand growth (predicted by a whole-stand model) among trees in the tree list, which is a list of tree diameters in a plot. Cao (2021) presented an integrated system in which the tree model contains identical terms as does the stand model, with an additional term to describe individual-tree variations. Recently, Cao (2019) showed how one can derive a tree survival model from any existing stand survival model; the level of accuracy and precision depended on the stand model performance and on whether or not tree-level survival data were available.

In the case when tree-level survival data are not available, Cao (2022a) developed a two-step procedure to derive a tree-level growth model from a stand-level model by use of a tree list. In the first step, an individual-tree survival equation can be derived from the stand survival equation as follows:

$$p_{ij} = 1 - exp\left[\alpha_1\left(d_{1ij} - 0.95\ Dmin_{1i}\right)\right] \quad (4)$$

where

p_{ij} = survival probability of tree j in plot i

d_{1ij} = diameter at breast height (d.b.h., cm) of tree j in plot i at age A_{1i}

$Dmin_{1i}$ = minimum d.b.h. of trees in plot i

α_1 is the solution of

$$\hat{N}_{2i} = \sum_{j=1}^{n_{1i}} p_{ij}/s \quad (5)$$

where

p_{ij} = survival probability of tree j in plot i at age A_{1i}

s = plot size in hectares

In the second step, the predicted d.b.h. at time 2 ($\hat{d}_{2ij}$) is computed from the d.b.h. at time 1 (d_{1ij}) as follows:

$$\hat{d}_{2ij} = d_{1ij} exp(\alpha_2 d_{1ij}) \quad (6)$$

where

α_2 is solved such that

$$\hat{B}_{2i} = K \sum_{j=1}^{n_{1i}} p_{ij} \hat{d}_{2ij}^2/s \quad (7)$$

The derived tree model was evaluated against other individual-tree models in terms of their abilities to predict tree diameter and survival. The derived tree model performed better than either the unadjusted or disaggregated tree model. The sum of the ranks (lower is better) was 12.91 for the derived tree model, 15.83 for the unadjusted tree model, and 17.28 for the disaggregated tree model (Cao 2022a). Note that both the derived and disaggregated models were constrained by output from the stand-level model.

INPUT: DIAMETER DISTRIBUTION

In this scenario, a PDF is used to describe the diameter distribution of a forest stand. Instead of the commonly used 2-cm diameter classes, 0.2-cm diameter classes are used; each could be considered as a single tree. The diameter of this tree is the midpoint of the diameter class; the number of trees per hectare that this tree represents is obtained by integrating the PDF for each 0.2-cm class. Cao (2022b) used this technique to generate a tree list. Now that the input is a tree list, an individual-tree model can be derived from the stand model by use of the technique described in the previous section.

This method was evaluated against the methods of 1) parameter recovery from $\hat{\bar{D}}_2$ and $\hat{Q}_2$, which are predicted from stand variables, 2) parameter recovery $\hat{\bar{D}}_2$ from $\hat{Q}_2$ and, which are projected from D_1 and Q_1, and 3) growing the PDF from time 1 to time 2. Results from Cao (2022b) indicated that the new method performed better than the first two methods, yielding lower error indices (Reynolds and others 1988) and relative differences between observed and predicted number of trees in each diameter class. The

third method requires many steps involving mathematical integration. The new method is analogous to solving the numerical integration problem by use of the trapezoid integration approach, and as a result, its performance was almost identical to that of the third method.

Bailey (1980) observed that if the diameter distributions at both ages A_1 and A_2 belong to the same family of distributions (e.g., Weibull), then the diameter random variable Y at age A_2 is implied to be transformed from variable X at age A_1. The attempt to link a diameter-distribution model to an individual-tree model was later expanded by Cao (1997) and Qin and others (2007). The new method described in this section treats the diameter variable X as individual trees, each with its own expansion factor (number of trees per hectare) depending on its diameter. This process 1) converts a diameter-distribution model at age A_1 into an individual-tree model, 2) grows the trees to age A_2, and 3) converts them back to a diameter-distribution model. The method provides a linkage across the spectrum of growth and yield models: individual-tree, diameter-distribution, and whole-stand models.

INPUT: STAND TABLE

In a ground-breaking paper, Nepal and Somers (1992) suggested the use of a truncated Weibull distribution to approximate the number of trees in each diameter class. After evaluating the uniform, Weibull, and S_B distributions, Cao and Coble (2014) concluded that the truncated Weibull distribution was most appropriate for modeling stand table data.

Cao (2023) proposed a procedure for handling stand-table input as follows:

- Fit a complete Weibull distribution to the stand table by use of the cumulative distribution function regression (Cao and Coble 2014).
- Establish a truncate Weibull distribution for each diameter class.
- Generate a tree list from 0.2-cm diameter classes (Cao 2022b).
- Derive an individual-tree growth model from the stand growth model (Cao 2022a).
- Convert the new tree list to the future stand table.

The new method was found to perform better than the existing stand-table projection models (Cao 2007, Cao and Baldwin 1999, Nepal and Somers 1992) in terms of two error indices (Reynolds and others 1988).

CONCLUSION

This paper outlines the recent progress that has turned into reality the concept of an integrated system, as envisioned by Daniels and Burkhart (1988). The heart of this integrated system is a whole-stand growth model. If the input is a tree list, then an individual-tree growth model can be derived from the stand growth model. If the input is either a diameter distribution or a stand table, a tree list can be generated, and then used for the derivation of the tree growth model. The advantage of this system is that outputs are constrained to have numerical consistency across all levels of resolution. Even though the system was developed from loblolly pine (*Pinus taeda*) plantations, the same approach could be applied to multispecies stands with some modifications.

LITERATURE CITED

Allen, M.G., II; Coble, D.W.; Cao, Q.V. [and others]. 2011. A modified stand table projection growth model for unmanaged loblolly and slash pine plantations in east Texas. Southern Journal of Applied Forestry. 35(3): 115–122. https://doi.org/10.1093/sjaf/35.3.115.

Bailey, R.L. 1980. Individual tree growth derived from diameter distribution models. Forest Science. 26: 626–632.

Burkhart, H.E.; Tomé, M. 2012. Modeling forest trees and stands. Lexington, KY: Springer. 457 p.

Cao, Q.V. 1997. A method to distribute mortality in diameter distribution models. Forest Science. 43: 435–442.

Cao, Q.V. 2007. Incorporating whole-stand and individual-tree models in a stand-table projection system. Forest Science. 53: 45–49.

Cao, Q.V. 2019. A method to derive a tree survival model from any existing stand survival model. Canadian Journal of Forest Research. 49: 1598–1603. https://doi.org/10.1139/cjfr-2019-0171.

Cao, Q.V. 2021. A unified system for tree- and stand-level predictions. Forest Ecology and Management. 481: 118713. 8 p. https://doi.org/10.1016/j.foreco.2020.118713.

Cao, Q.V. 2022a. Deriving a tree growth model from any existing stand growth model. Canadian Journal of Forest Research. 52: 169–174. https://doi.org/10.1139/cjfr-2021-0106.

Cao, Q.V. 2022b. Predicting future diameter distributions given current stand attributes. Canadian Journal of Forest Research. 52: 561–567. https://doi.org/10.1139/cjfr-2021-0216.

Cao, Q.V. 2023. A new approach to project stand tables. Canadian Journal of Forest Research. 53: 714–720. https://doi.org/10.1139/cjfr-2023-0030.

Cao, Q.V.; Baldwin, V.C., Jr. 1999. A new algorithm for stand table projection models. Forest Science. 45: 506–511.

Cao, Q.V.; Coble, D.W. 2014. Deriving a diameter distribution from stand table data. Forest Science. 60: 628–635. https://doi.org/10.5849/forsci.12-150.

Carretero, A.C.; Alvarez, E.T. 2013. Modelling diameter distributions of *Quercus suber* L. stands in "Los Alcornocales" Natural Park (Cádiz-Málaga, Spain) by using the two parameter Weibull functions. Forest Systems. 22: 15–24. http://dx.doi.org/10.5424/fs/2013221-02142.

Daniels, R.F.; Burkhart, H.E. 1988. An integrated system of forest stand models. Forest Ecology and Management. 23: 159–177. https://doi.org/10.1016/0378-1127(88)90080-1.

Karlsson, K.; Norell, L. 2005. Modelling survival probability of individual trees in Norway spruce stands under different thinning regimes. Canadian Journal of Forest Research. 35: 113–121. https://doi.org/10.1139/x04-161.

Naing, Y.M. 2020. Growth and yield models for thinned teak stands in Taungoo District, Bago Region of Myanmar. International Journal of Scientific and Research Publications. 10: 546–557.

Nepal, S.K.; Somers, G.L. 1992. A generalized approach to stand table projection. Forest Science. 38: 120–133.

Qin, J.; Cao, Q.V.; Blouin, D.C. 2007. Projection of a diameter distribution through time. Canadian Journal of Forest Research. 37: 188–194. https://doi.org/10.1139/x06-216.

Reynolds, M.R., Jr.; Burk, T.E.; Huang, W.-C. 1988. Goodness-of-fit tests and model selection procedures for diameter distribution models. Forest Science. 34: 373–399.

Ritchie, M.W.; Hann, D.W. 1997. Implications of disaggregation in forest growth and yield modeling. Forest Science. 43: 223–233.

Stankova, T.V. 2016. A dynamic whole-stand growth model, derived from allometric relationships. Silva Fennica. 50: 1406. 21 p. https://doi.org/10.14214/sf.1406.

Subedi, N.; Sharma, M. 2011. Individual-tree diameter growth models for black spruce and jack pine plantations in northern Ontario. Forest Ecology and Management. 261: 2140–2148. https://doi.org/10.1016/j.foreco.2011.03.010.

Sun, S.; Cao, Q.V.; Cao, T. 2019. Evaluation of distance-independent competition indices in predicting tree survival and diameter growth. Canadian Journal of Forest Research. 49: 440–446. https://doi.org/10.1139/cjfr-2018-0344.

Zhang, X.; Cao, Q.V.; Qu, Y.; Zhang, J. 2022. Deriving tree growth models from stand models based on the self-thinning rule of Chinese fir plantations. iForest. 15: 1–7. https://doi.org/10.3832/ifor3792-014.

A STAND-LEVEL GROWTH AND YIELD MODEL SYSTEM FOR SHORTLEAF PINE PLANTATIONS IN THE WESTERN HALF OF THE SOUTHEASTERN UNITED STATES: AN INITIAL ATTEMPT

Curtis L. VanderSchaaf

ABSTRACT

A system of growth and yield equations is presented for shortleaf pine (*Pinus echinata*) plantations using data from across the western half of the Southeastern United States. Data used in model fitting were obtained from Arkansas, Mississippi, Oklahoma, and Tennessee (n = 168 plot measurements). Planting densities ranged from 194 to 1,742 seedlings per acre, and measurement ages ranged from 1 to 30 years. The majority of the measurement ages were ≤16 years of age (n = 152). A mix of cutover and old-field sites were modeled. Predictions of dominant height, trees per acre, basal area per acre, and total volume per acre were obtained directly. Quadratic mean diameter was mathematically derived from the number of trees and basal area per acre estimates. Hopefully, as interest grows, a more complete dataset can become available resulting in an improved model system as compared to this initial modeling effort.

INTRODUCTION

Shortleaf pine (*Pinus echinata*) is native to much of the western half of the Southeastern United States, as well as southern Missouri (Guldin 2019, Little 1971). According to the Shortleaf Pine Initiative (https://shortleafpine.org/) and Guldin (2019), shortleaf was often found in open woodlands in both pine-dominated and mixed pine-oak (*Quercus* spp.) forests, allowing for a diverse assortment of native wildlife to flourish. Over the last 30 to 40 years, this extensive shortleaf pine ecosystem has lost >50 percent of its former acreage (Clabo and Clatterbuck 2015, Guldin 2019). Changes in timber management practices, altered fire regimes, massive southern pine beetle (*Dendroctonus frontalis*) outbreaks in poorly managed stands, disease, and land-use changes have contributed to this rapid decline.

Coordinated efforts by groups like the Shortleaf Pine Initiative to restore the shortleaf pine ecosystem are beginning. Although overstory woody vegetation is a major component of the restoration effort, other components of these ecosystems such as understory vegetation, biological diversity, wildlife habitat, and recreation are also highly desired. The restoration process of this ecosystem will likely begin mostly through the use of plantations (Guldin 2019).

Managers need tools to examine how different management options will likely impact financial returns. Although a more open woodland setting may be the ultimate goal by the majority of agencies, timber production will help to offset the initial conversion and regeneration costs. Additionally, partners wanting to restore or manage for components of the ecosystem such as wildlife or water quality will benefit from gaining an idea of how these stands will develop.

According to a 2022 estimate by the Forest Inventory and Analysis Unit of the U.S. Department of Agriculture, Forest Service (https://apps.fs.usda.gov/fiadb-api/evalidator), there are approximately 255,000 acres of shortleaf pine plantations, and northwestern Arkansas and southeastern Oklahoma have considerable amounts of shortleaf-dominated, naturally regenerated stands (table 1). Shortleaf pine

Curtis L. VanderSchaaf, Assistant Professor, Central Mississippi Research and Extension Center, Mississippi State University, Raymond, MS 39154

Table 1—Acreage of shortleaf pine forest types in 2022 as defined by the Forest Inventory and Analysis program of the U.S. Department of Agriculture, Forest Service

	Natural		Planted		
State	Shortleaf	Shortleaf/oak	Shortleaf	Shortleaf/oak	Total
	~~ *acres* ~~~~~~~~~~~~~~~~~~~~~~				
Alabama	111,573	114,843	3,103	0	229,519
Arkansas	1,162,452	757,307	87,132	9,319	2,016,210
Louisiana	37,010	28,850	0	0	65,860
Mississippi	164,269	169,436	10,426	8,569	352,700
Missouri	153,297	303,267	85,864	9,563	551,991
Oklahoma	509,206	454,735	26,640	0	990,581
Tennessee	50,087	133,148	8,964	0	192,199
Texas	252,732	127,357	5,425	0	385,514
Total	2,440,626	2,088,943	227,554	27,451	
Total	4,529,569		255,005		4,784,574

The Forest Inventory and Analysis program has two forest type classifications with shortleaf pine as a dominant species: shortleaf pine (FORTYPCD = 162) and shortleaf pine/oak (FORTYPCD = 404).

stands are also relatively abundant in Mississippi, Missouri, and Texas. A small amount of planting has occurred in the southern parts of Illinois and Indiana.

Projections from this current modeling effort will be compared with the only known comprehensive stand-level model system for shortleaf pine plantations. Smalley and Bailey (1974) presented a system of equations to predict yields of unthinned old-field plantations in the highlands of Alabama, Georgia, and Tennessee. Site index was predicted using an equation presented in Smalley and Bower (1971), while survival rates were predicted as a function of the average height of dominants and codominants (i.e., site quality), trees planted, and age. All (n = 104) but one plantation had planting densities exceeding 500 seedlings per acre with the majority being 1,000 seedlings per acre or greater. All equations presented in Smalley and Bailey (1974) were estimated using cross-sectional data obtained within temporarily established plots, or measurements across a range of ages obtained at the same time. Thus, projections are not based on repeated plot measures.

Smalley and Bailey (1974) predicted basal area development through the use of the survival equation and a three-parameter Weibull parameter-prediction equation system. For their per acre yield estimates, initially, the number of trees within a particular diameter class was estimated based on the stand-level survival equation and parameter estimates of the three-parameter Weibull distribution using the parameter prediction approach. Following this, an equation was used to estimate average height in each diameter class, and then individual tree volumes were predicted using an equation presented by Smalley and Bower (1968). These individual tree volumes were then multiplied by the estimated number of trees within each diameter class. Volumes across all diameter classes were then summed to produce a per acre estimate.

The objective of this paper is to present a system of equations using data from across the western half of the Southeastern United States to determine the site index if desired, predict dominant height, trees per acre, basal area per acre, and total volume per acre of shortleaf pine plantations. Quadratic mean diameter can be mathematically derived from the trees per acre and basal area per acre estimates and estimates of tons per acre can be obtained by using common conversion factors between volume and weight.

METHODS

Data

A variety of sources were used in model development (table 2, fig. 1). The first data source was a shortleaf pine spacing study established in February of 1955 on the University of Arkansas, Southwest Research and Extension Center (then named the Fruit and Truck Branch Experiment Station) near Hope, AR. Four planting densities were established of an Arkansas seed source (most likely 1-0 bareroot seedlings): 1,742 (5 feet by 5 feet), 1,210 (6 feet by 6 feet), 908 (6 feet by 8 feet), and 681 (8 feet by 8 feet) seedlings per acre. Three replications were established of each planting density. Total tree heights were recorded annually following the first through fifth growing seasons on every live tree, and then again in the seventh growing season. Diameters at breast height (d.b.h., 4.5 feet above ground level) were recorded at ages 3, 5, and 7 years. Total cubic foot volumes were estimated using an equation developed by Lenhart and others (1987).

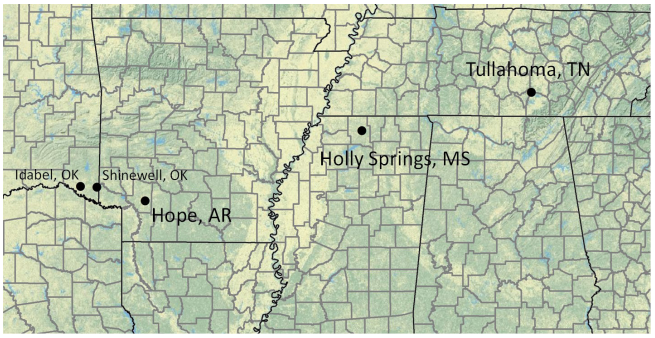

Figure 1—Locations of the five general study sites used within this study.

A second data source was from old-field plantations established on Mississippi State University's Agricultural and Forestry Experiment Station, North Mississippi Branch Experiment Station at Holly Springs, MS. This study was reported earlier by Kushla (2009).

Table 2—Number of observations, measurement ages, and summary statistics

		Study and State					
		1	2	3	4	5	
		Arkansas	Mississippi	Tennessee	Oklahoma	Oklahoma	All
Characteristic	Statistic	(n = 72)	(n = 4)	(n = 32)	(n = 28)	(n = 32)	(n = 168)
Age (years)	Range	1 to 5, 7	16	5, 10, 22, 30	1 to 5, 9, 10	1 to 5, 8, 9, 10	1 to 5, 7, 8, 9, 10, 16, 22, 30
Dominant height (feet)	Mean	9	36	42	11	13	17
	SD	5	2	28	10	10	19
	Min	2	34	6	1	1	1
	Max	21	39	73	28	31	73
Trees per acre	Mean	1,068	500	423	377	318	674
	SD	375	28	301	33	101	445
	Min	626	460	139	326	198	139
	Max	1,742	520	1,139	415	442	1,742
Basal area (square feet per acre)	Mean	26	148	83	20	16	38
	SD	29	7	55	24	19	46
	Min	0	140	13	0	0	0
	Max	89	155	221	56	61	221
Volume (cubic feet per acre)	Mean	228	2,673	2,242	356	254	813
	SD	229	229	1,842	305	258	1,281
	Min	3	2,437	34	45	6	3
	Max	712	2,911	6,731	791	843	6,731

SD = standard deviation; min = minimum; max = maximum.

Subsoil tillage treatments were performed prior to planting in December 2004 using a 20-inch ripping shank pulled by a tractor on 10-foot planting centers. Bareroot 1-0 seedlings were then planted on the retired farmland in March 2005 at a nominal density of 622 trees per acre (10 feet by 7 feet spacing). Herbaceous weed control (HWC), consisting of a combination of herbicide application and mowing, was conducted during the first year. Herbicide was applied in April of 2005 and row centers were mowed three times during the growing season on treated areas.

Four 1/20th-acre circular plots were established, and total tree height and d.b.h. were measured in late December of 2021. For modeling purposes, the plantation was considered to be 16 years old (trees were 17 years old from seed). Total cubic foot volumes were estimated using equations presented by Smalley and Bower (1968).

A third data source was published information from a cutover spacing study established southeast of Tullahoma, TN, at the Highland Rim Forestry Experiment Station (Schubert and others 2004). In 1965, the native forest on the site was harvested, cull trees injected with 2,4-D and the area mist-blown to kill the herbaceous vegetation (Schubert and others 2004). In the late winter of 1966, 1-0 seedlings (most likely bareroot) from the Tennessee Valley Authority's Clinton, TN, nursery were planted. Four planting densities were established: 1,210 (6 feet by 6 feet), 538 (9 feet by 9 feet), 303 (12 feet by 12 feet), and 194 (15 feet by 15 feet) seedlings per acre. A total of four blocks were established, but due to extremely poor survival on two blocks, only observations from the other two blocks were used in model development. Percent survival and average d.b.h. were available for ages 5 and 10 (Omiyale 1976), 22 (Schubert 2001), and 30 (Schubert and others 2004).

No site index or dominant height values were reported. However, arithmetic mean height was presented by block for ages 5, 10, 22, and 30. Thus, to estimate dominant height for each of the two blocks at a particular age, first, across all of the three other sources of data described herein, the average ratio between dominant height (the average height of the tallest 50 percent of all surviving trees) and arithmetic mean height was calculated, a value of 1.1664. Second, this average value was then multiplied by the reported arithmetic mean heights to produce estimates of dominant heights for each of the two blocks and for each measurement age.

Reported volumes were to a 4-inch diameter outside bark. Hence, to be consistent with the other studies where volume is total cubic foot volume (or to the tip of the tree), a volume-basal area ratio (VBAR) approach was used. First, the VBAR on a per acre basis for this study was estimated from Smalley and Bower (1968: Table 2):

$$VBAR = 0.5020\ Ht_{Arith} - 5.9771\ /\ QMD^2 \quad (1)$$

where

Ht_{Arith} is arithmetic mean height (feet)

QMD is quadratic mean diameter (inches)

For the purpose of this study, Ht_{Arith} and QMD replace individual tree height and d.b.h. in Smalley and Bower's equation. Second, the reported basal areas per acre were multiplied by the predicted VBARs to estimate total cubic foot volume per acre.

A fourth source was cutover plantations from two sites in southeastern Oklahoma: Idabel and Shinewell (Dipesh and others 2015). The study was a generalized randomized complete block design; i.e., a randomized complete block study was installed at each site. No mechanical site preparation was conducted, but there was some level of chemical site preparation. At the Shinewell site (study site 4 in table 2) 1-0 bareroot seedlings were planted in late February 2002 and at Idabel 1-0 (study site 5 in table 2) bareroot seedlings were planted early April 2002. Improved seedlings were from the Western Gulf Tree Improvement Program grown at the Oklahoma Forestry Services nursery in Goldsby, OK, from seed collected at the Oklahoma Forestry Services seed orchard in Idabel, OK. Each site had four blocks. Trees were planted at a nominal spacing of 8 feet by 10 feet (544 trees per acre). Each measurement plot ranged from 0.1236 to 0.1730 acres, depending on the available space, and contained 69–90 trees and an additional strip of buffer trees on each side.

Height was measured annually following the first through fifth growing seasons on every live tree at both sites. At the Idabel site, d.b.h. was measured after the third, fourth, and fifth growing seasons but at the Shinewell site only after the fourth and fifth growing seasons. Additionally, at the Idabel site, height and d.b.h. measurements were conducted at ages 8, 9, and 10. At the Shinewell site, height and d.b.h. measurements resumed at ages 9 and 10. Total cubic foot volumes were predicted using an equation presented in Dipesh and others (2015).

Stand-Level Model Development

Equations were selected based on both biological and statistical considerations. Parameter estimates were examined for consistency with biological theory (e.g., with age, basal area per acre or volume per acre should increase). Additionally, predicted values of stand development were examined to determine whether a particular equation or set of equations produced reasonable estimates across a range of ages and values of the regressors. After accounting for biologically meaningful variables and parameter estimates, the function with the highest adjusted R^2 was selected. Residuals were examined for trends. Proc Model (SAS Institute, Inc. 1989) and the Gauss-Newton algorithm were used to estimate parameters of all equations.

Model Fitting

Dominant height equation

A simple anamorphic guide curve approach (Clutter and others 1983) was selected using equation (2):

$$Ht_D = e^{\beta_0 + \beta_1 A^{-1}} \quad (2)$$

where

Ht_D = dominant height (feet), defined as the average height of the tallest 50 percent of surviving shortleaf pine trees (other species were not selected)

A = plot age (plantation age from time of planting, not from seed)

β_0, β_1= parameters to be estimated

A total of 168 observations were used in model development. After fitting (adjusted R^2 = 0.9430) and algebraic manipulation, equation (2) became:

$$SI = Ht_D e^{(25^{-1} - A^{-1})^{-12.5133}} \quad (3)$$

where

SI = dominant height at a reference base age of 25 years and others as previously defined.

Equation (4) was used to predict dominant height:

$$Ht_D = SI e^{(25^{-1} - A^{-1})^{12.5133}} \quad (4)$$

When using equation (3), and based on the oldest plot age, the average site indices at the Mississippi site (n = 4 plots), Idabel, OK (study site 5, n = 4 plots), Shinewell, OK (study site 4, n = 4 plots), and the Tennessee site (n = 8 plots) were 48.2, 58.6, 57.5, and 64.1 feet (base age 25), respectively. Due to the young maximum age of only 7 years at the Arkansas site, site index was not calculated. Figure 2 shows dominant height across time for six site indices (base age 25).

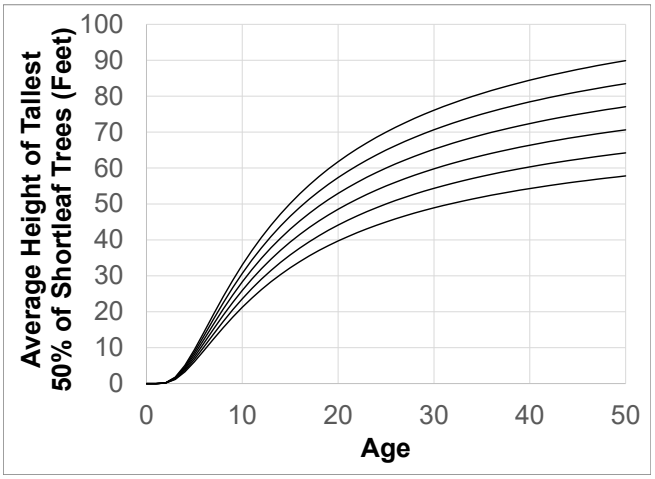

Figure 2—Dominant height across time for site indices of 45, 50, 55, 60, 65, and 70 feet (base age 25).

Mortality equation

Prior to any model fitting, biological considerations were taken into account for the selected model form. Mortality patterns over time of each experimental unit (repeated measurements of a plot) were plotted over age, dominant height (Ht_D), and existing trees per acre to determine if any trends in mortality were visually observable. Some slight relationships

were visually observed between mortality rates and stand-level variables; however, the relationships did not appear strong. Hence, model fitting criteria (adjusted R^2) were used to a greater degree to determine the final equation selected. Additionally, the biological appropriateness of parameter estimates was strongly considered. Illogical behavior in the sign or magnitude of parameter estimates could result in the exclusion of a model form. Many equation forms were considered (Zhao and others 2007) and the following equation was selected:

$$N_2 = N_1 e^{-0.06109\,(A_2^{\,0.546548} - A_1^{\,0.546548})} \quad (5)$$

n = 168, adjusted R^2 = 0.9846

where

N_2 = trees per acre at the current time (or time 2)

N_1 = previous trees per acre (or time 1)

A_2 = current age (or age at time 2)

A_1 = previous age (or age at time 1)

Equation (5) assumed the instantaneous rate of mortality changes with age.

All parameter estimates had a significance of P <0.0001. To examine for trends in model residuals, the residuals of equation (5) were plotted over age and previous trees per acre (N_1); residuals were also examined for other acceptable equations. Residual plots appeared similar regardless of the equation tested; however, equation (5) provided a better prediction of those values with large predictive errors.

Future work will concentrate on a more complete analysis of mortality patterns in these stands. Most likely a mortality estimation system that includes two predictive steps will provide more biologically meaningful mortality estimates (Woollons 1998, Zhao and others 2007). In the first step, the probability of any mortality occurring is predicted, and then the second step, if mortality is actually predicted to occur, predicts the reduction in trees per acre (or conversely, the number of trees surviving per acre). This approach is biologically superior because often mortality is sporadic and does not occur in a consistently systematic manner every year over time. Predictive behavior of equation (5) is demonstrated in figure 3.

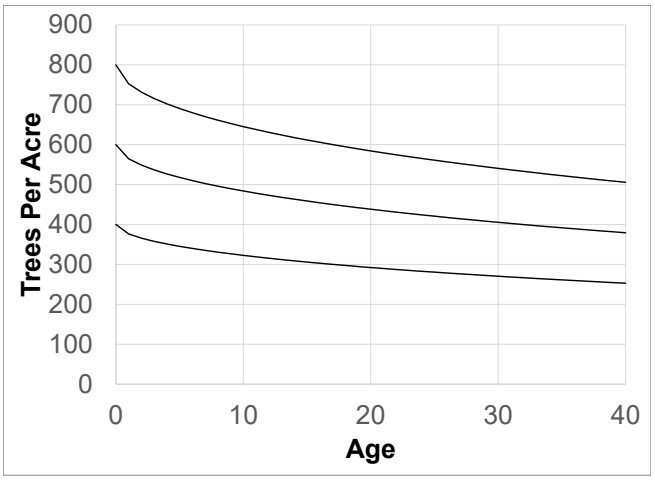

Figure 3—Number of planted shortleaf pine trees surviving per acre for planting densities of 400, 600, and 800 seedlings per acre. Curves of a particular planting density begin at the planting density.

Basal area per acre equation

Several equation forms were initially evaluated and considered to estimate basal area per acre (BAA). An algebraic difference equation, in addition to other variables, includes the lag, or the previous measurement, of basal area to predict the current basal area. Due to reductions in sample size mainly resulting from the lack of diameter measurements at younger ages, and hence basal area lag values (or the basal area per acre at the previous measurement age), no algebraic difference equations were included in the selection of the final equation. Hence, several equations from the literature, not of an algebraic difference form, were tested that included dominant height (a measure of site quality). Ultimately, equation (6) was selected:

$$BAA = e^{\,1.57094 + 0.63234\ln N - 5.88784\,(1/A) - 18.6497(1/Ht_D)} \quad (6)$$

n = 124, adjusted R^2 = 0.8947

All parameter estimates had a P <0.0197. The signs of all parameter estimates seem biologically reasonable. Basal area per acre estimates when using equation (6) are demonstrated in figure 4.

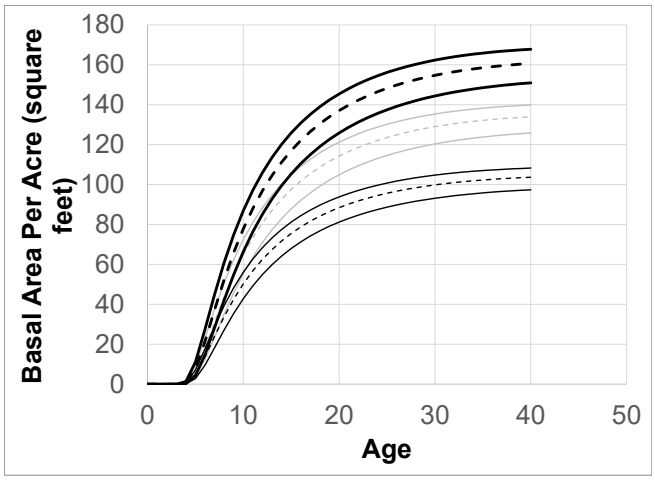

Figure 4—Basal area per acre of planted shortleaf pine for the combination of three site indices and three planting densities per acre. The light black curves = planting density of 400 seedlings per acre; gray curves = planting density of 600 seedlings per acre; and bold black curves = planting density of 800 seedlings per acre. For a particular planting density, the lower solid curve = site index of 45 feet (base age 25); dashed curve = site index of 55 feet; and upper solid curve = site index of 65 feet.

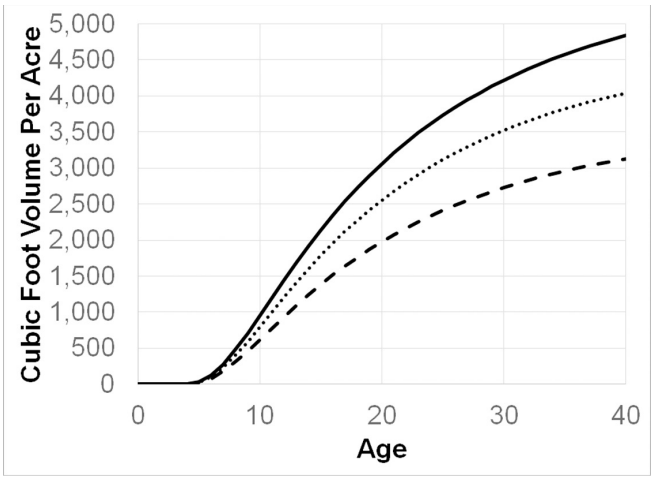

Figure 5—Total cubic foot volume per acre of planted shortleaf pine for a site index 55 feet site (base age 25) and three planting densities of 400 (dashed line), 600 (dotted line), and 800 (solid line) seedlings per acre.

Volume per acre (VAA) equation

Volume per acre can often be expressed as a function of site quality and stocking (Hall and others 2020). Thus, a constant form factor approach can be used, where volume is a function of dominant height, representing site quality, and basal area per acre, representing stocking. Additionally, the use of these two variables can be justified on a functional—even structural—level. This is because volume is a three-dimensional variable, and therefore basal area can be seen as providing two dimensions while dominant height provides the third dimension. Additionally, as expressed in the Schumacher (1939) model, the inverse of age can be included as a regressor. These two model forms were tested. Similar to fitting the basal area per acre equations, due to reductions in sample size mainly resulting from the lack of diameter measurements at younger ages, and hence volume lag values (or the volume per acre at the previous measurement age), no algebraic difference equations were considered for the equation form to be selected. Ultimately, the Schumacher (1939) equation form was selected:

$$VAA = e^{0.606567 + 0.996428 \ln BAA + 0.691797 \ln Ht_D - 3.38497 (1/A)} \quad (7)$$

n = 124, adjusted R^2 = 0.9977

For both equations, parameter estimates had a significance of P <0.0014. Equation (7) (Schumacher 1939) was selected because it had a higher fit index, but it also produced better residual error behavior. The maximum residual error was reduced in magnitude, from 967 cubic feet for the equation from Hall and others (2020) to 186 cubic feet for equation (7), and the residuals from equation (7) formed a random pattern unlike those residuals from the equation found in Hall and others (2020) that exhibited a systematic relationship across the domain of the regressors. Cubic foot volume per acre estimates when using equation (7) are demonstrated in figure 5.

Simultaneous estimation of the equation system (Borders 1989) was not conducted. The procedure was not applied because sample sizes would have been reduced for the dominant height and survival equations because d.b.h. was not measured at younger ages, which resulted in smaller sample sizes when fitting basal area and volume per acre equations.

VALIDATION AND DISCUSSION

For all equations, parameter estimates exhibited logical and biologically sound magnitudes and signs.

The model system of equations (4) to (7) were tested using two sources of published data (Williston 1958, 1985). The first validation study

Table 3—Observed values by validation study and age, predicted values for the current system of equations, and predictions from Echinata

Study	Age	Trees per acre			Basal area			Dominant height		
		Obs	Current system Eq. (5)	Echinata	Obs	Current system Eq. (6)	Echinata	Obs	Current system Eq. (4)	Echinata
		~ ~ ~ ~ ~ ~ ~ ~ ~ ~ *number* ~ ~ ~ ~ ~ ~ ~ ~ ~ ~			~ ~ ~ ~ *square feet per acre* ~ ~ ~ ~			~ ~ ~ ~ ~ ~ ~ ~ *feet* ~ ~ ~ ~ ~ ~ ~ ~		
Williston (1958)	13	1,098	1,359	1,432	132	162	134	23	32	30
	23	1,064	1,241	1,153	231	228	198	48[a]	–	–
Williston (1985)	11	692	552	649	99	89	108	32	35	33
	14	692	534	629	133	110	131	38	44	42
	16	685	524	607	147	120	142	44	50	47
	26	499	482	443	162	144	165	67[a]	–	–

Echinata uses the model system presented by Smalley and Bailey (1974).

The Williston (1958) study was planted at 1,742 seedlings per acre; the Williston (1985) study was planted at 692 seedlings per acre.

For equations (5) and (6), estimates used dominant height estimates from equation (4). Similarly, for Echinata, trees per acre and basal area predictions were based on its internal dominant height equation.

Dominant height for the younger ages was predicted separately for the current model system and Echinata using their respective estimated site indices.

Obs = observed values from Williston (1958) or Williston (1985).

[a] Site index was determined separately for the current system (equation 3) and Echinata (Smalley and Bower 1971) by using the dominant height nearest to the base age of 25 years.

– = no data.

(Williston 1958) was from north Mississippi (Benton County) and data are average conditions of 10 plantations at ages 13 and 23 (table 3). Plantations were established at a planting density of 1,742 seedlings per acre. The second validation study was from north central Mississippi in a plantation established at a density of 692 seedlings per acre (Williston 1985). Since equation (7) only predicted total cubic foot volume (all d.b.h., to the tip of a tree), volume predictions could not be validated using these two studies due to differences in the minimum d.b.h. of trees included. Beyond that, both studies reported inside-bark volume to a 3-inch top.

Additionally, the equation system presented here was compared to predictions from an equation system presented by Smalley and Bailey (1974), using the two validation sources mentioned above. The equation system presented by Smalley and Bailey (1974) was incorporated into an Excel (Microsoft, Redmond, WA) spreadsheet to produce a timber decision support system (TDSS) entitled Echinata. Data used in model fitting were from Alabama, Georgia, and Tennessee highlands.

Trees per Acre

It appeared that for higher planting densities (Williston 1958), equation (5) overpredicted survival (table 3). Similarly, so did Echinata. Of 168 observations used in the model fitting of equation (5), only 44 observations were established using a planting density of 1,210 seedlings per acre or greater. A total of 18 observations were established with a planting density of 1,742 seedlings per acre, but all of them were from the Arkansas planting density study. A total of 26 observations had a planting density of 1,210 seedlings per acre, split between the Arkansas and Tennessee studies. Hence, the current model system may not be the most applicable for such high planting densities. For Williston (1985), equation (5) produced better predictions relative to Echinata at age 26, but Echinata produced superior estimates for the younger ages.

Basal Area per Acre

Similar to trees per acre, the current predictive model system overpredicted basal area for higher planting densities at age 13 (Williston 1958). However, Echinata produced a very accurate estimate. At age 23 for Williston (1958), the current model system produced a highly accurate estimate while Echinata underpredicted. Perhaps the current model system predicted accurate carrying capacity values for these high planting densities, just not mortality rates prior to when the carrying capacities are reached.

Based on trees per acre and basal area per acre, the current model system likely overpredicts for younger ages in high planting density plantations. It is difficult to tell if the overpredictions were exclusively due to a greater amount of lower planting density observations (only 18 observations used in model fitting exceeded a planting density of 1,500 seedlings per acre) within the model-fitting dataset or perhaps also due to improved survival rates in the current modeling effort because of better reforestation practices.

For Williston (1985), the current model system and Echinata had essentially the same amount of bias at age 11, but with opposing signs. However, Echinata produced very good estimates at ages 14, 16, and 26. Across all ages, the current model system underpredicted. Across all ages at the Williston (1985) site, at most, the current model system predicted incorrectly by 18.4 percent (age 16). For both sites when using the current model system, for ages in excess of 20 years, errors were 1.3 percent at age 23 (Williston 1958) and 11.1 percent at age 26 (Williston 1985).

Dominant Height

When using equation (4) the estimated site indices (base age 25) were 50 and 66 feet for Williston (1958) and (1985), respectively. Similar estimates were observed when using Echinata and Smalley and Bower (1971) and were 51 feet and 65 feet, respectively. When predicting dominant heights based on the estimated site indices (base age 25) calculated using the oldest ages of 23 (Williston 1958) and 26 (Williston 1985) years, equation (4) and Echinata had very similar predictions. However, Echinata, as a whole, produced slightly better results. The results of most interest, for current plantations, are likely those from Williston (1985). It appears that reasonable predictions will be obtained for dominant height regardless of whether equation (4) or Echinata are used. For Williston (1985), equation (4) had prediction errors of 15.8 percent or less.

One issue with Echinata is that for planting densities less than 500 seedlings per acre, Echinata will often produce illogical estimates of survival. Often, the number of trees per acre surviving at ages 10 and 15 will exceed the number planted. This behavior likely results from the model system by Smalley and Bailey (1974) being fit using only one observation with a planting density less than 500 seedlings per acre and the particular mortality equation form selected. Of 104 plots, 75 of them had planting densities in excess of 1,000 seedlings per acre. Conversely, for the current model dataset, half of the observations have planting densities less than 545 seedlings per acre, and a total of 16 plots have a planting density less than 500 seedlings per acre. For planting densities less than 500 seedlings per acre, predictions of survival are logical when using equation (5).

FUTURE WORK

Future work will need to concentrate on validating using observed plot-level data and not published, plot summarized data. Additionally, validation should be of more recently established plantations. Nonetheless, this brief validation analysis provides some idea of the predictive ability of the current model system [equations (3) to (7)].

This is an initial attempt to summarize shortleaf pine growth and yield literature; identify and consolidate sources of data; gain insight into the biological growth relationships associated with shortleaf pine plantations; and fit, examine, and test various equation functional forms. Future work needs to concentrate on a much more comprehensive dataset. The dataset should be geographically comprehensive in the western half of the Southeastern United States, but also comprehensive in terms of site conditions such as old-field versus cutover, bareroot versus container, genetics of seedlings, planting density, planting configuration, the inclusion of thinning, among many other considerations. It is hoped that this initial work will spark interest in sharing current datasets and establishment of additional studies.

LITERATURE CITED

Borders, B.E. 1989. Systems of equations in forest stand modeling. Forest Science. 35: 548–556. https://doi.org/10.1093/forestscience/35.2.548.

Clabo, D.C.; Clatterbuck, W., comps. and eds. 2015. A Tennessee landowner and practitioner guide for establishment and management of shortleaf and other pines. PB 1751 (Revised). Knoxville, TN: University of Tennessee, Agricultural Extension Service. 50 p. https://extension.tennessee.edu/publications/Documents/PB1751.pdf.

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

Dipesh, K.C.; Will, R.E.; Lynch, T.B. [and others]. 2015. Comparison of loblolly, shortleaf, and pitch x loblolly pine plantations growing in Oklahoma. Forest Science. 61: 540–547. https://doi.org/10.5849/forsci.14-004.

Guldin, J.M. 2019. Restoration of native fire-adapted southern pine-dominated forest ecosystems: diversifying the tools in the silvicultural toolbox. Forest Science. 65(4): 508–518. https://doi.org/10.1093/forsci/fxz005.

Hall, K.B., Stape, J.L.; Bullock, B.P. [and others]. 2020. A growth and yield model for *Eucalyptus benthamii* in the southeastern United States. Forest Science. 66: 25–37. https://doi.org/10.1093/forsci/fxz061.

Kushla, J.D. 2009. Afforestation in north Mississippi on retired farmland using *Pinus echinata*: first-year results. Southern Journal of Applied Forestry. 33(3): 142–144. https://doi.org/10.1093/sjaf/33.3.142.

Lenhart, J.D., Hackett, T.L.; Laman, C.J. [and others]. 1987. Tree content and taper functions for loblolly and slash pine trees planted on non-old-fields in east Texas. Southern Journal of Applied Forestry. 11(3):147–151. https://doi.org/10.1093/sjaf/11.3.147.

Little, E.L., Jr. 1971. Atlas of United States trees. Vol. 1. Conifers and important hardwoods. Misc. Publ. 1146. Washington, DC: U.S. Department of Agriculture, Forest Service. 320 p.

Omiyale, O. 1976. Spacing effects on planted pines of four species. M.S. thesis. Knoxville, TN: University of Tennessee. 60 p.

SAS Institute, Inc. 1989. SAS/ETS user's guide. Version 6. 4th ed. Cary, NC: SAS Institute, Inc. 560 p.

Schubert, M.R. 2001. Planting density effects on growth and survival of four pine species after 22 and 30 years on the eastern Highland Rim, TN. M.S. thesis. Knoxville, TN: University of Tennessee. 78 p.

Schubert, M.R.; Rennie, J.C.; Schlarbaum, S.E. 2004. Four pine species grown at four spacings on the eastern Highland Rim, Tennessee, after 30 years. In: Connor, K.F., ed. Proceedings of the 12th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 433–436.

Schumacher, F.X. 1939. A new growth curve and its application to timber-yield studies. Journal of Forestry. 37: 819–820. https://doi.org/10.1093/jof/37.10.815.

Smalley, G.W.; Bailey, R.L. 1974. Yield tables and stand structure for shortleaf pine plantations in Tennessee, Alabama, and Georgia highlands. Res. Pap. SO-97. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 57 p. https://www.fs.usda.gov/research/treesearch/841.

Smalley, G.W.; Bower, D.R. 1968. Volume tables and point-sampling factors for shortleaf pines in plantations on abandoned fields in Tennessee, Alabama, and Georgia highlands. Res. Pap. SO-39. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 13 p. https://www.fs.usda.gov/research/treesearch/595.

Smalley, G.W.; Bower, D.R. 1971. Site index curves for loblolly and shortleaf pine plantations on abandoned fields in Tennessee, Alabama, and Georgia highlands. Res. Note SO-126. New Orleans, LA: U.S. Department of Agriculture Forest Service. Southern Forest Experiment Station. 6 p.

Williston, H.L. 1958. Shortleaf versus loblolly pine in north Mississippi. Journal of Forestry. 56: 761. https://doi.org/10.1093/jof/56.10.761.

Williston, H.L. 1985. Growth and yield of planted loblolly and shortleaf pines in a north Mississippi creek bottom. Southern Journal of Applied Forestry. 9: 247–249. https://doi.org/10.1093/sjaf/9.4.247.

Woollons, R.C. 1998. Even-aged stand mortality estimation through a two-step regression process. Forest Ecology and Management. 105: 189–195. https://doi.org/10.1016/S0378-1127(97)00279-X.

Zhao, D.; Borders, B.; Wang, M.; Kane, M. 2007. Modeling mortality of second-rotation loblolly pine plantations in the Piedmont/Upper Coastal Plain and Lower Coastal Plain of the southern United States. Forest Ecology and Management. 252: 132–143. https://doi.org/10.1016/j.foreco.2007.06.030.

AN INDIVIDUAL-TREE MEASURE OF GROWING STOCK AND STAND DENSITY

Thomas J. Dean

ABSTRACT

Implicit in the concepts of growing stock and stand density is that they are functionally related to the number and size of the trees in a stand. If they are additive, they should be a function of the size and number of trees within a stand. To test this idea, data from a slash pine (*Pinus elliotii*) spacing and thinning study were fit to the nonlinear regression model $\frac{i_v}{i_h} = \beta_0 D_0^{\beta_1} + \epsilon$ where i_v = periodic annual increment of stem volume, i_h = periodic annual increment of stem height, D_0 = initial stem diameter at breast height, and β_0 and β_1 are parameters to be estimated. Previous research indicates that $\Sigma \frac{i_v}{i_h}$ is proportional to Dq^xN, the general mathematical form of Reineke's stand density index. Since Dq^xN can be closely approximated by $\sum D_0^x$, the relationship should also extend to single trees. The value of $\beta_0 D_0^{\beta_1}$ will indicate the value of a single tree to total growing stock or stand density. Results show that the power function fits the data well with little bias due to initial spacing and age. For these data the value of a single slash pine tree in terms of $\frac{i_v}{i_h}$ is $8.5 \times 10^{-4} D_0^{1.8}$. This result may be useful in furthering our understanding of how tree structure affects growth and also in determining the appropriate mix of stem diameters in uneven-aged stand structure.

INTRODUCTION

Each tree occupies space in a forest, contributes to forest growth, and competes with other trees for more space. The total space occupied increases with tree growth and tree density. Since the raw materials and resources for growth are distributed in space, the total space occupied by the trees is an important trait characterized in different ways depending on the function under consideration. For example, stand density refers to competition and growth limitations (Curtis 1970). Growing stock also refers to growth but from a yield perspective. Long (1985) related the two concepts through Langsaeter's curve whereby stand growth increases with growing stock or stand density as site occupancy increases to a maximum and individual-tree growth decreases with increased tree occupancy. While the relationship between tree occupancy and total stand growth varies between saturating to linear, the effect of increasing competition on individual tree growth is consistently negative (Dean and Baldwin 1996).

Stand growth, stand density, growing stock, and tree occupancy are naturally additive variables. Consider stand density. Reineke's stand density index (SDI) is $N\left(\frac{Dq}{25}\right)^x$ where N is tree density, Dq is quadratic mean diameter, and x is 1.6. This equation is approximately equal to $\Sigma\left(\frac{Dq}{25}\right)^x$ summed over trees per hectare (Ducey and Larson 2003). According to these equations, each tree's contribution to stand density is proportional to d.b.h.^{x}, where d.b.h. is diameter at breast height. However, adding another tree to the population has been thought to have a diminishing effect, especially with regards to competition.

Dean and others (2013) reported a standwise metric of tree growth that is consistently additive and proportional to the general form of Reineke's SDI, $N\left(\frac{Dq}{25}\right)^x$ with x ranging from 1.55 to 1.80 depending on the species. This transforms Reineke's SDI from an abstract measure of occupancy to a concrete measure of stand growth. Specifically, periodic increment

Thomas J. Dean, Professor, Louisiana State University Agricultural Center, School of Renewable Natural Resources, Baton Rouge, LA 70803

in stem volume divided by periodic increment in height growth ($\frac{i_v}{i_h}$) can be treated as a marginal height cost as discussed in Dean and others (2023). Its relationship with stem diameter is a consequence of the thickness of the growth layer needed to physically and physiologically support an additional meter of height. Since both total marginal height cost and SDI are additive, the contribution of a single tree to the overall growing stock or stand density should be given by the following power function:

$$\frac{i_v}{i_h} = \beta_0 D_0^{\beta_1} + \epsilon \quad (1)$$

where

i_v = periodic annual increment of stem volume

i_h = periodic annual increment of stem height

D_0 = initial stem diameter at breast height

β_0 and β_1 = parameters to be estimated

ε = the error in the model

The objective of this study was to fit individual-tree data to this regression model to determine whether the results obtained at the stand level by Dean and others (2013) are simply the sum of equation (1) across the trees in a stand. If so, then equation (1) provides the contribution of an individual tree to the total growing stock and stand density.

METHODS

Data for this analysis came from a slash pine (*Pinus elliotii*) spacing and thinning study conducted in cooperation with the U.S. Department of Agriculture, Forest Service, Southern Research Station, and the Louisiana Department of Agriculture and Forestry in Pineville, LA. The study design and the effects of spacing and thinning on plot growth were reported by Baldwin and others (1995) and Schexnayder and others (2002), respectively. The study was laid out in a randomized complete block design. Storm damage created an unequal replication of treatments within blocks. Sixty-four trees were planted in each plot at seven spacings (1.2 m by 1.2 m, 1.2 by 1.8 m, 1.8 m by 1.8 m, 1.8 m by 2.4 m, 2.4 m by 2.4 m, 3.1 m by 3.1 m, and 4.3 m by 4.3 m). All plots were thinned to a 35 percent of maximum SDI when the trees were 17 years old. Diameter at breast height (d.b.h.) (1.37 m) and total height were measured on all trees at ages 15 and 17 years (before thinning); thereafter, trees were measured every year until age 23. Periodic annual increment of stem volume and height were calculated for a 2-year period: 15–17 (before thinning), 18–20, and 21–23 years. Individual-tree stem volume was calculated with the equation from Lohrey (1985)

$$\ln V = -5.42 + 2.08 \ln d.b.h. + 0.846 \ln H \quad (2)$$

where

V = stem volume in cubic feet

d.b.h. is in inches

H = total height in feet

Data were fit to equation (1). To avoid autocorrelation due to repeatedly measured data on the same tree, one tree was randomly selected from each combination of block and spacing across measurement ages to produce a data matrix consisting of block, spacing, tree, age, d.b.h., and $\frac{i_v}{i_h}$. This was repeated 100 times to produce 100 estimates of β_0 and β_1. On average, each resampled dataset contained 25 trees. Goodness of fit was evaluated on the basis of bias with respect to initial spacing and age with residual plots. Each residual is the difference between the predicted and observed values of $\frac{i_v}{i_h}$ for each tree in the replicate. Bias was also evaluated with the mean of the residuals and the mean absolute difference between the observed and predicted values.

The implicit hypothesis tested with this analysis is that $\frac{i_v}{i_h}$ is related to a common power function of stem diameter regardless of initial spacing, thinning history, or age. Analysis of variance was used to test for spacing and age effects with a randomized complete block design with replicates treated as a blocking effect. Spacing and age effects on the means of the residuals were tested with a Tukey's test with the root mean square errors for the interaction between replicate and spacing and between replicate and age, respectively.

RESULTS

The null hypothesis that the 100 estimates of β_1 were normally distributed could not be rejected with the Shapiro-Wilk test ($P = 0.79$) (fig. 1). The mean value of β_1 was 1.79 and the median value was 1.84. The null hypothesis that spacing and age had no effect on the fit of $\frac{i_v}{i_h}$ to the power function of d.b.h. was rejected for both effects ($P < 0.001$). The mean values of the residuals grouped by spacing decreased from positive values for spacings closer than 2.4 m by 1.2 m to negative values for the wider spacings (table 1). Means comparison for age indicated that the means were significantly different for each age, starting negative for the 15- to 17-year age group to positive for the 21- to 23-year age group. While the mean of the residuals was significantly affected by initial spacing and age, these effects were not evident in scattergrams of the residuals plotted against the predicted values of $\frac{i_v}{i_h}$ (figs. 2 and 3). Within most spacings and age groups, a majority of the residuals were scattered around zero. The lack of significant bias in the fitted equation resulted in a close 1:1 relationship between predicted and observed plot sums of $\frac{i_v}{i_h}$ (fig. 4).

Table 1—Means testing for the effects of initial spacing and age grouping on the residuals of the fit of equation (1) to the 100 replications of the resampled slash pine data

	Group	n	Mean (m^3/m)
Spacing (m)	1.2 by 1.2	89	0.0068a
	1.2 by 1.8	354	0.0062a
	1.8 by 1.8	366	0.0087a
	1.8 by 2.4	274	0.0030a
	2.4 by 2.4	482	-0.0039b
	3.1 by 3.1	478	-0.0054bc
	4.3 by 4.3	292	-0.0099c
Age group (years)	15–17	787	-0.0106a
	18–20	842	-0.0031b
	21–23	706	0.0148b

Means in a group followed by the same letter were statistically similar.

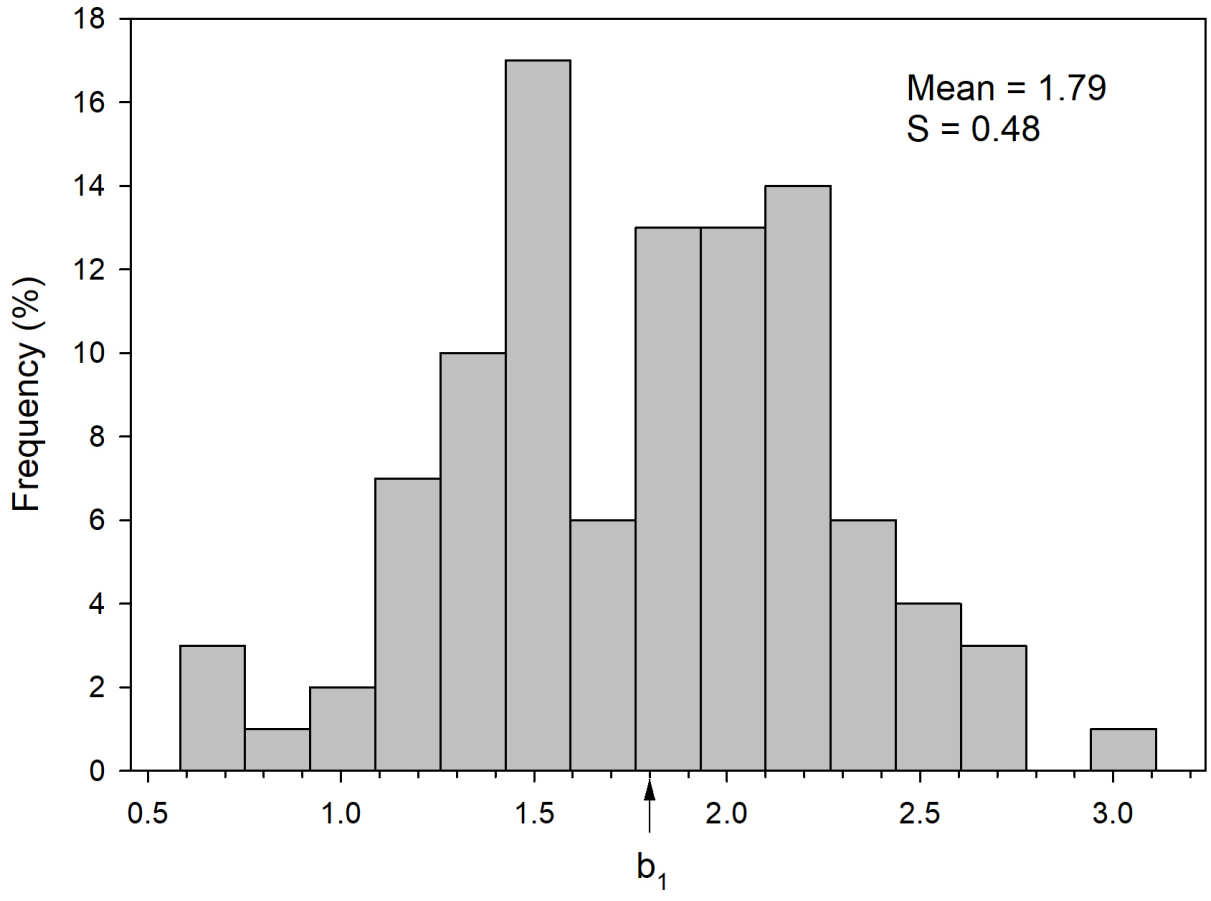

Figure 1—Histogram of the estimated values of β_1 for the nonlinear equation $\frac{i_v}{i_h} = \beta_0 D_0^{\beta_1} + \epsilon$ for 100 resampled datasets from the slash pine spacing and thinning study. Arrow shows the position of the mean value of β_1. S = standard deviation.

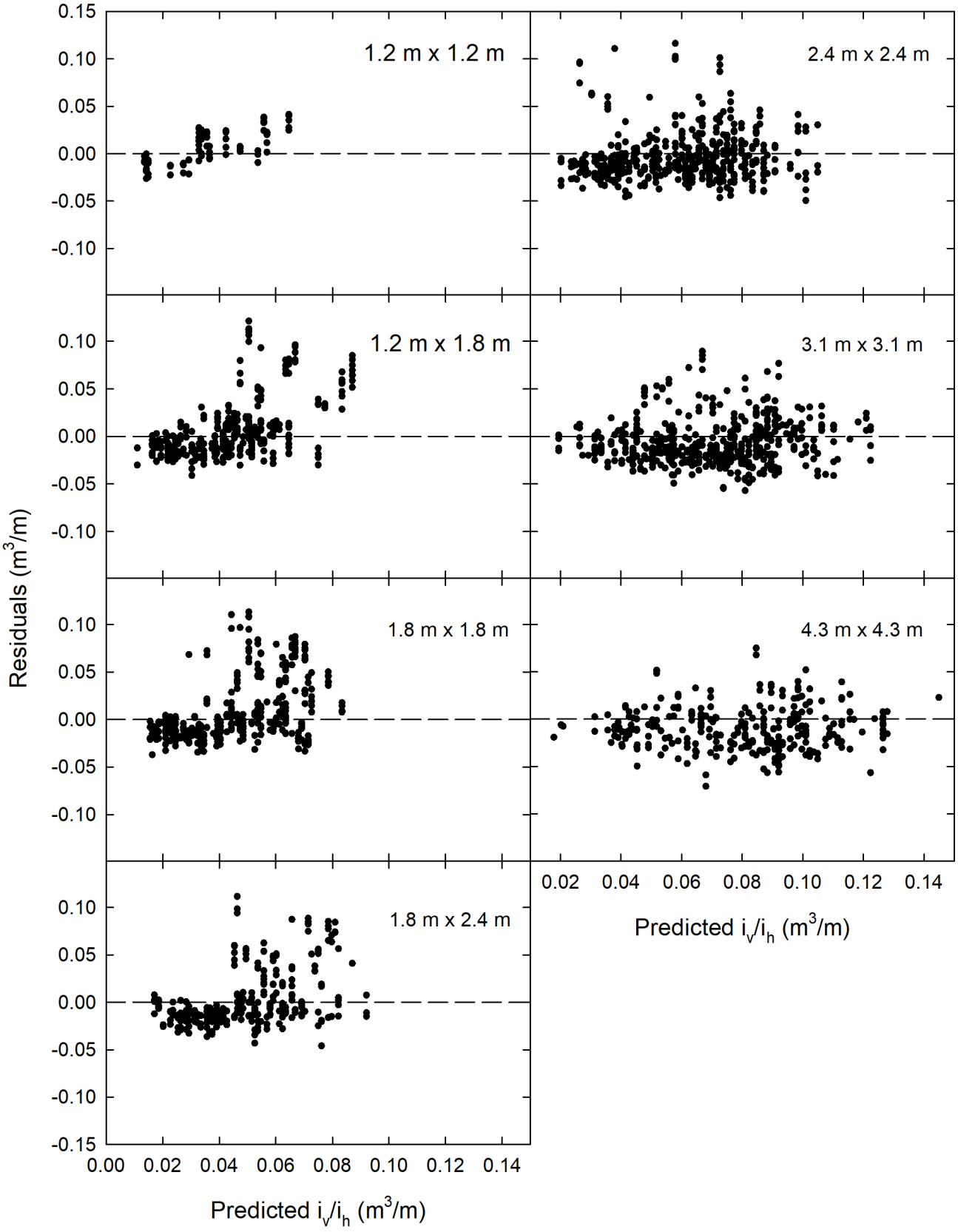

Figure 2—Residuals of fitting $\frac{i_v}{i_h} = \beta_0 D_0^{\beta_1} + \epsilon$ to the data from the slash pine spacing and thinning study by initial spacing.

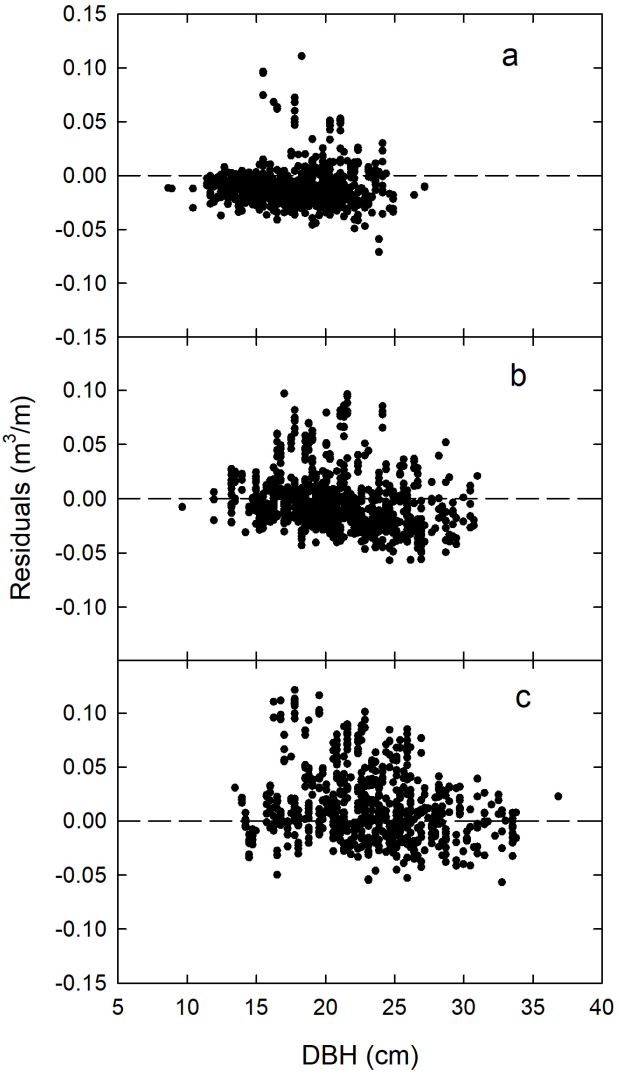

Figure 3—Residuals of fitting $\frac{i_v}{i_h} = \beta_0 D_0^{\beta_1} + \epsilon$ to the data from the slash pine spacing and thinning study by age group: (a) 15–17, (b) 18–20, and (c) 21–23.

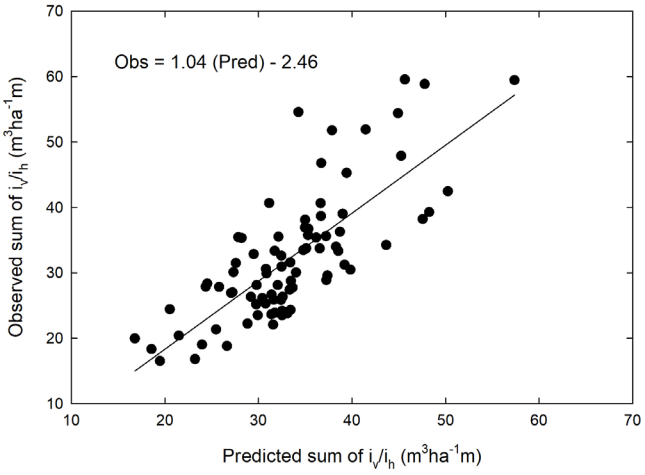

Figure 4—Observed plot sums of $\frac{i_v}{i_h}$ in relation to the plot sums predicted from the fit of equation (1) to the data from the slash pine spacing and thinning study.

DISCUSSION

Based on these results, each tree contributed 8.5×10^{-4}d. b. h.$^{1.8}$ to the stand density or growing stock of a stand. Although analysis of variance indicated that initial spacing and age have significant effects on this relationship, from a practical perspective the effects were negligible and can be ignored without loss of accuracy. According to Dean and others (2021), the relationship between marginal height cost and stem diameter may be affected by live-crown ratio, which can be influenced by prethinning spacing. Schexnayder and others (2002) note some residual effect of initial spacing on the growth response of these trees to thinning.

The mean value of β_1 was significantly less than 2 (t = 4, P <0.001), which would be the logical result of individual tree volume divided by tree height. This means that the ratio between stem volume increment and height increment is a fundamentally different relationship. As explained by Dean and others (2021), the predictable relationship between $\frac{i_v}{i_h}$ and d.b.h. seems to result from coordinated growth between height and stem volume where d.b.h. is the result of past height growth. The thickness of the growth layer needed to support additional height is manifest in stem diameter, and this increased girth is not simply proportional to the square of initial diameter.

The relationship between $\frac{i_v}{i_h}$ and d.b.h. may be a better method for setting target diameter distributions for uneven-aged stands compared to current methods that rely on the q factor, a somewhat arbitrary geometric mathematical progression. The q factor has a long and controversial history in creating the target structure in uneven-aged stands (Dean 2019) because it is more of an artifact of past disturbances rather than a biological constraint on development (Guldin 1991). Since the stand sum of $\frac{i_v}{i_h}$ is linearly related to stand density, equal proportions of the sum can be allocated across a range of diameter classes. This follows the same reasoning used by Long and Daniel

(1990) in allocating equal proportions of SDI to create the diameter distributions for Douglas-fir (*Pseudotsuga menziesii*). The number of trees allocated to each diameter class would be the expected value of $\frac{i_v}{i_h}$ calculated with the fitted values of equation (1) for each diameter class divided into the sum of $\frac{i_v}{i_h}$ allocated to that diameter class. If the periodic height increment for a diameter class is known, the periodic volume increment for that class can also be calculated. The total periodic volume increment for the stand would be the sum of the periodic volume increment across the diameter classes.

ACKNOWLEDGMENTS

I thank the U.S. Department of Agriculture, Forest Service, Southern Research Station and the Office of Forestry of the Louisiana Department of Agriculture and Forestry for their support and its cooperation in this study establishing and helping in data collection. This project was supported in part with McIntire-Stennis funds under LAB94512. This project was published with the approval of the Director of the Louisiana Agricultural Experiment Station as publication No. 2023-241-38936.

LITERATURE CITED

Baldwin, V.C., Jr.; Dean, T.J.; Ferguson, R.B. 1995. A comparison of density indices in controlling growing stock in a slash pine spacing-thinning study. In: Edwards, M.B., comp. 1995. Proceedings of the eighth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-1. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 461–466.

Curtis, R.O. 1970. Stand density measures: an interpretation. Forest Science. 16(4): 403–414. https://doi.org/10.1093/forestscience/16.4.403.

Dean, T.J. 2019. Advances in stand management and regeneration. In: Stanturf, J.A., ed. Achieving sustainable management of boreal and temperate forests. Cambridge, UK: Burleigh Dodds Science Publishing: 447–488. https://doi.org/10.19103/AS.2019.0057.17.

Dean, T.J.; Baldwin, V.C., Jr. 1996. Growth in loblolly pine plantations as a function of stand density and canopy properties. Forest Ecology and Management. 82: 49–58. https://doi.org/10.1016/0378-1127(95)03694-6.

Dean, T.J.; D'Amato, A.W.; Palik, B.J. [and others]. 2021. A direct measure of stand density based on stand growth. Forest Science. 67(1): 103–115. https://doi.org/10.1093/forsci/fxaa038.

Dean, T.J.; Harrington, C.A.; D'Amato, A.; Palik, B.J. 2023. Response of marginal height costs and marginal height benefits to competition. Forest Ecology and Management. 528: 120647. 12 p. https://doi.org/10.1016/j.foreco.2022.120647.

Dean, T.J.; Roberts, S.D.; Seymour, R.S. 2013. Toward developing a direct relation between gross volume increment and stand density. Canadian Journal of Forest Research. 43: 852–860. https://doi.org/10.1139/cjfr-2013-0038.

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. https://doi.org/10.1093/wjaf/18.3.179.

Guldin, J.M. 1991. Uneven-aged *BDq* regulation of Sierra Nevada mixed conifers. Western Journal of Applied Forestry. 6(2): 27–32. https://doi.org/10.1093/wjaf/6.2.27.

Lohrey, R.E. 1985. Stem volume, volume ratio, and taper equations for slash pine in the West Gulf Region. In: Shoulders, E., ed. Proceedings of the third biennial southern silvicultural research conference. Gen. Tech. Rep. SO-54. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station: 451–459.

Long, J.N. 1985. A practical approach to density management. Forestry Chronicle. 61: 23–27. https://doi.org/10.5558/tfc61023-1.

Long, J.N.; Daniel, T.W. 1990. Assessment of growing stock in uneven-aged stands. Western Journal of Applied Forestry. 5(3): 93–96. https://doi.org/10.1093/wjaf/5.3.93.

Schexnayder, J.C.; Dean, T.J.; Baldwin, V.C., Jr. 2002. Diameter growth of a slash pine spacing study five years after being thinned to a constant stand density index. In: Outcalt, K.W., ed. Proceedings of the eleventh biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-48. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 156–158.

BIOMASS ESTIMATES FOR NATURAL-ORIGIN HARDWOOD SAPLINGS OF SOUTHEASTERN ARKANSAS

Mohammad M. Bataineh

ABSTRACT

Biomass estimates are integral to carbon and feed stocks, fuel loads, and primary productivity. Understories of mature, natural-origin, pine (*Pinus* spp.) and pine-hardwood stands in southeastern Arkansas are dominated by shade-tolerant hardwoods which are poorly represented by existing biomass equations. Destructive sampling of 120 saplings [(0.5–12.2 cm in diameter at breast height (d.b.h.)] was used to develop biomass models for six hardwood species. A linear, two-parameter logarithmic model was used with d.b.h. as the main predictor. Local equations were compared to standard national-scale equations. Although local equations did not outperform standard equations, use of standard equations may result in estimation error of 4–5 kg. The Updated standard equations (Chojnacky and others 2014) had better fit to local data than the Jenkins standard equations (Jenkins and others 2003). A pooled biomass model provided a good fit to observations (R^2 = 0.97) indicating the potential for use of a general equation.

INTRODUCTION

Biomass estimates are essential in carbon and biofuel stock, fuel loads, and primary productivity determinations. Inventories of carbon and biofuel stock rely on individual tree biomass, calculated from allometric equations, that are subsequently aggregated at forest and landscape scales to determine current and projected conditions (Bataineh 2020, Heath and others 2009). Tree biomass derived from allometric equations is also fundamental in calibrating remote sensing datasets to provide biomass maps at regional and landscape scales (Vorster and others 2020). Biomass map validation commonly assumes that plot-level estimates are error free and can be directly compared to biomass predicted from spectral, textural, or vegetation index variables (Lu 2006). However, errors associated with allometric equations are propagated at higher aggregation levels. Consequently, uncertainty associated with allometric equations may even exceed that associated with calibration of remote sensing datasets, especially when an independent dataset is used for the validation (Vorster and others 2020). Although locally developed biomass equations better reflect tree species characteristics and growing conditions, and thus may be expected to be more accurate (Vorster and others 2020, Weiskittel and others 2015), local equations do not necessarily outperform national or standard equations (Nelson and others 2014). In contrast, standard equations may not be applicable to the intended locality leading to significant estimation errors (Nelson and others 2012, Wang and others 2002).

Allometric equations for tree biomass previously focused on large trees of commercial value (Bullock and Burkhart 2003). As interest in forest carbon and biofuel inventories grew, allometric equations for small-diameter trees were developed and expanded to encompass more species, including those commonly dominant in the understories of natural-origin pine (*Pinus* spp.) and pine-hardwood stands (Bragg and Scott 2014). Consistent, national scale, diameter-based allometric equations were later developed to support carbon inventories (Jenkins and others 2003). In these national-scale equations, hereafter referred to as Jenkins standard equations, pseudodata from published diameter-based

Mohammad M. Bataineh, Associate Professor, College of Forestry, Agriculture, and Natural Resources, University of Arkansas at Monticello, Monticello, AR 71656

biomass equations were used to develop more generalizable models. Forty published equations were combined in the development of Jenkins standard equations for 21 hardwood species (i.e., mixed hardwood group) that included winged elm (*Ulmus alata*), flowering dogwood (*Cornus florida*), white ash (*Fraxinus americana*), and blackgum (*Nyssa sylvatica*) (Jenkins and others 2003). The 10 Jenkins standard equations were later refined and updated, hereafter referred to as updated equations, to provide 35 allometric equations, 18 updated and refined for hardwood species (Chojnacky and others 2014). Jenkins and the updated standard equations minimum diameters were 2.5 and 3 cm, respectively.

Although Jenkins and the updated equations are best validated using large, representative biomass data encompassing tree species ranges, comparisons with independent datasets not included in the standard equations would highlight the applicability of these equations for a specific locality and determine the need for locally developed biomass equations. The objectives of this study were to 1) develop a local set of total aboveground biomass equations in dry weight terms for saplings ranging in diameter from 0.5 to 12.2 cm of six hardwood species commonly present in the understories of mature, natural-origin pine and pine-hardwood stands of southeastern Arkansas, and 2) compare local models and observations against standard models (Jenkins and updated) to verify standard models using an independent, yet limited, dataset.

MATERIALS AND METHODS

Study Area

Two unmanaged, mature (80–120 years old), natural-origin pine-hardwood stands in Cleveland County, Arkansas, constituted the study area. The stands had a similar disturbance history with a harvest in the late 19th century followed by no management intervention. Stand overstories were composed of a mixture of pine-hardwood with shortleaf pine (*P. echinata*), loblolly pine (*P. taeda*), southern red oak (*Quercus falcata*), cherrybark oak (*Q. pagoda*), post oak (*Q. stellata*), and white oak (*Q. alba*) as main components (Woods and others 2004). Dense hardwood-dominated understories were present with an abundance of shrubs and vines. Stands were 11 km apart. The two stands are within the Upper West Gulf Coastal Plain, which is described as rolling plains punctuated with fluvial terraces, bottomlands, and low cuestas. Climate of the area is warm in summers and mild in winters with 135 cm of mean annual precipitation, most of which occurs from November through May, and a mean annual temperature of 17 °C (Larance and others 1968). Annual frost-free period is 236 days. Forests cover the landscape with loblolly pine, shortleaf pine, sweetgum (*Liquidambar styraciflua*), oak, gum (*Nyssa* spp.), and hickory (*Carya* spp.) as main species (Woods and others 2004). Elevation is between 11 and 213 m above sea level. Soils series of the study area include Savannah (siliceous, semiactive, thermic Typic Fragiudults), which are described as very deep, moderately well drained, and slowly permeable on uplands and terraces; Wehadkee (mixed, active, nonacid, thermic Fluvaquentic Endoaquepts), which are very deep, poorly to very poorly drained along streams and on floodplains in bottomlands; Boswell (mixed, active, thermic Vertic Paleudalfs), which are very deep, moderately well drained, and very slowly permeable fine sandy loams on uplands; Caddo (siliceous, active, thermic Typic Glossaqualfs), which are described as very deep, poorly drained, acidic soils, with little organic matter in upland locations; and Stough (siliceous, semiactive, thermic Fragiaquic Paleudults), which are poorly drained, acidic, of low fertility, coarse loams along stream terraces (Larance and others 1968).

Sampling Design and Measurements

In 2015, 120 saplings were destructively sampled. Six species were selected, namely: flowering dogwood (*Cornus florida*), white ash (*Fraxinus americana*), American holly (*Ilex opaca*), blackgum (*Nyssa sylvatica*), eastern hophornbeam (*Ostrya virginiana*), and winged elm (*Ulmus alata*). Saplings ranged in diameter at breast height (d.b.h.) from 0.5 to 12.2 cm and were 1.5 to 14.4 m tall (table 1). Saplings were selected randomly from each d.b.h. distribution quintile for each stand to ensure a representative sample across the available size ranges. For each of six

Table 1—Characteristics of saplings sampled from two unmanaged stands in Cleveland County, AR, for each of six hardwood species commonly present in the understories of mature, natural-origin pine and pine-hardwood stands of southeastern Arkansas

Species	n	d.b.h.	GLD	Height
		cm	*cm*	*m*
Flowering dogwood (*Cornus florida*)	20	6.2 (1.0–12.2)	8.5 (2.0–16.5)	6.2 (2.3–9.1)
White ash (*Fraxinus americana*)	20	5.4 (0.5–11.4)	7.8 (1.8–16.0)	7.3 (1.8–14.4)
American holly (*Ilex opaca*)	20	2.0 (0.5–4.3)	3.2 (1.3–5.6)	2.8 (1.5–5.2)
Blackgum (*Nyssa sylvatica*)	20	6.2 (1.0–10.7)	8.9 (2.0–15.5)	6.8 (2.3–10.6)
Eastern hophornbeam (*Ostrya virginiana*)	20	3.3 (0.5–10.7)	5.3 (1.5–15.5)	5.3 (1.6–12.2)
Winged elm (*Ulmus alata*)	20	4.5 (1.5–12.2)	6.8 (2.5–17.0)	5.9 (3.0–12.4)

Mean values with ranges in parenthesis.
d.b.h. = diameter at breast height; GLD = groundline diameter.

species, two saplings were sampled within each quintile for a total of 10 individuals per stand, and a total of 20 individuals per species. Saplings near roads, in decline, or showing damage symptoms were not included in sampling. Saplings were measured for d.b.h., groundline diameter, diameter at crown base, height to crown base, and total height. Saplings were cut at the groundline above the root collar. Total aboveground biomass was defined as all aboveground material including foliage, branches, and stem. Cut saplings were oven dried at 100 °C to a constant weight.

Analytical Approach

To develop a local set of total aboveground biomass equations in dry weight terms, the two-parameter logarithmic model was used. This is the same model used in Jenkins and the updated standard equations and is appealing for simplicity in fitting procedures and rescaling of variance to address heteroscedasticity (Baskerville 1972, Parresol 1999). Although the two-parameter logarithmic model is reported to result in underestimates of biomass when antilogarithms are used to backtransform to original units, correction of backtransformed biomass to remedy this downward bias tends to overestimate the true bias and is sample-size dependent (Jenkins and others 2003, Parresol 1999). To keep with Jenkins and the updated standard equations, in this study of comparing local and standard equations, no correction factors were used (Jenkins and others 2003). Local biomass equations were fit for pooled species data as well as each species separately using natural logarithm transformed biomass and d.b.h. values. Scatter plots depicting total aboveground biomass against d.b.h. and other measured dimensions were examined to screen data and determine adequacy of model form (Parresol 1999). All equations were fit using the lm function of the R base package version 3.0.2 (R Development Core Team 2013). No indication of patterns in residual plots were detected as evidence of violation of assumptions of normality and homogeneity of variance, thus no weighting of observations was necessary (Baskerville 1972).

To ensure use of the same model form, transformed model parameters were used to compare local and standard models (Jenkins and updated). Local and standard models were also compared graphically. Observations used to develop local models were used in calculating accuracy and precision metrics for the standard models. Accuracy was evaluated by calculating the root mean square error (RMSE), whereas precision was evaluated using mean absolute error (MAE) for standard models (Vorster and others 2020). Jenkins mixed hardwood group equation was used to estimate total aboveground biomass for five of the six species examined in this study (Jenkins and others 2003). Eastern hophornbeam with a wood specific gravity of 0.63 (Miles and Smith 2009) was estimated using Jenkins hard maple/oak/hickory/beech group equation. In estimating biomass using the updated equations, white ash and eastern hophornbeam had their own generalized equations, whereas the remaining four species were estimated using the

mixed hardwood group equation, representing members of the Cornaceae, Ericaceae, Lauraceae, Platanaceae, Rosaceae, and Ulmaceae (Chojnacky and others 2014).

RESULTS AND DISCUSSION

Total aboveground biomass equations were developed for saplings (0.5–12.2 cm in d.b.h.) of six hardwood species commonly present in the understories of mature, natural-origin, pine and pine-hardwood stands of southeastern Arkansas to better reflect local species characteristics and growing conditions (table 2). Goodness of fit measures for each of the individual species' equations indicated overall good model fits with R^2 values ranging from 0.93 to 0.99 (table 2). RMSE values ranged from 0.16 to 0.32 kg, indicating a low level of uncertainty in predicting total aboveground biomass of hardwood saplings. All models had low MAE values not exceeding 0.26 kg. A pooled species equation, n = 120, produced a good model fit with RMSE and MAE values within the range observed for individual species equations, albeit closer to maximum values of the range (table 2). Pooled species equation coefficients were within the range observed for individual species equations (table 2), indicating that a pooled model may be sufficient for estimating aboveground biomass of those six hardwood species (fig. 1).

Local aboveground biomass equation coefficients were similar to those provided by Jenkins and updated (Chojnacky and others 2014, Jenkins and others 2003) standard equations (tables 2 and 3). Local intercept coefficients were, on average, 0.8 higher than the standard equations. Local slope parameter values were, on average, 0.4 lower than those provided by the standard equations. Both Jenkins and updated standard equations showed a good fit to the local data collected in this study, despite having a slightly higher minimum d.b.h. threshold (fig. 2). Although local equations inherently had lower RMSE values, standard equations RMSE values ranged from 0.72 to 5.17 kg for Jenkins equations and from 0.50 to 3.56 kg for updated equations (table 3). MAE values ranged from 0.53 to 3.26 kg for Jenkins equations and from 0.37 to 2.00 kg for updated equations. The updated equations appeared to better fit the local data than the Jenkins standard equations (fig. 2). Generally, standard equations appeared to be applicable to the data collected in this study, in contrast to that reported elsewhere (Nelson and others 2012, Wang and others 2002). Local equations did not seem to drastically outperform national or standard equations at the individual level, which is similar to that reported elsewhere (Nelson and others 2014). However, validation of standard equations using an independent, yet limited, dataset showed that use of standard equations may result in small-diameter tree biomass estimation error of upwards of 4–5 kg, which may be propagated at higher aggregation levels of stands and landscapes (Vorster and others 2020).

Table 2—Total aboveground biomass model {ln(biomass in kg = β_0 + β_1 ln[d.b.h in cm])} coefficients, standard errors, *P*-values, and goodness of fit measures for saplings (0.5–12.2 cm in d.b.h) of six hardwood species commonly present in the understories of mature, natural-origin pine and pine-hardwood stands of southeastern Arkansas

Species	β_0 (SE)	β_1 (SE)	*P*	R^2	RMSE	MAE
Flowering dogwood (*Cornus florida*)	-1.59 (0.11)	2.17 (0.06)	<0.001	0.99	0.16	0.12
White ash (*Fraxinus americana*)	-1.46 (0.15)	2.06 (0.09)	<0.001	0.97	0.30	0.26
American holly (*Ilex opaca*)	-1.25 (0.09)	1.78 (0.11)	<0.001	0.93	0.32	0.25
Blackgum (*Nyssa sylvatica*)	-1.82 (0.20)	2.15 (0.11)	<0.001	0.95	0.26	0.17
Eastern hophornbeam (*Ostrya virginiana*)	-1.16 (0.09)	1.95 (0.07)	<0.001	0.98	0.28	0.22
Winged elm (*Ulmus alata*)	-1.71 (0.15)	2.20 (0.10)	<0.001	0.96	0.22	0.18
Pooled species	-1.34 (0.05)	1.98 (0.03)	<0.001	0.97	0.30	0.23

Coefficient estimates and standard errors (SE) in parenthesis.
d.b.h. = diameter at breast height; RMSE = root mean square error; MAE = mean absolute error.

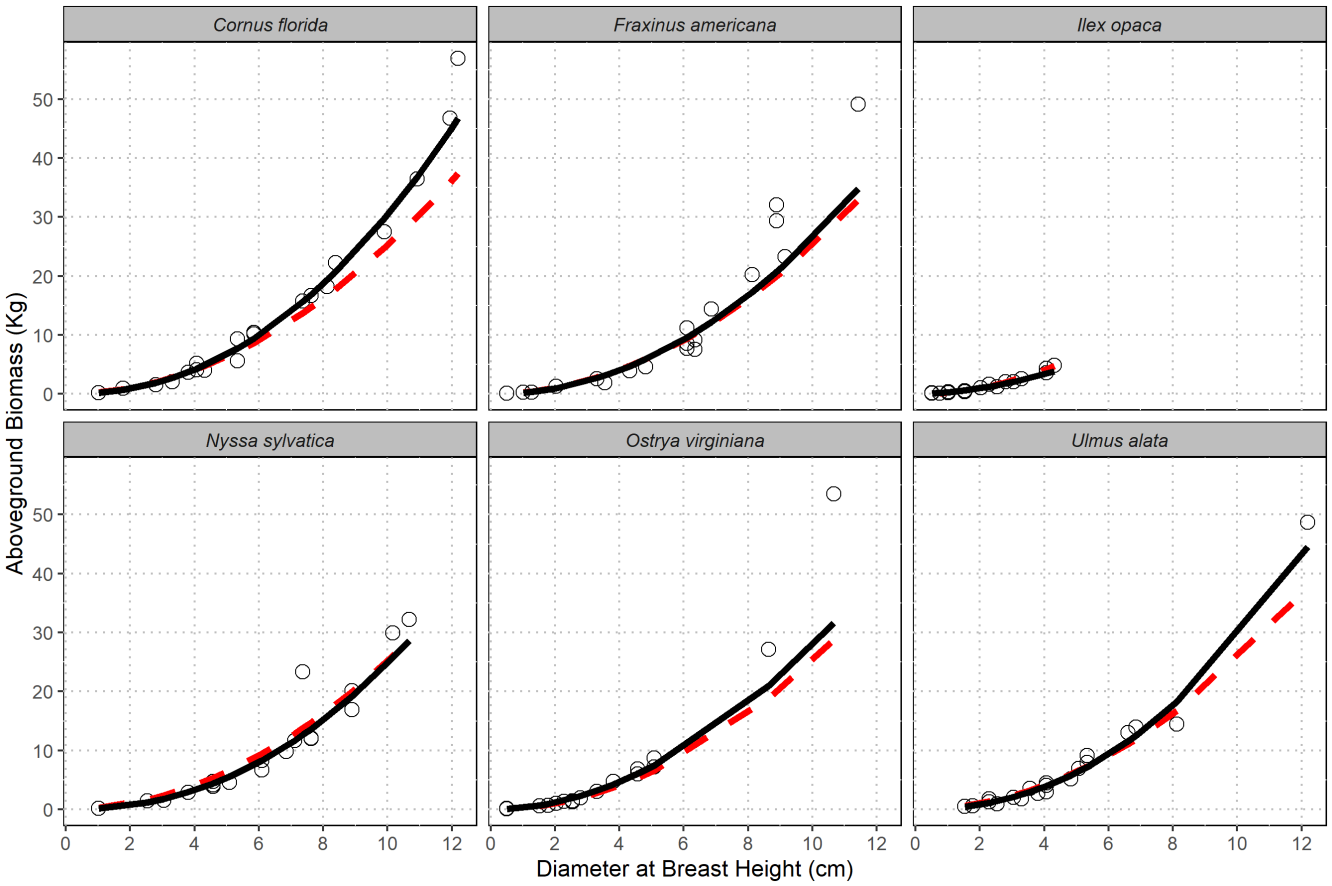

Figure 1—Individual (solid black line) and pooled (dashed red line) species models of total aboveground biomass for saplings (0.5–12.2 cm in d.b.h) of six hardwoods commonly present in the understories of mature, natural-origin, pine and pine-hardwood stands of southeastern Arkansas.

Table 3—Jenkins and updated standard equations {*ln*(biomass in kg = $\beta_0 + \beta_1$ ln[d.b.h in cm])} coefficients as well as precision and accuracy metrics for saplings (0.5–12.2 cm in d.b.h) of six hardwood species commonly present in the understories of mature, natural-origin pine and pine-hardwood stands of southeastern Arkansas

	Species	β_0	β_1	RMSE	MAE
Jenkins equations	**Flowering dogwood (*Cornus florida*)**	-2.48	2.48	4.66	3.26
	White ash (*Fraxinus americana*)	-2.48	2.48	5.17	2.98
	American holly (*Ilex opaca*)	-2.48	2.48	0.72	0.53
	Blackgum (*Nyssa sylvatica*)	-2.48	2.48	3.41	1.83
	Eastern hophornbeam (*Ostrya virginiana*)	-2.01	2.43	2.57	1.02
	Winged elm (*Ulmus alata*)	-2.48	2.48	2.41	1.68
	Pooled species	-2.48	2.48	4.10	2.16
Updated equations	**Flowering dogwood (*Cornus florida*)**	-2.21	2.41	3.07	1.94
	White ash (*Fraxinus americana*)	-1.84	2.35	2.49	1.76
	American holly (*Ilex opaca*)	-2.21	2.41	0.50	0.37
	Blackgum (*Nyssa sylvatica*)	-2.21	2.41	3.56	2.00
	Eastern hophornbeam (*Ostrya virginiana*)	-2.27	2.53	2.80	1.30
	Winged elm (*Ulmus alata*)	-2.21	2.41	1.54	1.16
	Pooled species	-2.21	2.41	3.29	1.64

d.b.h. = diameter at breast height; RMSE = root mean square error; MAE = mean absolute error.

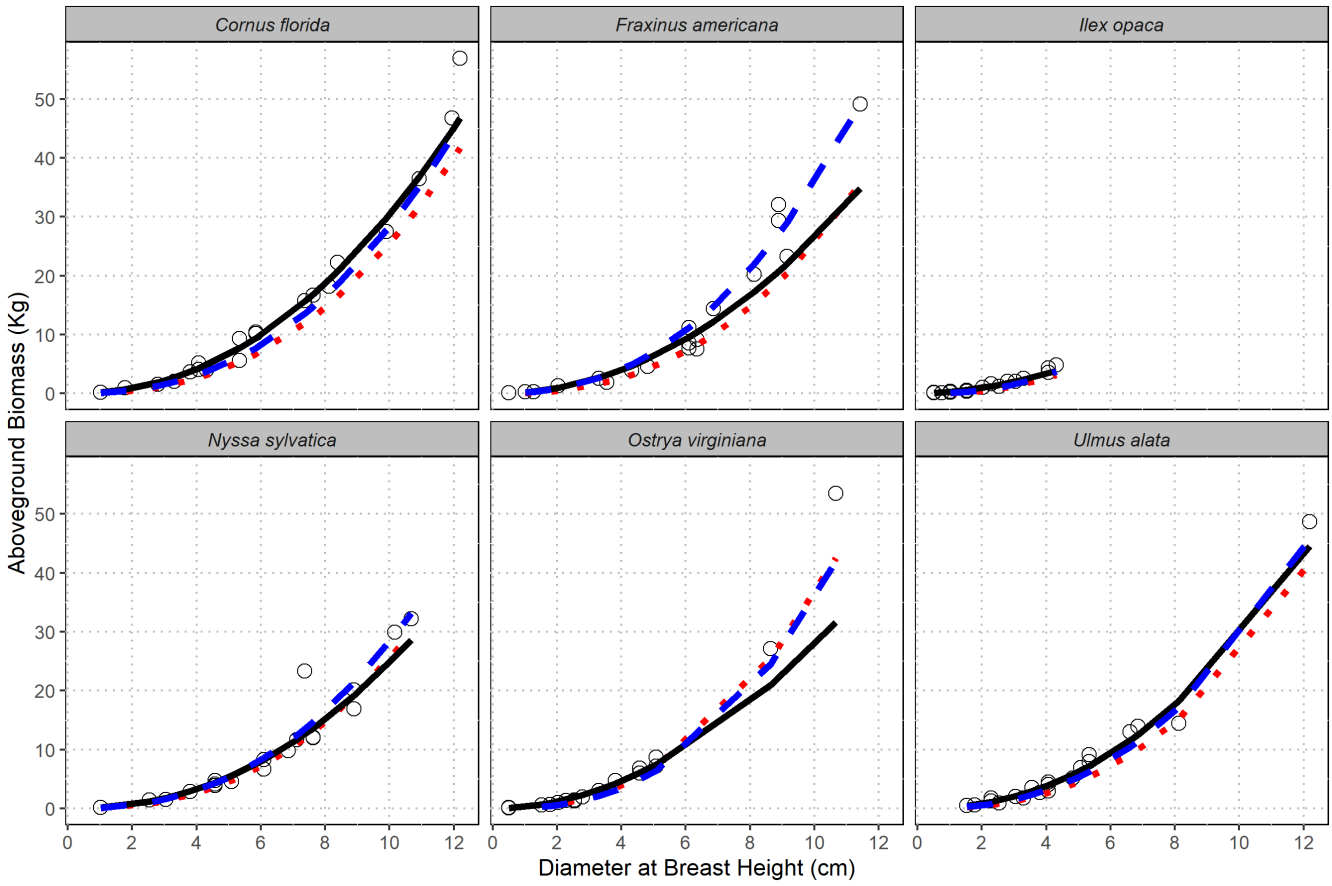

Figure 2—Locally developed total aboveground biomass equations (solid black line) compared to Jenkins standard equations (dotted red line) and updated standard equations (dashed blue line) for saplings (0.5–12.2 cm in d.b.h) of six hardwood species commonly present in the understories of mature, natural-origin, pine and pine-hardwood stands of southeastern Arkansas.

CONCLUSIONS

To better estimate and map regional carbon and biofuel stock, inventories should include understories of mature, natural-origin pine and pine-hardwood stands such as those in southeastern Arkansas. Shade-tolerant hardwood species constituting these understories are poorly represented by existing biomass equations. Before an existing biomass equation can be applied, errors associated with estimating individual tree biomass must be evaluated to determine the applicability of existing equations to the intended locality. A set of local total aboveground biomass equations were developed for small-diameter hardwoods of six commonly occurring understory species in mature, natural-origin, pine and pine-hardwood stands. Local models did not appear to drastically outperform standard, national-scale, diameter-based allometric equations at the individual level. The updated standard equations appeared to provide better fit than the Jenkins standard equations at local scales like those presented in this study. Locality specific applications, however, may still use the provided local equations to better represent intended populations.

ACKNOWLEDGMENTS

Thanks to Thomas Pagles, a former graduate student, who collected these data as part of an unfinished Master's thesis. Thanks to the Garner family and Les Reid for their support and for providing site access. Special thanks to Lynne Thompson, John Dennis, and Tyler Corbin for reviewing an earlier version of this paper.

LITERATURE CITED

Baskerville, G.E. 1972. Use of logarithmic regression in the estimation of plant biomass. Canadian Journal of Forestry. 2: 49–53. https://doi.org/10.1139/x72-009.

Bataineh, M.M. 2020. Above-ground carbon stock and distribution in managed and unmanaged mature, natural-origin, pine-hardwood forest stands. In: Bragg, D.C.; Koerth, N.E; Holley, A.G., eds. Proceedings of the 20th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-253. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 279–286.

Bragg, D.C.; Scott, D.A. 2014. A preliminary aboveground live biomass model for understory hardwoods from Arkansas, Louisiana, and Mississippi. In: Groninger, J.W.; Holzmueller, E.J.; Nielsen, C.K.; Dey, D.C., eds. Proceedings of the 19th Central Hardwood Forest conference. Gen. Tech. Rep. NRS-P-142. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 251–260.

Bullock, B.P.; Burkhart, H.E. 2003. Equations for predicting green weight of loblolly pine trees in the South. Southern Journal of Applied Forestry. 27(2): 153–159. https://doi.org/10.1093/sjaf/27.3.153.

Chojnacky, D.C.; Heath, L.S.; Jenkins, J. 2014. Updated generalized biomass equations for North American tree species. Forestry. 87(1): 129–151. https://doi.org/10.1093/forestry/cpt053.

Heath, L.S.; Hansen, M.H.; Smith, J.E. [and others]. 2009. Investigation into calculating tree biomass and carbon in the FIADB using a biomass expansion factor approach. In: McWilliams, W.; Moisen, G.; Czaplewski, R., comps. 2008 Forest Inventory and Analysis (FIA) symposium. Proc. RMRS-P-56CD. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 26 p.

Jenkins, J.C.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A. 2003. National-scale biomass estimators for United States tree species. Forest Science. 49(1): 12–35. https://doi.org/10.1093/forestscience/49.1.12.

Larance, F.C.; Jay, J.E.; Elder, W.R. [and others]. 1968. Soil survey of Cleveland County, Arkansas. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service. 71 p.

Lu, D. 2006. The potential and challenge of remote sensing-based biomass estimation. International Journal of Remote Sensing. 27(7): 1297–1328. https://doi.org/10.1080/01431160500486732.

Miles, P.D.; Smith, W.B. 2009. Specific gravity and other properties of wood and bark for 156 tree species found in North America. Res. Note NRS-38. Newton Square, PA: U.S. Department pf Agriculture Forest Service, Northern Research Station. 35 p. https://doi.org/10.2737/NRS-RN-38.

Nelson, A.S.; Weiskittel, A.R.; Wagner, R.G.; Saunders, M.R. 2012. Verification of the Jenkins and FIA sapling biomass equations for hardwood species in Maine. In: Morin, R.S.; Liknes, G.C., comps. Moving from status to trends: Forest Inventory and Analysis (FIA) symposium 2012. Gen. Tech. Rep. NRS-P-105. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 373–377.

Nelson, A.S.; Weiskittel, A.R.; Wagner, R.G.; Saunders, M.R. 2014. Development and evaluation of aboveground small tree biomass models for naturally regenerated and planted species in eastern Maine, U.S.A. Biomass and Bioenergy. 68: 215–227. https://doi.org/10.1016/j.biombioe.2014.06.015.

Parresol, B.R. 1999. Assessing tree and stand biomass: a review with examples and critical comparisons. Forest Science. 45(4): 573–593. https://doi.org/10.1093/forestscience/45.4.573.

R Development Core Team. 2013. R: a language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing.

Vorster, A.G.; Evangelista, P.H.; Stovall, A.E.L.; Ex, S. 2020. Variability and uncertainty in forest biomass estimates from the tree to landscape scale: the role of allometric equations. Carbon Balance and Management. 15: 8. 20 p. https://doi.org/10.1186/s13021-020-00143-6.

Wang, J.R.; Zhong, A.L.; Kimmins, J.P. 2002. Biomass estimation errors associated with the use of published regression equations of paper birch and trembling aspen. Northern Journal of Applied Forestry. 19(3): 128–136. https://doi.org/10.1093/njaf/19.3.128.

Weiskittel, A.R.; MacFarlane, D.W.; Radtke, P.J. [and others]. 2015. A call to improve methods for estimating tree biomass for regional and national assessment. Journal of Forestry. 113(4): 414–424. https://doi.org/10.5849/jof.14-091.

Woods, A.J.; Foti, T.L.; Chapman, S.S. [and others]. 2004. Ecoregions of Arkansas (color poster with map, descriptive text, summary tables, and photographs). Reston, VA: U.S. Geological Survey. https://gaftp.epa.gov/EPADataCommons/ORD/Ecoregions/ar/ar_eco_lg.pdf. [Date accessed: October 17, 2023].

A GROWTH AND YIELD SYSTEM OF DIFFERENTIAL EQUATIONS FOR SLASH PINE PLANTATIONS INCLUDING RESPONSE TO SILVICULTURAL TREATMENTS

Laura Ramirez, Cristian R. Montes, and Bronson P. Bullock

ABSTRACT

Slash pine (*Pinus elliottii*) is the second most important commercial species in the Southeastern United States and is usually established in poorly drained flatwoods where it outperforms other common pines. Modeling slash pine growth and how it responds to silvicultural treatments is of interest to forest managers wanting to maximize growth and profits. In this research, a growth and yield system of differential equations is proposed to model slash pine growth, including the effects of some silvicultural treatments (i.e., bedding and vegetation control). Data for this model came from a long-term study (30 years) established by the Plantation Management Research Cooperative (PMRC) of the University of Georgia across Georgia, Florida, and South Carolina. The model system describes the trajectory of three state variables: dominant height, mortality, and basal area. Treatment effects were incorporated into the dominant height and basal area models by using parameter modifiers and dummy variables associated with each of the treatments. Survival was not affected by the studied treatments, but the presence of fusiform rust (*Cronartium quercuum*) was found to be essential to determine the stand density trajectories for the evaluated stands. The three models were estimated simultaneously using maximum likelihood and the variance-covariance was modeled within the system.

INTRODUCTION

Differential equations have been used in forestry to construct growth and yield models and analyze how growth is both generated and constrained (Zeide 1993). Some of the most commonly used models, such as the Schumacher and the Chapman-Richards model (Pienaar and Turnbull 1972) when expressed in the differential form, rely on theories based on assumptions about how organisms grow (e.g., growth proportional to surface area) and how this growth is limited (e.g., maximum carrying capacity). Therefore, parameters in models constructed from differential equations have a biological interpretation which helps researchers to understand the underlying processes. Having interpretable parameters also facilitates the modification or construction of these models when the growth dynamic is changed by factors like silvicultural treatments or external pathogens.

Different strategies have been used in forestry to include silvicultural treatment effects into forest growth models. These usually consist of either including additional factors to the (integrated) model to account for the treatment effect (Pienaar and Rheney 1995), modifying the parameters of the model according to the treatment applied (Mason and Milne 1999), adding other covariables related to the treatment (Hynynen 1995), or using an age-shift approach in which the basic structure of the model is not modified (Snowdon 2002). Using models in the differential equation form facilitates the way treatment effects are incorporated into the growth model because the parameters that should be modified become more

Laura Ramirez, Graduate Student, Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA 30606

Cristian R. Montes, Associate Professor, Natural Resources, Biometrics and Timber Management, Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA 30606

Bronson P. Bullock, Professor, Forest Biometrics and Quantitative Forest Management, Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA 30606

intuitive. Thus, treatments that aim to increase growth rate (e.g., fertilization) can be modeled by modifying parameters that describe growth in the model, while parameters that describe the decline or limiting factors within the model can be modified if a treatment targets the reduction of limiting factors (e.g., vegetation control).

We used this strategy to model the response to bedding and vegetation control for slash pine (*Pinus elliottii*) plantations in the Southern United States. The technique of using parameter modifiers is a flexible methodology that allows for the inclusion of multiple treatments and their interactions (Ramirez and others 2022); therefore, this technique was tested to include the mentioned response. Survival was not affected by the treatments evaluated, but the presence of fusiform rust (*Cronartium quercuum*) was evaluated as a potential predictor to help describe mortality for the evaluated stands. Fusiform rust is the most common pathogen affecting slash pine in the Southern United States, and is associated with higher mortality rates and losses in product quality (Bailey and others 1989, Devine and Clutter 1985, Nance and others 1981, Susaeta 2020).

MATERIALS AND METHODS

Data

Data from the three variables of interest were available from a slash pine site preparation trial established in 1979 by the Plantation Management Research Cooperative (PMRC) of the University of Georgia. For this research, the treatments evaluated were bedding, competing vegetation control, and the combination of these two treatments. Measurements of diameter at breast height (d.b.h.), dominant height, stand density, and average fusiform rust infection rates per plot were available every 3 years from age 5 to age 31.

Growth responses were evaluated on three state variables (dominant height, survival, and basal area) and the growth and yield system constructed in this research incorporated the effect of bedding and vegetation control on dominant height and basal area growth. The three models composing the system were fitted simultaneously using maximum likelihood.

All models were fitted using the differential equation form and the initial value for solving the differential equation was taken as an additional parameter to be estimated on each one of the models (i.e., local parameters per plot). These models were fitted using the Julia programming language (Bezanson and others 2017). More details about the study are provided by Zhao and others (2007, 2009) and Ramirez and others (2022).

Dominant Height Model

A model form that had previously been used to model dominant height with the inclusion of bedding and vegetation control treatment effects was used (Ramirez and others 2023). This model is based on the Gompertz equation [equation (1)], and with the inclusion of the two silvicultural treatments of interest has the form of equation (2). For the construction of the growth and yield system, parameters of the model [equation (2)], were not taken directly from the Ramirez and others (2023) publication but were estimated following the methodology explained below.

$$\frac{dH}{dt} = \mu_0 H e^{-Kt} \quad (1)$$

$$\frac{dH}{dt} = \mu_0 b_1^{Z_1} H e^{-K b_2^{Z_2} t} \quad (2)$$

where

H is dominant height (m)

t is time in years

μ_0, K, b_1, and b_2 are parameters to be estimated

Z_1 and Z_2 are dummy variables equal to 1 if bedding or vegetation control was applied, respectively, and zero if otherwise.

Mortality Model

Mortality rates in slash pine plantations in the Southeastern United States have been demonstrated to be influenced by the presence of fusiform rust infection (Jones 1972, Sluder 1977, Wells and Dinus 1978). Recently, Ramirez and others (2023) developed a mortality model in which the rate of mortality depended on the proportion of fusiform rust infection at year 5. This model form [equation (3)] was used in this research as part of the growth and yield system:

$$\frac{dN}{dt} = (\alpha_0 + \alpha_1 * FR)\, N t^{\delta} \quad (3)$$

where

N is trees per hectare

t is time in years

α_0, α_1, and δ are parameters to be estimated

FR is the proportion of trees affected by fusiform rust on a given plot at age 5.

Similar to the dominant height component, parameters for this model were estimated as part of the system and not taken directly from Ramirez and others (2023). The effect of bedding and vegetation control on survival of slash pine plantations was found to be nonsignificant by Ramirez and others (2023), and therefore, this model was not further modified to include treatment effects.

Basal Area Model

Model from equation (4) was proposed to describe basal area trajectories. This model is based on the Korf growth model (Korf 1939, Zarnovican 1979) although modified to include dominant height and mortality as covariables, similar to the basal area model proposed by Pienaar and Shiver (1986) for slash pine plantations:

$$\frac{dB}{dt} = [\beta_1 + \beta_2 H + \beta_3 N]\frac{B}{t^{\beta_4}} \quad (4)$$

where

B is basal area (m^2 ha^{-1})

H , N, and t are as defined before

β_1..β_4 are parameters to be estimated.

An analysis of variance showed that there were significant differences in basal area for the treatments evaluated at every age. Therefore, the model from equation (4) was modified similarly to the dominant height model to include treatment effects as follows:

$$\frac{dB}{dt} = [\beta_1 + \beta_2 H + \beta_3 N]\frac{B}{t^{\beta_4}}\ \delta_1^{Z_1}\delta_2^{Z_2} \quad (5)$$

where Z_1 and Z_2 are the same dummy variables related to the treatments and defined for the dominant height model and δ_1 and δ_2 are additional parameters to be estimated.

In order to determine if the proposed modifications to the dominant height and basal area models were effective to include treatment effects, two growth and yield systems were proposed and compared, one with no treatment effects (system 1) and another with the treatment effects in both models (system 2). These systems are summarized in table 1. The system with better fit statistics was chosen to model the three variables of interest.

Parameter Estimation

Parameters for the two systems proposed were estimated using maximum likelihood in the Julia programing language (Bezanson and others 2017). In addition to the parameters for each one of the models from table 1 (i.e., global parameters), the initial values that define the trajectories (in any of the three variables) for each one of the plots, were defined as additional parameters to be estimated as part of the optimization procedure (i.e., local parameters per plot).

Each of the models from table 1 was first fitted individually assuming a normal distribution of the errors [i.e., H: $\varepsilon_i \sim N(0, \sigma_H)$, N: $\varepsilon_i \sim N(0,\sigma_N)$, B: $\varepsilon_i \sim N(0,\sigma_B)$]. Then, the system was simultaneously estimated by fixing the local parameters previously estimated per plot. That is, when

Table 1—Models for growth and yield systems 1 (without treatment effects) and 2 (includes treatment effects in dominant height and basal area)

Variable	System 1		System 2	
Dominant height	$\frac{dH}{dt} = \mu_0 H e^{-Kt}$	Eq. 1	$\frac{dH}{dt} = \mu_0 b_1^{Z_1} H e^{-K b_2^{Z_2} t}$	Eq. 2
Mortality	$\frac{dN}{dt} = (\alpha_0 + \alpha_1 * FR)\, N t^{\delta}$	Eq. 3	$\frac{dN}{dt} = (\alpha_0 + \alpha_1 * FR)\, N t^{\delta}$	Eq. 3
Basal area	$\frac{dB}{dt} = [\beta_1 + \beta_2 H + \beta_3 N]\frac{B}{t^{\beta_4}}$	Eq. 4	$\frac{dB}{dt} = [\beta_1 + \beta_2 H + \beta_3 N]\frac{B}{t^{\beta_4}}\delta_1^{Z_1}\delta_2^{Z_2}$	Eq. 5

H = dominant height (m); N = trees per ha; B = basal area ($m^2 ha^{-1}$); t = time in years; Z_1 and Z_2 = dummy variables equal to 1 if bedding or vegetation control was applied, respectively, and zero otherwise; FR = the proportion of trees affected by fusiform rust on a given plot at age 5; and μ_0, K, b_1, b_2, α_0, α_1, δ, β_1, β_2, β_3, β_4, δ_1 and $\delta 2$ = parameters to be estimated.

simultaneous estimation was done, only the global parameters were estimated, this was necessary to simplify the estimation procedure. For the simultaneous estimation, a multinormal distribution (*MN*) was assumed in which the errors are described by the following variance-covariance matrix:

$$\vec{\epsilon} \sim MN(\vec{0}, \Sigma),\ \Sigma = \begin{bmatrix} \sigma_H^2 & \sigma_{HxN} & \sigma_{HxB} \\ .. & \sigma_N^2 & \sigma_{NxB} \\ .. & .. & \sigma_B^2 \end{bmatrix} = \begin{bmatrix} \sigma_H^2 & \sigma_H\sigma_N\rho_{HxN} & \sigma_H\sigma_B\rho_{HxB} \\ .. & \sigma_N^2 & \sigma_N\sigma_B\rho_{NxB} \\ .. & .. & \sigma_B^2 \end{bmatrix}$$

where

$\vec{\epsilon}$ is the vector of errors (observed–predicted)

Σ is the variance-covariance matrix (symmetrical)

σ_x^2 is the variance of the X variable (H, N, or B)

σ_{X_iY} is the covariance between variables X and Y

ρ_{X_iY} is the correlation between variables X and Y

The following statistics were calculated to evaluate model fit:

Root mean square error (RMSE):

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(Y_i - \hat{Y}_i)^2}{n-p}} \quad (6)$$

Mean difference (MD) or bias:

$$MD = \frac{\sum_{i=1}^{n}(Y_i - \hat{Y}_i)}{n} \quad (7)$$

Mean absolute difference (MAD):

$$MAD = \frac{\sum_{i=1}^{n}|Y_i - \hat{Y}_i|}{n} \quad (8)$$

and AIC was used for model comparison:

$$AIC = -2\,loglik + 2p \quad (9)$$

where

n is the total number of observations

p is the number of parameters in each model

Y_i is the observed value

$\hat{Y}_i$ is the estimated value.

RESULTS

Results comparing the two growth and yield systems tested are presented in table 2. The AIC value for system 1 (no treatments) was 1,536.86, while for system 2 it was 778.36, indicating that the system with treatment factors performed better than the system without treatment effects (lower AIC values indicate better fit). From table 2 it can be seen how there was significant gain in precision (reduction of error and bias) when using treatment modifiers. After adding these modifiers, the average error (RMSE) for the dominant height model was reduced from 0.71 to 0.57 m (20 percent reduction), while the error for the basal area model was reduced from 1.88 to 1.25 $m^2\ ha^{-1}$ (34 percent reduction). Bias (MD) was reduced 67 percent for dominant height and 78 percent for basal area. The RMSE for mortality did not differ between the systems [46.04 trees per hectare (TPH)] and the average bias was 0.83 TPH. Ramirez and others (2023) showed that mortality was not affected by bedding and vegetation control, but that the inclusion of fusiform rust was crucial to describe the survival trajectories. An additional comparison was then made for the mortality component by fitting the same model but excluding the fusiform rust information [i.e., model from equation (3) with $\alpha_1 = 0$]. When fusiform rust was not included into the model, the average error (RMSE) increased from 46.04 to 77.53 TPH and the bias changed from 0.83 to -0.43 TPH. Given these results, system 2 was selected as the best growth and yield system and therefore, the results presented in this section refer to this system.

Parameter estimates and fit statistics for each of the components of system 2 are presented in tables 3 and 4, respectively. The estimated values in table 3 correspond to the values obtained with the simultaneous estimation of the global parameters using the multivariate normal in which the correlation between the variables was estimated and the local values per plot

Table 2—Fit statistics for dominant height and basal area models with the two systems proposed

Variable	Growth and yield system	RMSE	MD	MAD
Dominant height (m)	1	0.71	0.03	0.55
	2	0.57	0.01	0.45
Basal area (m^2ha^{-1})	1	1.88	0.18	1.48
	2	1.25	0.04	0.94

System 1 is without treatment effects, system 2 is with treatment effects in dominant height and basal area

RMSE = root mean square error; MD = mean difference; MAD = mean absolute difference.

were taken from the independent estimation. To test for the effect of modeling the correlation between the variables, the same model was fitted assuming zero correlation between the errors of the variables. With this model, which simulated independent estimation of each variable in the model, parameter estimates were not different for the dominant height model but differed for the mortality and basal area models. When comparing the estimated values of the standard error of the parameters, no differences were found for the dominant height model and opposite trends were found for the other two variables, with an increase in the standard error for the mortality parameters, but a decrease in the standard error for the basal area parameters when including correlation. Despite these differences, the fit statistics were not significantly different when comparing the systems with and without correlation (table 4).

Table 3—Parameter estimates for the best growth and yield system for slash pine

Variable	Parameter	Estimated value	Standard error
Dominant height (m)	μ	0.3737	0.0042
	K	0.1128	0.0010
	b_1	0.9178	0.0020
	b^2	1.1057	0.0019
Mortality (trees per ha)	α_0	-0.0151	0.0026
	α_1	-0.3596	0.0571
	δ	-0.6292	0.0674
Basal area (m^2ha^{-1})	β_1	10.4090	0.3921
	β_2	-0.2776	0.0097
	β_3	-1.4977	0.0649
	β_4	1.5512	0.0183
	δ_1	0.8747	0.0024
	δ_2	0.7449	0.0021
Variance-covariance matrix (Σ)	$\ln(\sigma_H)$	-0.5556	0.0290
	$\ln(\sigma_N)$	-3.0782	0.0293
	$\ln(\sigma_B)$	0.2253	0.0294
	$\rho_{H\chi N}$	-0.0054	0.0418
	$\rho_{H\chi B}$	0.4408	0.0340
	$\rho_{N\chi B}$	0.4172	0.0354

H = dominant height (m); N = trees per ha; B = basal area (m^2ha^{-1}); t = time in years; Z_1 and Z_2 = dummy variables equal to 1 if bedding or vegetation control was applied, respectively, and zero otherwise; FR = the proportion of trees affected by fusiform rust on a given plot at age 5; and $\mu_o, K, b_1, b_2, \alpha_0, \alpha_1, \delta, \beta_1, \beta_2, \beta_3, \beta_4, \delta_1, \delta_2, \sigma_H, \sigma_N, \sigma_B, \rho_{H\chi N}, \rho_{H\chi B}$, and $\rho_{N\chi B}$ = the parameters estimated.

Table 4—Fit statistics for the three variables of the best growth and yield system for slash pine

Variable	Fit statistic	Value with correlation	Value without correlation
Dominant height (m)	RMSE	0.57	0.57
	MD	0.01	0.00
	MAD	0.45	0.45
Mortality (trees per ha)	RMSE	46.04	45.67
	MD	0.83	-0.14
	MAD	32.85	32.64
Basal area (m^2ha^{-1})	RMSE	1.25	1.24
	MD	0.04	-0.01
	MAD	0.94	0.94

RMSE = root mean square error; MD = mean difference; MAD = mean absolute difference.

Although no improvement was obtained in terms of model fit when accounting for correlation, accounting for this value during the estimation procedure is statistically sound and therefore, the estimated parameter values from table 3 were used to plot the predicted trajectories and calculated residuals for the three variables of interest in figure 1. Overall, a good fit was obtained, and the residuals did not show any trend. The small errors and bias observed from the proposed growth and yield system might be the result of using local parameters per plot, which are responsible for the individual curves observed in figure 1. Therefore, the robustness of the proposed system was evaluated by testing how accurate the predictions were when local parameters were not used. This consisted of using only the estimated global parameters from table 3 and utilizing the observed values as starting points for solving the differential equations to predict the trajectories forward. This simulates how the system can be used with a new dataset. Different starting points were tested. For example, the first observation (at age 5) was first used as the starting point to predict the whole series until year 31, then, the second observation (at age 8) was used to predict

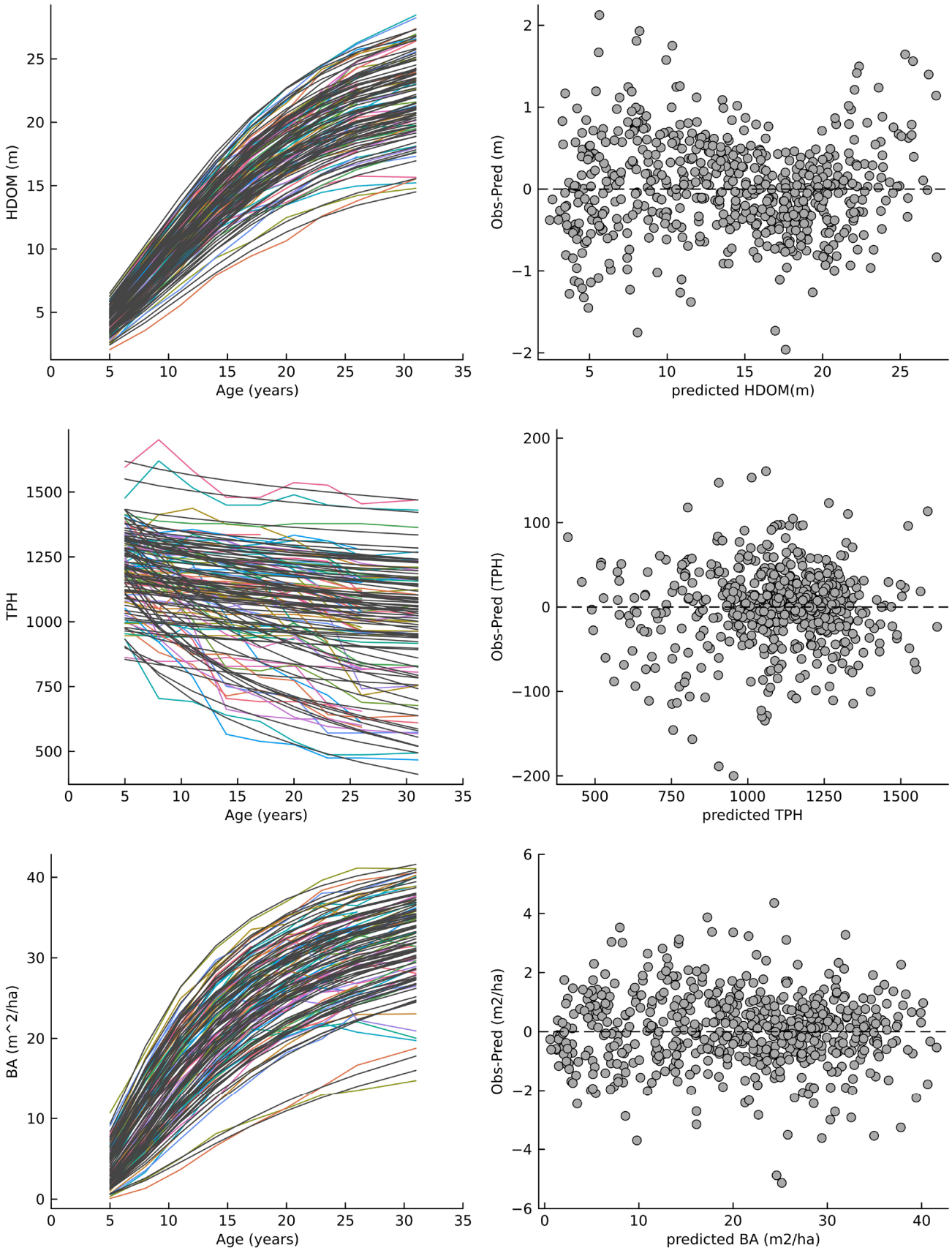

Figure 1—Dominant height (HDOM), mortality (TPH), and basal area (BA) predicted curves and residuals using the best growth and yield system for slash pine.

the last observation. An additional prediction was made when using only the closest point to predict the following observation (e.g., observation at age 5 used to predict observation at age 8, observation at age 8 used to predict observation at age 11, and so on). For these different scenarios, the average error (RMSE) and bias were calculated and the results were summarized in figure 2.

In forestry applications, accuracy and precision decrease when the projection interval increases (Wang and others 2020). This fact was particularly important for the growth and yield system described here given that the differential equations proposed are highly influenced by the starting point of the state variables. This was checked and it is shown in figure 2. In this graph, the average length of the projection interval was plotted against the RMSE and the average bias (MD). Higher errors were obtained when the first observation (at year 5) was used to predict all the points forward (until year 31). This error was reduced when the projection interval was reduced. In figure 2, the red diamond indicates the error obtained when the projection was made using the closest previous observation to predict the current observation; this meant a projection length of 3 years for most of the observations. Although not all possible combinations of projection lengths were used for evaluating the model, it is clear that this system has better performance with short predictions. The RMSE was consistently reduced when the projection length was reduced. No clear trend was observed for bias. Overall, when evaluating the magnitude of the errors, basal area was the most sensitive variable, while mortality was the least affected variable. Results from this analysis are relevant to model users who would want to apply this model for a given projection length.

From the growth and yield system presented, the dominant height and mortality model can be integrated analytically. These two models are presented in equation (10) and equation (11). Nevertheless, the basal area model cannot be integrated analytically and therefore, the differential equation form needs to be used when

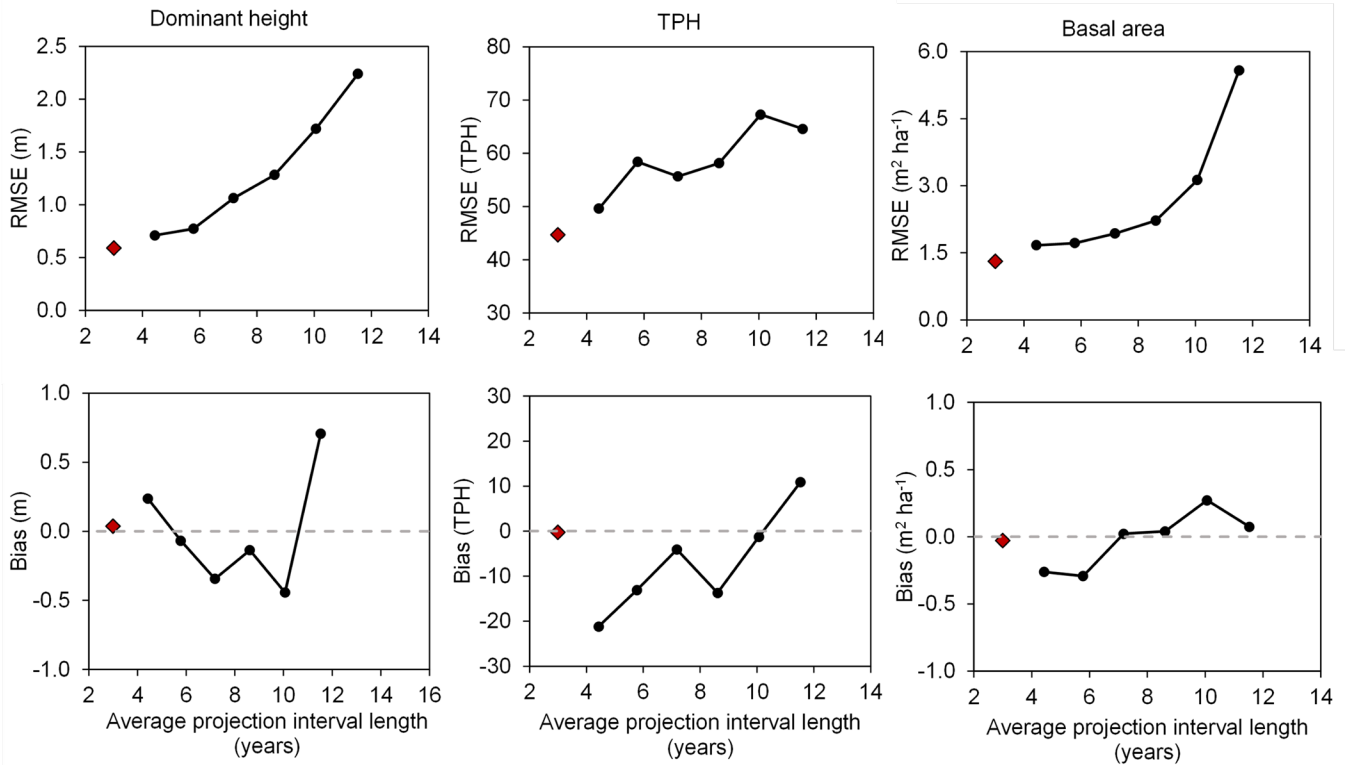

Figure 2—Root mean square error (RMSE) and bias for dominant height (m), mortality (TPH), and basal area (m^2ha^{-1}) when projecting starting from a different age. The red diamond indicates prediction using the immediate previous point to project.

using the model. Therefore, a routine in R (R Core Team 2018) to use the proposed growth and yield system was included as an appendix to this manuscript.

$$H_2 = H_1\, exp\left[\frac{\mu_0 b_1^{Z_1}}{K b_2^{Z_2}}\left(e^{-K b_2^{Z_2} t_1} - e^{-K b_2^{Z_2} t_2}\right)\right] \quad (10)$$

$$N_2 = N_1\, exp\left[\frac{\alpha_0 + \alpha_1 FR}{\delta + 1}\left(t_2^{\delta+1} - t_1^{\delta+1}\right)\right] \quad (11)$$

DISCUSSION

In this research, the effect of bedding and vegetation control was successfully incorporated into a growth and yield system for slash pine using parameter modifiers. This technique has been commonly used in forestry models (e.g., Hynynen and others 1998) and has the advantage of incorporating treatment interactions and using both control and treatment plots during the fitting procedure, which avoids assigning all of the variability of the data to the treatment effect, which is the case when treatment effects are incorporated as an additive function to a base model (Gyawali and Burkhart 2015, Logan and Shiver 2006, Mason and Milne 1999, Pienaar and Rheney 1995). Both average error and bias were reduced when treatment modifiers were added to the base growth and yield system, accounting for the treatment effects. These treatments, nevertheless, did not affect mortality as shown in other research by Ramirez and others (2023), and therefore, the system proposed did not include parameter modifiers for this component of the system, although it did include fusiform rust infection rates which showed to improve model fit, reducing both error and bias significantly.

When constructing growth and yield systems for forest stands, it is common to find variables that appear in both the left and right side of the equations, generating an interdependent system of equations that requires simultaneous estimation to get unbiased estimates of the parameters of the system (Borders and Bailey 1986, Goelz and Burk 1996). In addition, the correlation between the errors of the variables should be addressed. Although unbiased estimates of the parameters can be obtained when correlation is present, the standard error of the parameters is not accurate when correlations exist and are not taken into account. The parameters of the system proposed were estimated simultaneously using maximum likelihood and the variance-covariance matrix was estimated for the system, addressing these issues.

Several authors have modelled before the response to silvicultural treatments in growth and yield systems, although without making use of simultaneous estimation (Bailey and others 1989, McTague 2008, Pienaar and Rheney 1995). When including simultaneous estimation, most of the authors have focused on the baseline system, in which silvicultural treatments are not considered (Borders and Bailey 1986, Gallagher and others 2019, Murphy and Sternitzke 1979, Pienaar and Harrison 1989, Sullivan and Clutter 1972). Fewer authors have included both response to silvicultural treatments and simultaneous estimation (Fang and others 2001, Martin and others 1999). To the knowledge of the authors this is the first growth and yield system for slash pine plantations where silvicultural treatments were included using a system of differential equations and simultaneous estimation was used. Nevertheless, accounting for the correlation between the errors of the different components of the system did not improve model fit and although the estimated value for the standard error changed for the mortality and basal area models, this change was not unidirectional.

The use of differential equations generally helps to construct the models when they need to be modified to incorporate treatment effects or the influence of pathogens like fusiform rust infection. This is because it is relatively easy to determine what parameters should be modified according to the treatment being considered. An additional factor that must be considered when using differential equations is how the equation behaves upon different starting values. For the models proposed, a big influence of these values was found because the trajectories of the variable being modeled will highly differ with small variations on the initial value. For these models, the initial starting point defines the asymptote for each one of the curves. For this reason, when using the observed values instead of the predicted values as initial points to solve the differential equation, the errors increased considerably.

Nevertheless, when points closer to the asymptote (later ages) and shorter projection intervals were used, the errors decreased.

CONCLUSIONS

A growth and yield system including dominant height, mortality, and basal area was fitted for slash pine plantations. The effect of bedding and vegetation control was successfully incorporated into dominant height and basal area components by using dummy variables. The average error and bias were reduced when these modifiers were added, compared to base models without these factors. Fusiform rust infection rates were crucial to describe mortality rates. These rates were modeled as a function of the initial infection rate per plot. Dominant height and mortality were used as predictor variables for the basal area model proposed, adding correlation within the system which was addressed by estimating the variance-covariance matrix of the system. The growth and yield system proposed was shown to have lower errors for short projection intervals, and therefore its use is recommended for these intervals.

ACKNOWLEDGMENTS

This research was conducted with support from the Plantation Management Research Cooperative (PMRC), where resources have been invested in establishing and collecting data from this long-term research trial. Special thanks to the PMRC Field Crew who collected the data for this research over the course of 3 decades. In addition, this work was supported by the Warnell School of Forestry and Natural Resources at the University of Georgia, the J.L. Clutter Forest Biometrics Scholarship, and the McIntire-Stennis Project GEOZ020205-MS.

LITERATURE CITED

Bailey, R.L.; Burgan, T.M.; Jokela, E.J. 1989. Fertilized midrotation-aged slash pine plantations—stand structure and yield prediction models. Southern Journal of Applied Forestry. 13(2): 76–80. https://doi.org/10.1093/sjaf/13.2.76.

Bezanson, J.; Edelman, A.; Karpinski, S.; Shah, V.B. 2017. Julia : A fresh approach to numerical computing. SIAM Review. 59: 65–98. https://doi.org/10.1137/141000671.

Borders, B.E.; Bailey, R.L. 1986. A compatible system of growth and yield equations for slash pine fitted with restricted three-stage least squares. Forest Science. 32(1): 185–201.

Devine, O.J.; Clutter, J.L. 1985. Prediction of survival in slash pine plantations infected with fusiform rust. Forest Science. 31(1): 88–94.

Fang, Z.; Bailey, R.L.; Shiver, B.D. 2001. A multivariate simultaneous prediction system for stand growth and yield with fixed and random effects. Forest Science. 47(4): 550–562.

Gallagher, D.A.; Bullock, B.P.; Montes, C.R.; Kane, M.B. 2019. Whole stand volume and green weight equations for loblolly pine in the Western Gulf Region of the United States through age 15. Forest Science. 65(4): 420–428. https://doi.org/10.1093/forsci/fxy068.

Goelz, J.C.G.; Burk, T.E. 1996. Measurement error causes bias in site index equations. Canadian Journal of Forest Research. 26(9): 1585–1593. https://doi.org/10.1139/x26-178.

Gyawali, N.; Burkhart, H.E. 2015. General response functions to silvicultural treatments in loblolly pine plantations. Canadian Journal of Forest Research. 45(3): 252–265. https://doi.org/10.1139/cjfr-2014-0172.

Hynynen, J. 1995. Predicting the growth response to thinning for Scots pine stands using individual-tree growth models. Silva Fennica. 29(3): 225–246. https://doi.org/10.14214/sf.a9210.

Hynynen, J.; Burkhart, H.E.; Allen, H.L. 1998. Modeling tree growth in fertilized midrotation loblolly pine plantations. Forest Ecology and Management. 107: 213–229. https://doi.org/10.1016/S0378-1127(97)00334-4.

Jones, E.P., Jr. 1972. Fusiform rust affects planted slash pine. Journal of Forestry. 70(6): 350–352. https://doi.org/10.1093/jof/70.6.350.

Korf, V. 1939. A mathematical definition of stand volume growth law. Lesnická práce. 18: 337–339.

Logan, S.R.; Shiver, B.D. 2006. Adjusting slash pine growth and yield for silvicultural treatments. 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: 328–332.

Martin, S.W.; Bailey, R.L.; Jokela, E.J. 1999. Growth and yield predictions for Lower Coastal Plain slash pine plantations fertilized at mid-rotation. Southern Journal of Applied Forestry. 23(1): 39–45. https://doi.org/10.1093/sjaf/23.1.39.

Mason, E.G.; Milne, P.G. 1999. Effects of weed control, fertilization, and soil cultivation on the growth of *Pinus radiata* at midrotation in Canterbury, New Zealand. Canadian Journal of Forest Research. 29(7): 985–992. https://doi.org/10.1139/x99-087.

McTague, J.P. 2008. Modelling the response of loblolly pine to juvenile fertilization. Open Forest Science Journal. 1: 80–88. https://doi.org/10.2174/1874398600801010080.

Murphy, P.A.; Sternitzke, H.S. 1979. Growth and yield estimation for loblolly pine in the West Gulf. Res. Pap. SO-154. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 8 p.

Nance, W.L.; Froelich, R.C.; Shoulders, E. 1981. Effects of fusiform rust on survival and structure of Mississippi and Louisiana slash pine plantations. Res. Pap. SO-172. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 11 p.

Pienaar, L.V.; Harrison, W.M. 1989. Simultaneous growth and yield prediction equations for *Pinus elliottii* plantations in Zululand. South African Forestry Journal. 149: 48–53. https://doi.org/10.1080/00382167.1989.9628992

Pienaar, L.V.; Rheney, J.W. 1995. Modeling stand level growth and yield response to silvicultural treatments. Forest Science. 41(3): 629–638.

Pienaar, L.V.; Shiver, B.D. 1986. Basal area prediction and projection equations for pine plantations. Forest Science. 32(3): 626–633.

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.

R Core Team. 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing.

Ramirez, L.; Montes, C.R.; Bullock, B.P. 2022. Long-term term effect of bedding and vegetation control on dominant height of slash pine plantations in the southeastern United States. Forest Ecology and Management. 522: 120479. 11 p. https://doi.org/10.1016/j.foreco.2022.120479.

Ramirez, L.; Montes, C.R.; Bullock, B.P. 2023. Modeling slash pine mortality rates incorporating responses to silvicultural treatments and fusiform rust infection rates. Forest Ecology and Management. 532: 120832. 9 p. https://doi.org/10.1016/j.foreco.2023.120832.

Sluder, E.R. 1977. Fusiform rust in loblolly and slash pine plantations on high-hazard sites in Georgia. Res. Pap. SE-160. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 10 p.

Snowdon, P. 2002. Modeling Type 1 and Type 2 growth responses in plantations after application of fertilizer or other silvicultural treatments. Forest Ecology and Management. 163: 229–244. https://doi.org/10.1016/S0378-1127(01)00582-5.

Sullivan, A.D.; Clutter, J.L. 1972. A simultaneous growth and yield model for loblolly pine. Forest Science. 18: 76–86.

Susaeta, A. 2020. Implications of future risk of fusiform rust on optimal forest management of even-aged slash pine plantations. Forest Policy and Economics. 116: 102183. 6 p. https://doi.org/10.1016/j.forpol.2020.102183.

Wang, M.; Montes, C.R.; Bullock, B.P.; Zhao, D. 2020. An empirical examination of dominant height projection accuracy using difference equation models. Forest Science. 66(3): 267–274. https://doi.org/10.1093/forsci/fxz079.

Wells, O.O.; Dinus, R.J. 1978. Early infection as a predictor of mortality associated with fusiform rust of southern pines. Journal of Forestry. 76: 8–12. https://doi.org/10.1093/jof/76.1.8.

Zarnovican, R.1979. Fonction de la croissance de Korf. Forestry Chronicle. 55(5): 194–197. https://doi.org/10.5558/tfc55194-5.

Zeide, B. 1993. Analysis of growth equations. Forest Science. 39(3): 594–616.

Zhao, D.; Kane, M.; Borders, B.; Harrison, W.M. 2007. Slash pine site preparation study: results through age 26. Tech. Rep. 2007-3. Athens, GA: Plantation Management Research Cooperative, University of Georgia. 53 p.

Zhao, D.; Kane, M.; Borders, B.; Harrison, M. 2009. Long-term effects of site preparation treatments, complete competition control, and repeated fertilization on growth of slash pine plantations in the flatwoods of the southeastern United States. Forest Science. 55(5): 403–410.

APPENDIX: R CODE FOR GROWTH AND YIELD SYSTEM FOR A SINGLE PREDICTION

```
# Loading the packages
library(deSolve)

#---------Estimated parameters----------------
#DH parms
u0 <- 0.3737; K <- 0.1128; b1 <- 0.9178; b2 <- 1.1057
#N parms
a0 <- -0.0151; a1 <- -0.3596; d <- -0.6292;
#BA parms
B1 <- 10.4090; B2 <- -0.2776; B3 <- -1.4977 ; B4 <- 1.5512; d1 <- 0.8747;

d2 <- 0.7449

parms_sys <- c(u0, K, b1, b2, a0, a1, d, B1, B2, B3, B4, d1, d2)

#--------Initial conditions------------------
hini  <- 3
Nini  <- 1100/1000 #Needs to be divided by 1000
BAini <- 1.5
t1    <- 5
t2    <- 8
FR    <- 0.1
Z1    <- 0.0
Z2    <- 0.0

#--------Define system (differential equations)------
System <- function(t, u, p) {
  h0  <- u[1]
  N0  <- u[2]
  BA0 <- u[3]

  dH <- (h0 * p[1] * p[3]^Z1) * exp(-p[2] * p[4]^Z2 * t)
  dN <- N0 * (p[5] + p[6] * FR) * t^p[7]
  dB <- (p[8] + p[9] * h0 + p[10] * N0)*BA0 / t^p[11]) * p[12]^Z1 * p[13]^Z2

  return(list(c(dH, dN, dB)))
}

#-----Solve the system numerically--------------------
y0    <- c(hini, Nini, BAini)
times <- c(t1,t2)
sol   <- ode(y = y0, times = times, func = System, parms = parms_sys)

#---- extract the values of interest----------------
HDOM_pred <- sol[2,2]
N_pred    <- sol[2,3] * 1000 #Needs to back transform the variable
BA_pred   <- sol[2,4]
```

DIAMETER ESTIMATION BASED ON LIDAR POINT CLOUDS AT STAND LEVEL OF LOBLOLLY PINE PLANTATIONS

Clinton Talmage, Yuhui Weng, Yanli Zhang, and Jason Grogan

ABSTRACT

Variables such as average tree diameter at breast height (d.b.h.) and total height are important stand characteristics for forest management. Conventionally, tree d.b.h. and height are measured; however, due to the difficulty of measuring tree height in the field, this variable is often estimated from tree d.b.h. Recent developments in remote sensing techniques, in particular the application of light detection and ranging (LiDAR) technology to forestry, provides a promising alternative. LiDAR point clouds can be used to accurately estimate tree total height and crown area, which can be used to calculate the number of trees per unit area. This study used LiDAR data-derived stand variables including total height, crown size, and the number of trees per acre collected from permanent growth and yield plots established in loblolly pine (*Pinus taeda*) plantations in east Texas to develop alternative models to estimate stand average d.b.h. The models were evaluated based on the mean difference between observed and predicted d.b.h., mean absolute difference, and root mean square error, respectively. Results indicate that stand-level d.b.h. could be accurately estimated using derived stand variables. The estimated stand d.b.h., paired with LiDAR-derived height and number of trees per acre, can then be employed to estimate stand-level volumes.

INTRODUCTION

The Southeastern United States produces more than 50 percent of the country's timber and as much as 25 percent of total world timber supply (Prestemon and Abt 2002). Covering more than 39.5 million acres in the Southeastern United States as of 2010 (Hartsell and Connor 2013), the commonly planted, rapidly growing, and disease resistant native loblolly pine (*Pinus taeda*) is the most important commercial forest species in the United States. Due to the significance of loblolly pine and timber production in the South, effective management of this resource is particularly important. Effective management requires knowledge of current stand volume and other variables, such as average diameter at breast height (d.b.h.), height, and basal area (Avery 1975).

Traditionally, stand volume estimations have relied on direct field-based measurements of average tree dimensions and stocking estimated from sample plots. However, measurements of some variables, such as tree height, can often be difficult, time consuming, or inefficient to obtain, especially in stands with complex canopy structure (Larjavaara and Muller-Landau 2013). To address this, height-diameter models often are utilized in conventional forest inventories to make inferences about stand height.

Recent advancements in remote sensing technology, such as light detection and ranging (LiDAR) have been gaining popularity due to their ability to quickly collect measurements of entire forest stands. For example, measurements of tree location, height, and crown size are more

Clinton Talmage, Graduate Research Assistant, Arthur Temple College of Forestry and Agriculture, Stephen F. Austin State University, Nacogdoches, TX 75962

Yuhui Weng, Associate Professor of Forestry, Arthur Temple College of Forestry and Agriculture, Stephen F. Austin State University, Nacogdoches, TX 75962

Yanli Zhang, Marilyn Cranford Distinguished Professor of Geospatial Sciences, Arthur Temple College of Forestry and Agriculture, Stephen F. Austin State University, Nacogdoches, TX 75962

Jason Grogan, Research Specialist, Arthur Temple College of Forestry and Agriculture, Stephen F. Austin State University, Nacogdoches, TX 75962

effective to gather from aerial LiDAR than from manual measurements (Aubry-Kientz and others 2019). Popescu and others (2003) were successful in estimating individual tree crown diameter derived from LiDAR point clouds. Koch and others (2006) found delineating individual crowns in conifer stands to be more consistent than mixed hardwoods.

While aerial LiDAR is particularly good at measuring tree height (Hyyppä and others 2000), direct measurement of tree d.b.h. using LiDAR is a challenge. However, LiDAR-derived metrics have been used to estimate individual tree or stand-level d.b.h., as strong evidence of allometric relationships between tree height and tree diameter exists for some species (Zhang and others 2020). For instance, models relating d.b.h. to height are well established (Curtis 1967, Hanus and others 1999, Temesgen and others 2006) and previous studies have shown that LiDAR-derived metrics could be used to estimate tree d.b.h. from height (Cao and Dean 2013, Hyyppä and others 2001, Parker and Evans 2004, Salas and others 2010). The objective of this paper was to determine if it was possible to develop a height-to-d.b.h. equation for intensively managed loblolly pine plantations throughout east Texas and western Louisiana using both field-collected data from permanent plots established by the East Texas Pine Plantation Research Project and corresponding LiDAR data.

MATERIALS AND METHODS

Field Data

The East Texas Pine Plantation Research Project (ETPPRP) initiated its Phase II growth and yield study in 2004 by installing 136 permanent plots in intensively managed loblolly pine plantations throughout east Texas and western Louisiana (fig. 1). All plots were treated similarly to the plantations they were within, i.e., if a plantation was thinned or fertilized, then the plot within also was thinned or fertilized. The dimensions for each plot were 100 feet by 100 feet. Within the plot, all trees were tagged and measurements of tree d.b.h., height, height to live crown, and defects were made every 3 years. Individual tree d.b.h. was measured using diameter calipers and heights were measured using a clinometer or Vertex hypsometer. The individual tree data were summarized to plot level. While the plots were repeatedly measured, only data which corresponded temporally with LiDAR data were used. Table 1 represents summary statistics of plot-level mean tree height, d.b.h., and age.

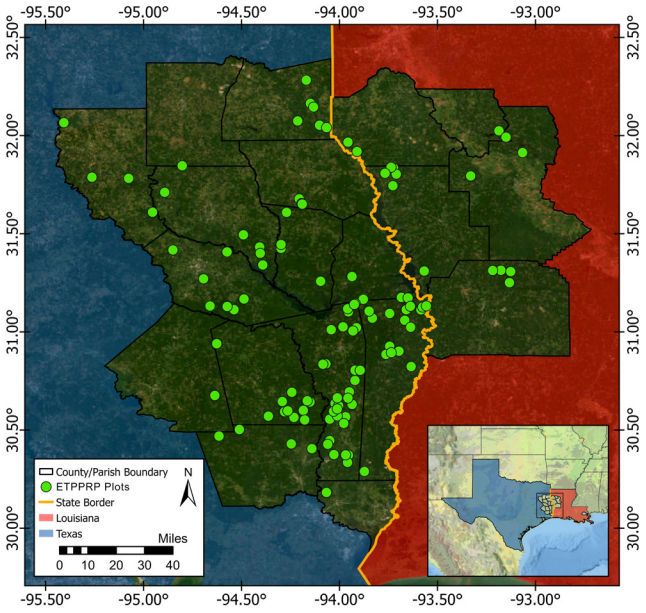

Figure 1—East Texas Pine Plantation Research Project (ETPPRP) plot locations in east Texas and western Louisiana.

Table 1—Summary statistics of field measured and LiDAR-derived stand attributes

Source	Attribute	Mean	Minimum	Maximum	Standard deviation
Field	Height (feet)	52.0	30.0	88.0	10.0
	d.b.h. (inches)	7.9	5.0	14.0	1.8
	Age (years)	16.0	7.0	23.0	3.4
LiDAR	Height (feet)	49.0	26.0	86.0	9.9
	Crown area (square feet)	18.0	1.5	48.0	7.5
	Crown diameter (feet)	4.6	1.2	7.6	1.0

Aerial LiDAR Data

The LiDAR-based 3D Elevation Program (3DEP) is managed by the U.S. Geological Survey to produce high-quality topographic data to update the National Elevation Dataset. LiDAR is an efficient method to collect elevation data within a desired location. When used in conjunction with global navigation satellite systems, LiDAR can produce highly accurate three-dimensional point measurements, also known as point clouds (White and others 2016). Using these point clouds, it is possible to produce accurate representations of a forest canopy. Canopy height models were derived using the difference between tree canopy returns and corresponding ground returns. Digital terrain models of ground returns and digital surface sampled models of first returns were derived from point clouds. LiDAR data was collected during the month of March between 2016 and 2018. ETPPRP field data was collected during the summer between May and August of corresponding years. In most cases, data from LiDAR and field-measured plots were matched year to year. However, in some instances an exact year-to-year match could not be made. In this situation, growth and yield modeling (Coble and others 2017) was utilized to prorate field data to the nearest appropriate year.

The grid size for each elevation model was 2.3 feet or the horizontal accuracy of the LiDAR dataset. This process allows for point clouds to be preprocessed into a product for further analysis. The resulting raster then was further processed by the ForestTools package developed for R (Plowright 2022). This package employs a variable window filter and watershed algorithm to delineate tree canopies, resulting in metrics of individual tree count, height, and canopy size (Plowright 2022). LiDAR parameters for this study include a nominal point density of 0.23 points per square foot, nominal point spacing of 2 feet, a scanning angle of 15 degrees, and vertical accuracy of 0.32 feet. Point cloud samples were collected from the same coordinates as each ETPPRP plot. With plots located throughout a large geographic area (eastern Texas and western Louisiana), it was not feasible to match the thousands of measured trees with their LiDAR data, especially given the low point density of the 3DEP LiDAR data. Instead, a plot-level average was used for both LiDAR-derived and field-measured variables.

Statistical Models

Both linear and nonlinear regression were used to predict plot d.b.h. using LiDAR-derived tree traits. Multiple height-to-d.b.h models were explored in preliminary analyses but only the top five models were selected to use in this study (table 2). SAS software was used in model development (Plowright 2022). Model assumptions (equality of variance, independence, and normality) were checked using residual plots.

Table 2—Candidate equations for estimating d.b.h. from height

Model number	Models	Equations	References
1	Parker and Evans	d.b.h. = a + bH^c * TPA^d	Parker and Evans 2004
2	Cao and Dean	d.b.h. = a + bH^c	Cao and Dean 2013
3	Hyyppä and others	d.b.h. = a + bH + cCD	Hyyppä and others 2001
4	Curtis	d.b.h. = H^2/(a + bH + cH^2)	Curtis 1967
5	Talmage	d.b.h. = a + bH + cTPA * dCA	–

H = tree height; *TPA* = trees per acre; *CD* = crown diameter; *CA* = crown area; a, b, c, d = parameter coefficients.

Model Accuracy Assessment

The data (field data and LiDAR data) were randomly split into two groups with 80 percent used for model training and 20 percent for model testing. Model performance was assessed using the test dataset by calculating coefficients of determination (R^2), root mean square error (RMSE) and relative root mean square error (rRMSE), and mean absolute difference (MAD) as follows:

$$R^2 = 1 - \left(\frac{SSR}{SST}\right) \quad (1)$$

$$\mathrm{RMSE} = \sqrt{\sum_{i=1}^{N}(y_i - \hat{y}_i)^2} \quad (2)$$

$$\mathrm{rRMSE} = \frac{\sqrt{\sum_{i=1}^{N}(y_i - \hat{y}_i)^2}}{(\sum_{i=1}^{N} y_i)/N} \times 100 \quad (3)$$

$$\mathrm{MAD} = \frac{\sum_{i=1}^{N}|\hat{y}_i - y_i|}{N} \quad (4)$$

where

SSR = sum of square residuals

SST = sum of square total

y_i is the field-measured variable

$\hat{y}_i$ is the LiDAR-estimated variable

RESULTS

The correlation between field-measured and LiDAR-derived height was 0.895 before prorating height using the ETPPRP model to account for difference in year. After adjusting field data, the correlation between field-measured and LiDAR-derived height increased to 0.95. Table 3 provides the metrics for the top five assessed models based on testing datasets. Model 1 (Parker and Evans 2004) had an R^2 of 0.67 and a RMSE of 0.80, which was approximately 10.2 percent of the mean (table 3, fig. 2). The other four equations all resulted in an R^2 of 0.28 or less (table 3, fig. 2). Even though model 3 had an R^2 of 0.28, its rRMSE of 15.5 percent was considered acceptable.

Our resulting model, based on model 1 (Parker and Evans 2004), was as follows:

$$\text{d.b.h.} = 3.54 + 0.0603(H^{1.36483648})(TPA^{-0.2087}) \quad (5)$$

where

H = tree height (feet)

TPA = trees per acre

Table 3—The five models with metrics of analyses that were greater than others, processed using the validation dataset

Model number	Author(s)	Coefficient of determination (R^2)	Root mean square error (RMSE)	Relative RMSE	Mean absolute difference
			inches	*percent*	*inches*
1	Parker and Evans	0.67	0.80	10.24	0.75
2	Cao and Dean	0.23	1.20	15.07	1.65
3	Hyyppä and others	0.28	1.30	15.47	1.03
4	Curtis	0.24	1.23	14.96	1.26
5	Talmage	0.26	1.33	16.30	1.40

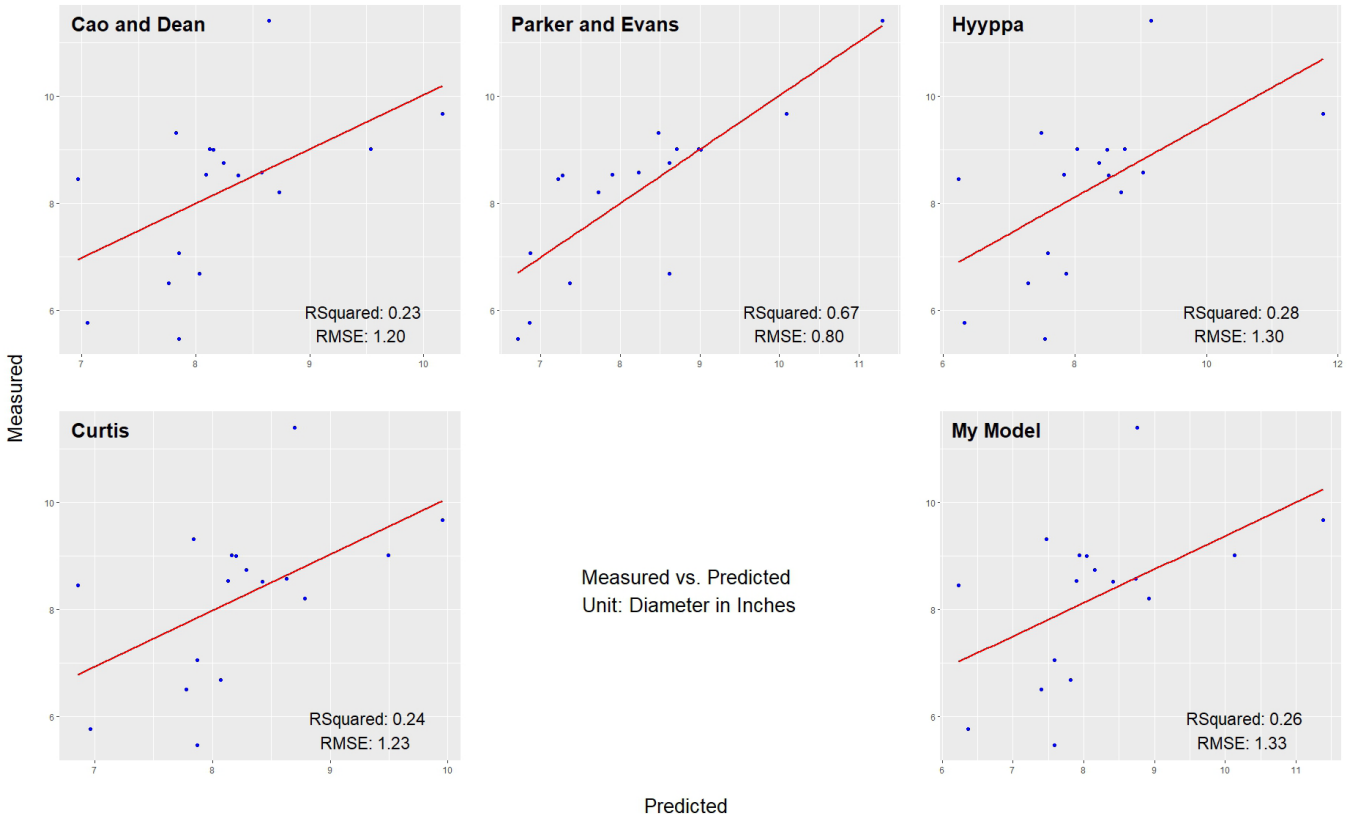

Figure 2—Field-measured versus LiDAR-predicted diameter at breast height of East Texas Pine Plantation Research Project plots.

The standard errors of the parameter coefficients were 1.12 (a = 3.54), 0.11 (b = 0.06), 0.38 (c = 1.36), and 0.13 (d = -0.21). Figure 3 shows the residual plots for the model.

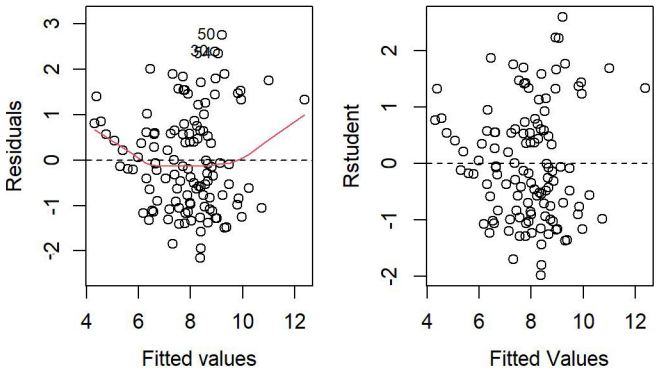

Figure 3—Residual plots of test data for Model 1.

DISCUSSION

We sought to determine if a conventional model could adequately estimate d.b.h. from LiDAR-derived variables. Toward this end, an R^2 of 0.67 and a rRMSE of 10 percent from model 1 is encouraging and seems to indicate that this methodology is viable (table 2). The model requires two predictors, LiDAR-based plot tree height and trees per acre. While LiDAR-based height correlated strongly with field measurement-based height, the association between LiDAR-based trees per acre (TPA) and field measurement-based TPA was weak (fig. 4). Thus, further study is needed to improve the LiDAR-based TPA estimates, which may improve the accuracy of model 1.

Models evaluated in this study did not include variables that could not be obtained by LiDAR data. For example, of the models we evaluated, many did not consider tree density, which plays an important role in predicting diameter. Similar results were experienced in other studies as well (e.g., Cao and Dean 2013). This likely could be attributed to the concept that tree diameter and height are also density dependent. However, because of computerized record-keeping in tools such as geographic information systems, other

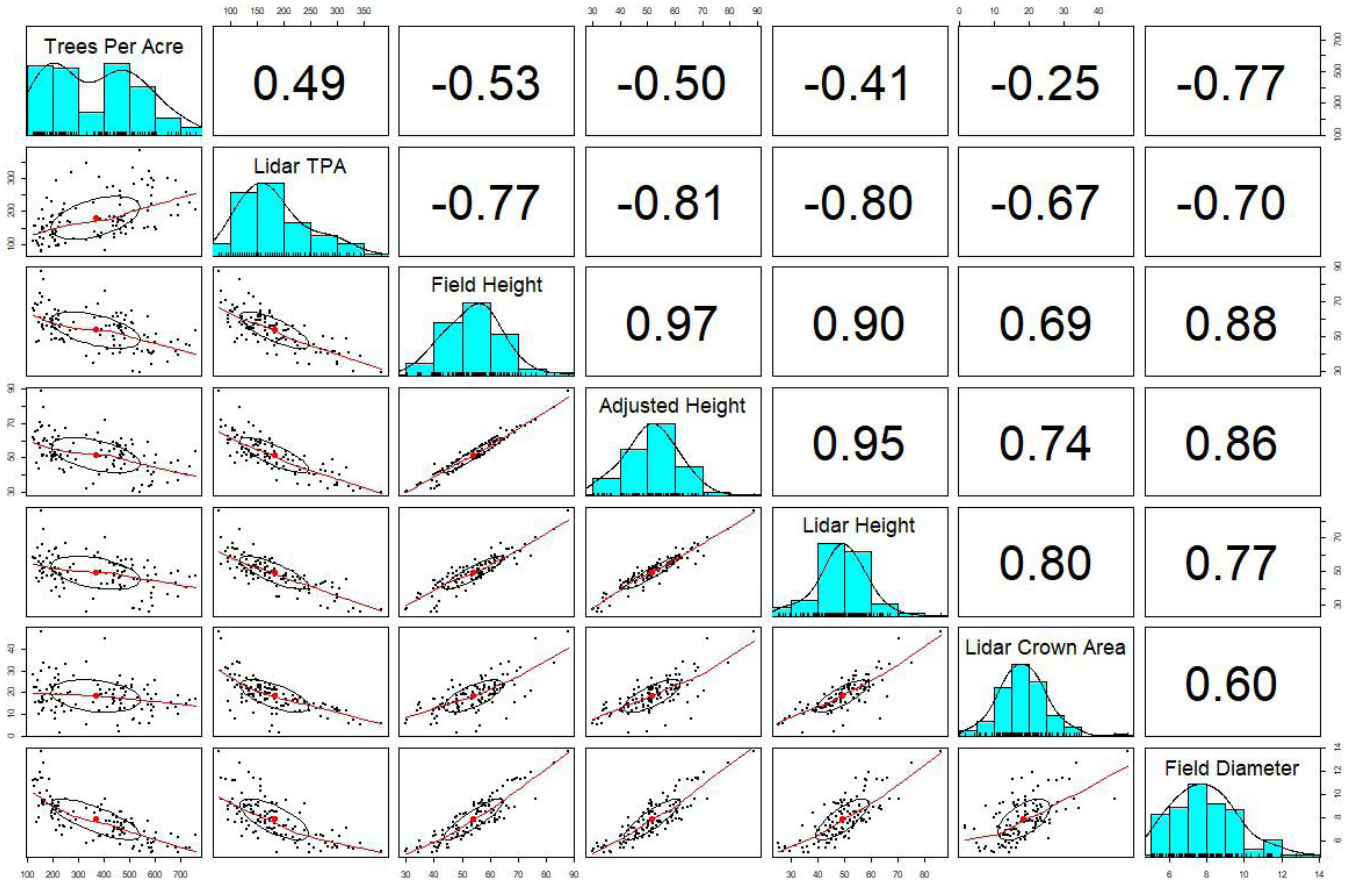

Figure 4—Correlation matrix of all analyzed variables including field-estimated trees per acre, height, height after adjustment, and field-measured diameter at breast height. LiDAR-derived variables included tree height, crown area, and field-measured diameter.

stand attributes are often known. For some, these additional attributes may include stand age, thinning density and timing, site index, and timing of other silvicultural treatments, and such attributes could contribute to further explaining the variation within a height-to-d.b.h. model.

Another factor that potentially could contribute to the accuracy and reliability within the assessed models is the LiDAR dataset itself. The 3DEP is a partnership program which utilizes a collection of elevation data sourced throughout the United States. Data collection for this program relies on assumptions that should imply consistency across the data collections. However, the LiDAR data is collected at different times and by various vendors. Given that this is a cooperative program, 3DEP has developed a baseline program-level set of specifications that are required for data to be considered for inclusion into the program. These specifications include a point spacing of 2.3 feet, an average of 0.19 points per square foot, and a vertical accuracy RMSE no greater than 0.3 feet as of the April 2022 revision (USGS 2022). With only 0.19 points per square foot, it is likely that the LiDAR scan failed to record the absolute top of every tree. Magnussen and others (1999) noted that laser scanners tend to miss treetops and underestimate the standard deviation of derived tree heights. Although the 3DEP LiDAR does not have a high pulse density (only 0.23 pulses per square foot) it did prove to be suitable for the purpose of this study. The relative success of this low-density LiDAR data favors the use of higher-density LiDAR to collect measurements of forest stands.

CONCLUSION

These results demonstrate the viability of estimating stand-level loblolly pine d.b.h. from LiDAR-derived variables and hence may contribute valuable information towards making forest management decisions. Our findings emphasize the value and potential of LiDAR as a tool to improve efficiency of forest inventories and support sustainable forestry practices. As technology continues to advance, utilizing LiDAR to assist in forest mensuration can help foresters meet the demands of the forestry industry.

LITERATURE CITED

Aubry-Kientz, M.; Dutrieux, R.; Ferraz, A. [and others]. 2019. A comparative assessment of the performance of individual tree crowns delineation algorithms from ALS data in tropical forests. Remote Sensing. 11(9): 1086. 21 p. https://doi.org/10.3390/rs11091086.

Avery, T.E. 1975. Natural resources measurements. 2nd ed. New York, NY: McGraw-Hill. 339 p.

Cao, Q.V.; Dean, T.J. 2013. Predicting diameter at breast height from total height and crown length. In: Guldin, J.M., ed. Proceedings of the 15th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-175. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 201–205.

Coble, D.W.; McTague, J.P.; Weng, Y. 2017. A whole-stand growth and yield model for intensively and extensively managed loblolly pine plantations in east Texas prior to first thin. Rep. 72. Nacogdoches, TX: Stephen F. Austin State University, Arthur Temple College of Forestry and Agriculture, East Texas Pine Plantation Research Project. 25 p.

Curtis, R.O. 1967. Height-diameter and height-diameter-age equations for second-growth Douglas-fir. Forest Science. 13(4): 365–375.

Hanus, M.L.; Marshall, D.D.; Hann, D.W. 1999. Height-diameter equations for six species in the coastal regions of the Pacific Northwest. Res. Contrib. 25. Corvallis, OR: Forest Research Laboratory, Oregon State University. 11 p.

Hartsell, A.J.; Conner, R.C. 2013. Forest area and conditions: a 2010 update of chapter 16 of the Southern Forest Resource Assessment. e-Gen. Tech. Rep. SRS-174. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 32 p. https://doi.org/10.2737/SRS-GTR-174.

Hyyppä, J.; Kelle, O.; Lehikoinen, M.; Inkinen, M. 2001. A segmentation-based method to retrieve stem volume estimates from 3-D tree height models produced by laser scanners. IEEE Transactions on Geoscience and Remote Sensing. 39(5): 969–975. https://doi.org/10.1109/36.921414.

Hyyppä, J.; Pyysalo, U.; Hyyppä, H.; Samberg, A. 2000. Elevation accuracy of laser scanning-derived digital terrain and target models in forest environment. EARSeL eProceedings 1: 139–147. http://eproceedings.uni-oldenburg.de/website/vol01_1/01_1_hyyppae1.pdf.

Koch, B.; Heyder, U.; Weinacker, H. 2006. Detection of individual tree crowns in airborne lidar data. Photogrammetric Engineering and Remote Sensing. 72(4): 357–363. https://doi.org/10.14358/PERS.72.4.357.

Larjavaara, M.; Muller-Landau, H.C. 2013. Measuring tree height: a quantitative comparison of two common field methods in a moist tropical forest. Methods in Ecology and Evolution. 4(9): 793–801. https://doi.org/10.1111/2041-210X.12071.

Magnussen, S.; Eggermont, P.; LaRiccia, V.N. 1999. Recovering tree heights from airborne laser scanner data. Forest Science. 45(3): 407–422.

Parker, R.C.; Evans, D.L. 2004. An application of LiDAR in a double-sample forest inventory. Western Journal of Applied Forestry. 19(2): 95–101. https://doi.org/10.1093/wjaf/19.2.95.

Plowright, A. 2022. ForestTools: tools for analyzing remote sensing forest data. R package version 1.0.0, https://github.com/andrew-plowright/ForestTools. [Date accessed: August 12, 2022].

Popescu, S.C.; Wynne, R.H.; Nelson, R.F. 2003. Measuring individual tree crown diameter with lidar and assessing its influence on estimating forest volume and biomass. Canadian Journal of Remote Sensing. 29(5): 564–577. https://doi.org/10.5589/m03-027.

Prestemon, J.P.; Abt, R.C. 2002. The southern timber market to 2040. Journal of Forestry. 100(7): 16–22. https://doi.org/10.1093/jof/100.7.16.

Salas, C.; Ene, L.; Gregoire, T.G. [and others]. 2010. Modelling tree diameter from airborne laser scanning derived variables: a comparison of spatial statistical models. Remote Sensing of Environment. 114(6): 1277–1285. https://doi.org/10.1016/j.rse.2010.01.020.

Temesgen, H.; Hann, D.W.; Monleon, V.J. 2006 Regional height-diameter equations for major tree species of southwest Oregon. Western Journal of Applied Forestry. 22(3): 213–219. https://doi.org/10.1093/wjaf/22.3.213.

U.S. Geological Survey [USGS]. 2022. Lidar base specification: revision history. https://www.usgs.gov/ngp-standards-and-specifications/LiDAR-base-specification-revision-history. [Date accessed: September 15, 2022].

White, J.C.; Coops, N.C.; Wulder, M.A. [and others]. 2016. Remote sensing technologies for enhancing forest inventories: a review. Canadian Journal of Remote Sensing. 42(5): 619–641. https://doi.org/10.1080/07038992.2016.1207484.

Zhang, B.; Sajjad, S.; Chen, K. [and others]. 2020. Predicting tree height-diameter relationship from relative competition levels using quantile regression models for Chinese fir (*Cunninghamia lanceolata*) in Fujian Province, China. Forests. 11(2): 183. 15 p. https://doi.org/10.3390/f11020183.

QUANTIFYING FOREST ABOVEGROUND BIOMASS USING REMOTE SENSING AND GROUND MEASUREMENTS FOR MIXED HARDWOOD FORESTS IN EAST TENNESSEE

Sakar Jha, Sheng-I Yang, D. Stuart Hale, David Shoch, Trisha Johnson, Donald G. Hodges, and Kevin P. Hoyt

ABSTRACT

Multispectral data collected from satellite-based sensors serve as a powerful tool for quantifying forest aboveground biomass. In this paper, we evaluated the performance of models that used satellite-image-derived variables and the models that used a combination of satellite-image-derived variables along with stand-level attributes to quantify forest aboveground biomass of natural mixed-hardwood forests. Imagery from Landsat 8 and Sentinel 2 satellite missions were used to derive the remote-sensing variables. We also identified the important predictors for these models using the recursive feature elimination method. The models that included variables derived from satellite images and stand-level attributes performed better than the models that only used variables derived from satellite images.

INTRODUCTION

Forest aboveground biomass (AGB) is quantified using *in situ* forest measurements, which are resource intensive to obtain (Flade and others 2020, Jones and others 2020). Multiple studies have been conducted for quantifying forest AGB using remote sensing as that technology has proven to be easier and more efficient (Csillik and others 2019, Deo and others 2017, Issa and others 2020). Satellite-based multispectral imagery is a cost-efficient alternative for quantifying AGB over a large area. The auxiliary information derived from satellite images captured from numerous satellite missions such as Landsat, Sentinel, and MODIS have all proved that they can quantify AGB at a reasonable precision. López-Serrano and others (2020), for example, used Landsat 8 images to model AGB in temperate forests of Mexico. Pandit and others (2020) used Sentinel 2 images to quantify AGB in the subtropical forests in Nepal. Blackard and others (2008) utilized MODIS data along with forest measurement data to produce the forest AGB map of the entire United States. Most of the studies that used multi-spectral imagery to quantify AGB only used predictors derived from them to model AGB. However, the AGB estimates provided by these models are not precise. To the best of our knowledge, no research has looked at combining stand-level attributes derived from ground-based measurements with variables derived from satellite imagery to quantify AGB in east Tennessee.

The aim of this project was to compare the accuracy and precision of AGB estimates produced by machine-learning models using only remote-sensing predictors and models combining remote-sensing predictors with stand-level attributes for natural mixed hardwood forests

Sakar Jha, Graduate Student, School of Natural Resources, University of Tennessee, Knoxville, TN 37996
Sheng-I Yang, Assistant Professor, School of Natural Resources, University of Tennessee, Knoxville, TN 37996
D. Stuart Hale, Forest Program Manager, US Carbon Markets Team, The Nature Conservancy, Arlington, VA 22203
David Shoch, Technical Director, TerraCarbon LLC, Charlottesville, VA 22902
Trisha Johnson, Senior Manager Operations, TerraCarbon LLC, Sparta, TN 38583
Donald G. Hodges, Professor and Director, School of Natural Resources, University of Tennesse, Knoxville, TN 37996
Kevin P. Hoyt, Center Director, Forest Resources AgResearch and Education Center, University of Tennessee, Knoxville, TN 37996

in east Tennessee. The models were built using the random forest algorithm, and the important variables were identified using the recursive feature elimination method.

METHODS

This study was carried out using forest carbon inventory data from four sites in east Tennessee (fig. 1). A total of 216 fixed-radius measurement plots were measured at four sites: Chestnut Mountain (n = 90), Oak Ridge (n = 29), Cumberland north (n = 57), and Cumberland south (n = 40). Additional details about sampling and measurement can be found in Yang and others (2022). These sites are largely composed of various species of maples (*Acer* spp.), oaks (*Quercus* spp.) and hickories (*Carya* spp.), with some shortleaf pines (*Pinus echinata*) and loblolly pines (*Pinus taeda*). The AGB of the trees was calculated using species-specific biomass equations published by Jenkins and others (2004). Mean tree height, trees/ha, and basal area (m^2/ha) were calculated for every plot.

Satellite images from Landsat 8 mission with operational land imager sensors and Sentinel 2 mission with multispectral imager sensors were downloaded for the sites during the leaf-on season. The images from both the satellite missions had a cloud cover of less than 5 percent. From these satellite images the blue, green, red, near infrared (NIR), and shortwave infrared 1 and 2 (SWIR 1 and SWIR 2) were extracted. Vegetation has high reflectance at NIR and low reflectance at the red region of the electromagnetic spectrum. Hence, the difference between these two bands describes the characteristics of vegetation. Using this property of vegetation, several vegetation indices have been derived from satellite images. In this study, six vegetation indices were calculated from the Landsat 8 and Sentinel 2 satellite images using the expressions in table 1. These

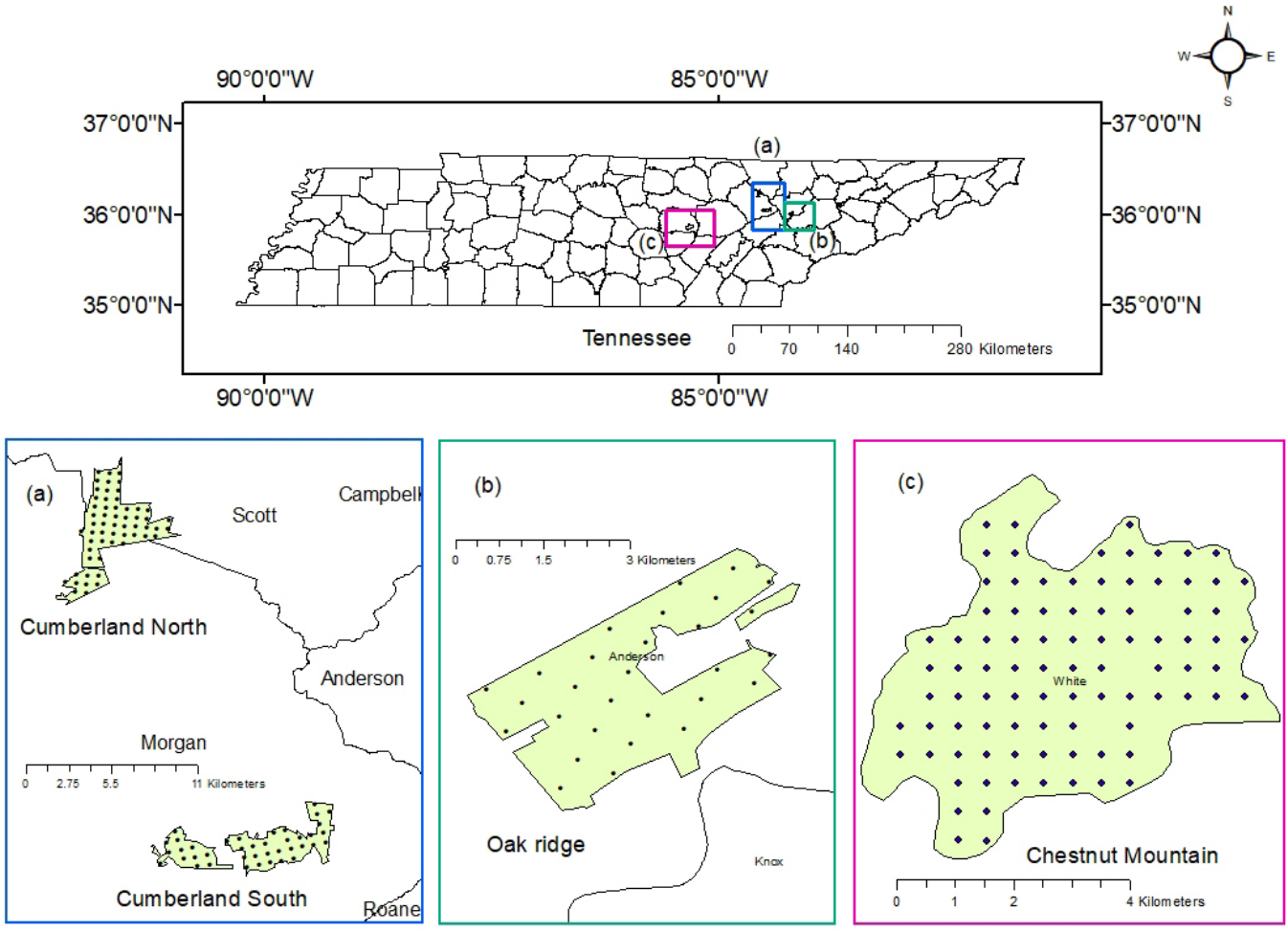

Figure 1—Map of the four study areas. Dots represent the location of forest measurement plots.

Table 1—Stand-level attributes and vegetation indices calculated from ground-based measurements and satellite images

Type	Variables	Expression	Reference
Stand-level attributes	TPH	Trees per hectare	–
	BA	Basal area (m^2/ha)	–
	Mean tree height	Average height of inventoried trees (m)	–
Spectral bands	Blue, Green, Red, NIR, SWIR-1, SWIR-2	–	–
Vegetation indices	Normalized difference vegetation index	(NIR-Red)/(NIR+Red)	Rouse 1974
	Soil-adjusted vegetation index	[(NIR-Red)/(NIR+Red+0.5)] *1.5	Huete 1988
	Enhanced vegetation index	2.5*((NIR-Red)/(NIR+6*Red-7.5*Blue) +1))	Huete and others 2002
	Atmospherically resistant vegetation index	(NIR-(Red-1*(Blue-Red)))/(NIR+(Red-1*(Blue-Red)))	Kaufman and Tanré 1992
	Ratio vegetation index	NIR/Red	Rondeaux and others 1996
	Difference vegetation index	NIR-Red	Tucker 1979

Blue = blue band; Green = green band; Red = red band; NIR = near infrared band; SWIR-1 and SWIR2 = shortwave infrared bands 1 and 2 from the satellite images.

vegetation indices were extracted from the pixels that directly correspond to each plot location. Similarly, texture metrics were calculated for each of the six satellite image bands from Landsat 8 and Sentinel 2 satellite images using the gray level co-occurrence matrix algorithm. Based on previous studies (Dube and Mutanga 2015, Li and others 2020, Pandit and others 2020), seven texture metrics (contrast, dissimilarity, mean, homogeneity, angular second moment, entropy, and variance) were calculated with a moving window size of 3 by 3 (table 2). Similar to vegetation indices, the values of texture metrics were extracted from the pixels that directly corresponded to each plot location.

Table 2—Expressions of seven texture metrics calculated from the satellite images as given by Haralick and others (1973)

Texture metric	Expression
Contrast	$\sum_{x,y=0}^{N-1} P_{x,y}(x-y)^2$
Dissimilarity	$\sum_{x,y=0}^{N-1} P_{x,y}\lvert x-y\rvert$
Mean	$\sum_{x,y=0}^{N-1} y(P_{x,y})$
Homogeneity	$\sum_{x,y=0}^{N-1} P_{x,y}/(1+(x-y)^2)$
Angular second moment	$\sum_{x,y=0}^{N-1} P_{x,y}^2$
Entropy	$\sum_{x,y=0}^{N-1} P_{x,y}(-\ln P_{x,y})$
Variance	$\sum_{x,y=0}^{N-1} P_{x,y}\,(y-\text{Mean})^2$

$P_{x,y}$ = the probability of reflectance value of pixel x and pixel y co-occurring in the moving window.
x and y = the reflectance value of a pixel x and pixel y in the moving window.
N = number of gray levels in the satellite band images.

The 6 spectral bands, 6 vegetation indices, 42 textural metrics, and 3 stand-level attributes were used to predict the AGB of the sites. The ground-based AGB calculated from the forest inventory data was used as the response variable. These variables were modeled using a random forest (RF) algorithm. It is a nonparametric approach to carrying out classification and regression problems, which employs bagging and boosting techniques to build decision trees (Breiman 2001). Within the RF algorithm, decision trees are created by randomly selecting two-thirds of the data to train the model, and the remaining one-third of the data to assess the model errors. Two sets of models were built for each satellite image (Landsat 8 and Sentinel 2): one using only satellite-image-derived variables as predictors and one using both satellite-image-derived variables and stand-level attributes.

The percent root mean square error (RMSE), percent mean bias (MB), and coefficient of determination (R^2) were calculated to evaluate the model performance. They were computed as:

$$RMSE(\text{percent}) = 100 * \sqrt{\frac{\sum_{k=1}^{n} ((Y_k - \widehat{Y_k})/Y_k)^2}{n}} \quad (1)$$

$$MB(\text{percent}) = 100 * \frac{\sum_{k=1}^{n} (Y_k - \widehat{Y_k})}{n * Y_k} \quad (2)$$

$$R^2 = 1 - \frac{\sum_{k=1}^{n} (Y_k - \widehat{Y_k})^2}{\sum_{k=1}^{n} (Y_k - \bar{Y})^2} \quad (3)$$

where

Y_k is the observed AGB of plot *k*,

$\widehat{Y_k}$ is the predicted AGB of plot *k*,

$\bar{Y}$ is the average AGB from all plots, and

n is the number of plots.

To identify the important variables for each RF model, the recursive feature elimination (RFE) method was used. The RFE method iteratively selects a subset of predictors by eliminating the weakest predictor from the model until the subset with the lowest RMSE is reached. The RFE method was implemented using the 5-fold crossvalidation method to select the subset of predictor variables that produced the lowest RMSE.

RESULTS AND DISCUSSION

Performance of Models

Among the two sets of models, the model that used only variables extracted from the Landsat 8 satellite image as predictors performed better than the model that used only variables extracted from Sentinel 2 image (fig. 2). The Landsat-8-only

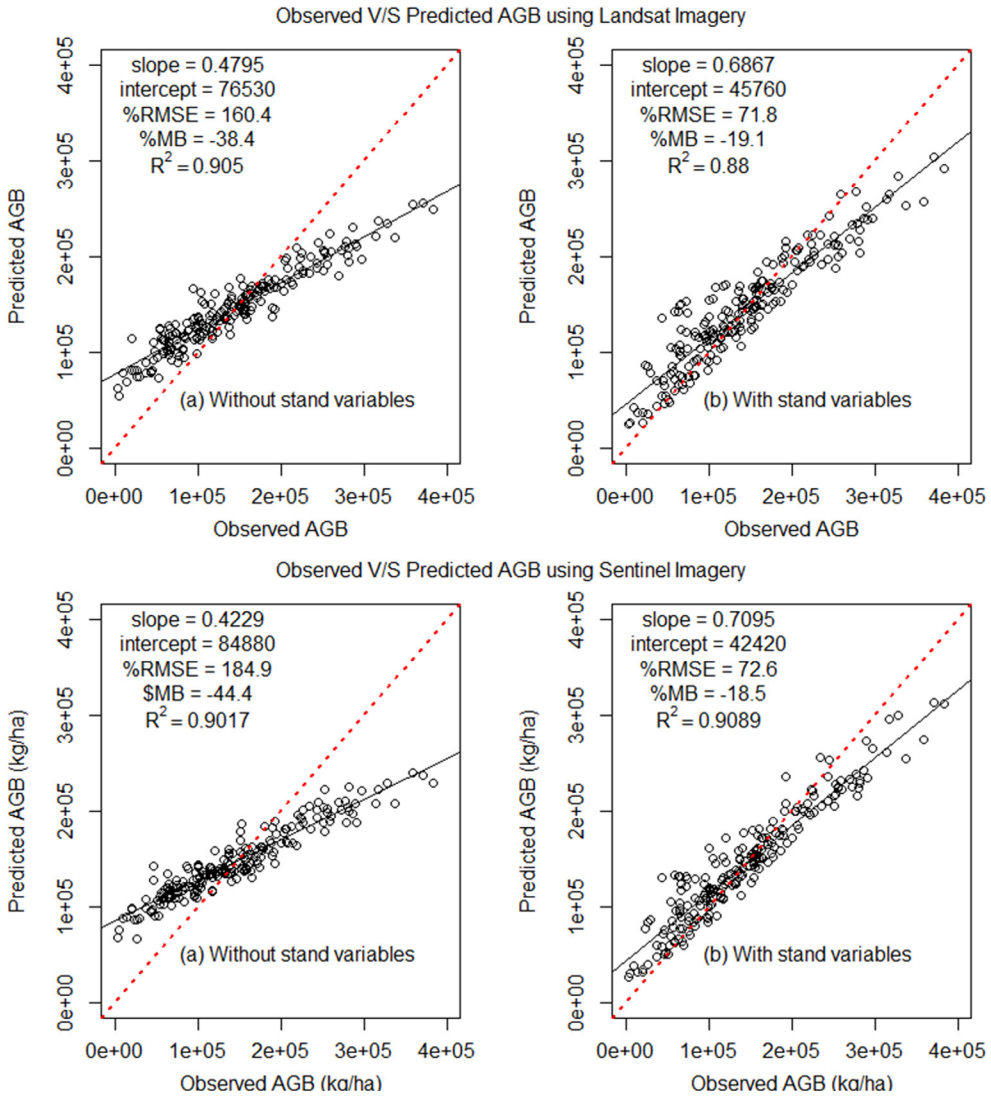

Figure 2—Observed versus predicted aboveground biomass (AGB) for the random forest models using Landsat 8 variables (top row) and Sentinel 2 variables (bottom row).

model had an RMSE of 160.4 percent and an MB of -38.4 percent, whereas the Sentinel-2-only model had a RMSE of 184.9 percent and MB of -44.4 percent. The slope of the best fit line between the observed and predicted AGB for the Landsat-8-only model was greater than that for the Sentinel-2-only model. When the remote-sensing variables were combined with stand-level attributes, the Sentinel 2 plus stand-level attributes model produced lower percent MB than Landsat 8 plus stand-level attributes model. The model with Sentinel 2 variables plus stand-level attributes had a RMSE of 72.6 percent and MB of -18.5 percent compared to 71.8 percent and -19.1 percent for the Landsat 8 plus stand-level attributes model. The slope of the best fit line between the observed and predicted AGB for Sentinel 2 plus stand-level attributes model was higher than that for the Landsat 8 plus stand-level attributes model. Figure 3 shows the spatial pattern of predicted aboveground biomass for the Sentinel 2 plus stand-level attributes model.

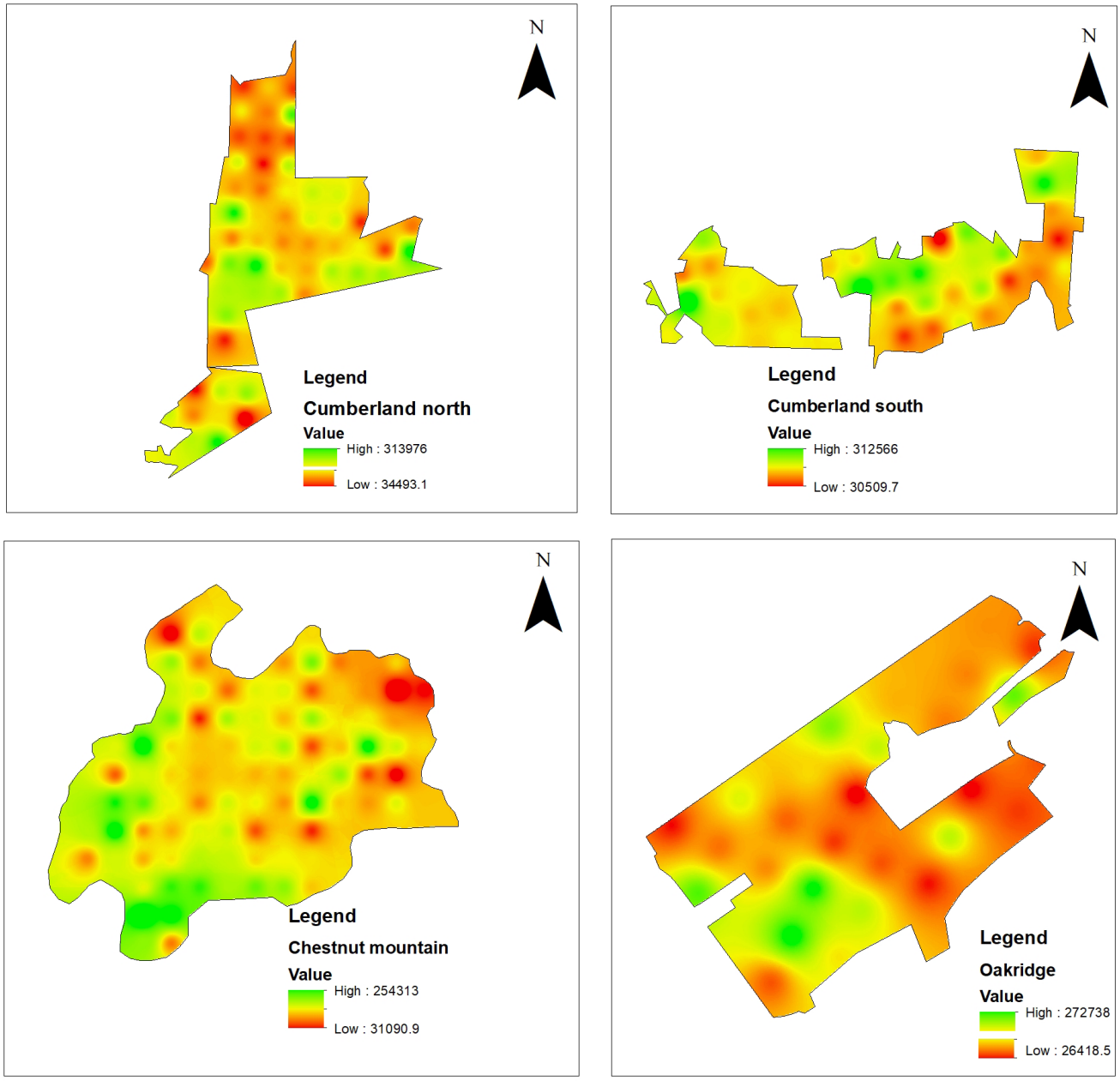

Figure 3—Spatial pattern of the predicted aboveground biomass for the Sentinel 2 plus stand-level attributes model for each of the four study areas. Areas with low aboveground biomass are shown in red, while areas with high aboveground biomass are shown in green.

Feature Importance

The RFE method selected a different subset of variables by removing the weakest feature for each of the four models (table 3). For the models with only remote-sensing variables, the texture metrics in green, blue, NIR, and SWIR bands were found to be more important than other vegetation indices and texture metrics. Among the seven different types of texture metrics, there was no clear pattern of one being more important than the rest. For models with both remote-sensing variables plus stand-level attributes, the stand-level attributes were more important, which were followed by texture metrics in green, blue, NIR, and SWIR bands. For the Sentinel-2-only model, 52 of the 54 variables were found to be important whereas for the Landsat-8-only model, only 10 of the variables were found to be important. Notably, in this study, stand-level attributes were calculated from field-obtained data. However, stand-level attributes can also be derived from active remote sensors, such as those developed from light detection and ranging (LiDAR) data. It would be worthwhile for further studies to evaluate the impact of stand-level attributes computed from different approaches on AGB estimation.

CONCLUSIONS

Inclusion of stand-level attributes as predictors along with remote-sensing variables can improve the estimation of forest AGB. The Landsat-8-only model performed better than the Sentinel-2-only model. When stand-level attributes are included, the Sentinel 2 plus stand-level attributes model performed better than the Landsat 8 plus stand-level attributes model. Texture metrics are important remote-sensing predictors for estimating AGB, particularly texture metrics of green, blue, NIR, and SWIR bands.

REFERENCES

Blackard, J.A.; Finco, M.V.; Helmer, E.H. [and others]. 2008. Mapping U.S. forest biomass using nationwide forest inventory data and moderate resolution information. Remote Sensing of Environment. 112(4): 1658–1677. https://doi.org/10.1016/j.rse.2007.08.021.

Breiman, L. 2001. Random forests. Machine Learning. 45: 5–32.

Csillik, O.; Kumar, P.; Mascaro, J. [and others]. 2019. Monitoring tropical forest carbon stocks and emissions using Planet satellite data. Scientific Reports. 9: 17831 12 p. https://doi.org/10.1038/s41598-019-54386-6.

Deo, R.K.; Russell, M.B.; Domke, G.M. [and others]. 2017. Evaluating site-specific and generic spatial models of aboveground forest biomass based on Landsat time-series and LiDAR strip samples in the eastern USA. Remote Sensing. 9: 598. 23 p. https://doi.org/10.3390/rs9060598.

Dube, T.; Mutanga, O. 2015. Investigating the robustness of the new Landsat-8 Operational Land Imager derived texture metrics in estimating plantation forest aboveground biomass in resource constrained areas. ISPRS Journal of Photogrammetry and Remote Sensing. 108: 12–32. https://doi.org/10.1016/j.isprsjprs.2015.06.002.

Flade, L.; Hopkinson, C.; Chasmer, L. 2020. Allometric equations for shrub and short-stature tree aboveground biomass within boreal ecosystems of northwestern Canada. Forests. 11: 1207. 16 p. https://doi.org/10.3390/f11111207.

Haralick, R.M.; Shanmugam, K.; Dinstein, I. 1973. Textural features for image classification. IEEE Transactions on Systems, Man, and Cybernetics. SMC-3(6): 610–621. https://doi.org/10.1109/TSMC.1973.4309314.

Huete, A.; Didan, K.; Miura, T. [and others]. 2002. Overview of the radiometric and biophysical performance of the MODIS vegetation indices. Remote Sensing of Environment. 83: 195–213. https://doi.org/10.1016/S0034-4257(02)00096-2.

Table 3—Optimal predictors indentified for the four models using the recursive feature elimination method

Models	Optimal variables
Landsat 8 only	variance_green, green, homogeneity_IR, mean_green, second_moment_IR, mean_blue, SWIR1, variance_blue, dissimilarity_IR, homogeneity_green
Landsat 8 plus stand-level attributes	BA, Mean tree height, TPH, variance_green, green, mean_blue
Sentinel 2 only	contrast_SWIR2, green, dissimilarity_SWIR2, entropy_SWIR1, entropy_IR, homogeneity_SWIR1, second_moment_SWIR1, second_moment_IR, contrast_SWIR1, entropy_SWIR2, entropy_Red, second_moment_SWIR2, variance_red, blue, entropy_green, mean_red, dissimilarity_SWIR1, second_moment_green, mean_SWIR2, mean_green, homogeneity_IR, variance_green, red, homogeneity_SWIR2, contrast_IR, contrast_red, variance_SWIR2, entropy_blue, variance_blue, second_moment_red, dissimilarity_red, homogeneity_red, contrast_green, arvi, SWIR2, evi, rvi, dissimilarity_IR, second_moment_blue, homogeneity_blue, mean_IR, savi, ndvi, mean_blue, homogeneity_green, variance_SWIR1, dissimilarity_blue, IR, contrast_blue, SWIR1, mean_SWIR1, variance_IR, dissimilarity_green
Sentinel 2 plus stand-level attributes	BA, Mean tree height, entropy_IR, TPH, second_moment_IR, mean_SWIR1

TPH = trees/ha; BA = basal area (m^2/ha).

Huete, A.R. 1988. A soil-adjusted vegetation index (SAVI). Remote Sensing of the Environment. 25: 295–309. https://doi.org/10.1016/0034-4257(88)90106-X.

Issa, S.; Dahy, B.; Ksiksi, T.; Saleous, N. 2020. A review of terrestrial carbon assessment methods using geo-spatial technologies with emphasis on arid lands. Remote Sensing. 12: 2008. 32 p. https://doi.org/10.3390/rs12122008.

Jenkins, J.C.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A. 2004. Comprehensive database of diameter-based biomass regressions for North American tree species. Gen. Tech. Rep. NE-319. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station. 45 p. https://doi.org/10.2737/NE-GTR-319.

Jones, A.R.; Raja Segaran, R.; Clarke, K.D. [and others]. 2020. Estimating mangrove tree biomass and carbon content: a comparison of forest inventory techniques and drone imagery. Frontiers in Marine Science. 6: 784. 13 p. https://doi.org/10.3389/fmars.2019.00784

Kaufman, Y.J.; Tanré, D. 1992. Atmospherically resistant vegetation index (ARVI) for EOS-MODIS. IEEE Transactions on Geoscience and Remote Sensing. 30(2): 261–270. https://doi.org/10.1109/36.134076.

Li, Y.; Li, M.; Li, C.; Liu, Z. 2020. Forest aboveground biomass estimation using Landsat 8 and Sentinel-1A data with machine learning algorithms. Scientific Reports. 10: 9952. 12 p. https://doi.org/10.1038/s41598-020-67024-3

López-Serrano, P.M.; Domínguez, J.L.C.; Corral-Rivas, J.J. [and others]. 2020. Modeling of aboveground biomass with Landsat 8 OLI and machine learning in temperate forests. Forests. 11: 11. 18 p. https://doi.org/10.3390/f11010011.

Pandit, S.; Tsuyuki, S.; Dube, T. 2020. Exploring the inclusion of Sentinel-2 MSI texture metrics in above-ground biomass estimation in the community forest of Nepal. Geocarto International. 35(16): 1832–1849. https://doi.org/10.1080/10106049.2019.1588390.

Rondeaux, G.; Steven, M.; Baret, F. 1996. Optimization of soil-adjusted vegetation indices. Remote Sensing of Environment. 55(2): 95–107. https://doi.org/10.1016/0034-4257(95)00186-7.

Rouse, J.W., Jr. 1974. Monitoring the vernal advancement of retrogradation (green wave effect) of natural vegetation. Type 1 Progress Report Number 7. College Station, TX: Texas A&M University, Remote Sensing Center. 8 p. https://ntrs.nasa.gov/api/citations/19740022555/downloads/19740022555.pdf.

Tucker, C.J. 1979. Red and photographic infrared linear combinations for monitoring vegetation. Remote Sensing of the Environment. 8: 127–150. https://doi.org/10.1016/0034-4257(79)90013-0.

Yang, S.; Cao, Q.V.; Shoch, D.T.; Johnson, T. 2022. Characterizing stand and biomass tables from diameter distribution models: a case study for mixed-hardwood forests in eastern Tennessee, USA. Forest Science. 68(1): 8–16. https://doi.org/10.1093/forsci/fxab051.

Fire

PREDICTING FIRE INTENSITY USING VEGETATION, FUEL, AND WEATHER VARIABLES

John Craycroft and Callie Schweitzer

ABSTRACT

When applying prescribed fire to achieve various management objectives, forest managers benefit from being able to accurately predict fire behavior with site- and time-specific data. At the William B. Bankhead National Forest in northcentral Alabama, we used three levels of thinning (none, light, heavy) and three fire return intervals (no fire, 9-year return, 3-year return) in mixed pine-oak (*Pinus* spp.-*Quercus* spp.) stands to move stands toward more desirable structures. After 38 fires over 12 years, we amassed a substantial store of thermocouple temperatures and burn condition data. We modeled maximum thermocouple temperature as a surrogate for fire intensity by using a mixed effects model that incorporated stand and vegetation variables, fuel loading measurements, and weather data. The random portion of our model estimated low correlation of maximum temperatures within stands, but moderate correlation within plots (plots were nested within stands). The fixed effects portion of our model revealed associations of maximum temperature with several vegetation (importance values of oaks and pines, seedling counts of pines and other species), fuels (loadings of both bark and duff), and weather variables (average ambient temperature and relative humidity over the 24 hours prior to ignition, fuel temperature, and fuel moisture).

INTRODUCTION

Forest managers must frequently balance multiple objectives in their stewardship duties. The general concept of achieving and maintaining healthy and resilient forests may imply a number of specific goals, including but not limited to maximizing floral and fauna biodiversity, managing the economic timber resource, facilitating recreational opportunities, enhancing forests' roles in providing clean air and water, and establishing wildlife habitat (USDA Forest Service 2004). Of the many tools forest managers have at their disposal, prescribed fire holds a unique place. On a general level, with wildland fires having been artificially suppressed for much of the 20th century, the use of prescribed fire can provide a return to a more natural state, restoring a process that has been diminished or lost. More specifically, prescribed fire may increase light in the understory, change the prevailing species mix, and alter fuel loading and structure (Brose and others 2013).

While prescribed fire is a tool that may offer many benefits, the relationship of fire to other elements within an ecosystem is complex. Forest stand characteristics, fuel composition, and fire behavior all interrelate in feedback loops (fig. 1). In this context, applying prescribed fire to achieve specific silvicultural outcomes is anything but simple. Consequently, the more that can be learned about the relationships among these elements, the better equipped forest managers will be to effectively apply prescribed fire. Our analysis seeks to inform one aspect of this system, by identifying associations between fire behavior and vegetation. Specifically, the primary objective of this analysis was to assess the null hypothesis of no association between the forest's vegetation species mix and maximum thermocouple temperatures, while controlling for other factors that affect fire intensity.

John Craycroft, ORISE Postdoctoral Fellow, U.S. Department of Agriculture, Forest Service, Southern Research Station, Huntsville, AL 35801
Callie Schweitzer, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Huntsville, AL 35801

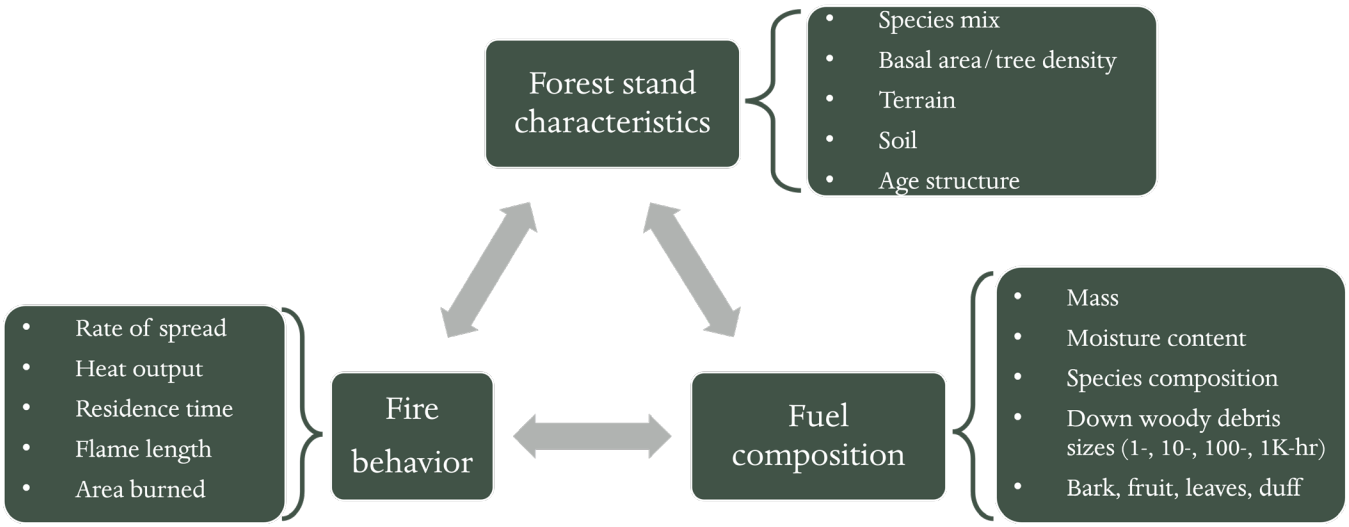

Figure 1—Forest-fuels-fires feedback loops: predicting fire behavior requires understanding how forest stand characteristics and fuel composition both influence and are influenced by fire behavior. Each of the three constructs in this feedback loop (fire behavior, forest stand characteristics, and fuel composition) is composed of several individual empirical metrics. For example, fire behavior is described by metrics such as rate of spread, heat output, residence time, flame length, and area burned.

METHODS

The William F. Bankhead National Forest (BNF) is a 180,000-acre national forest located in northcentral Alabama. The treatment stands, all located in the northern portion of the BNF, were selected to be similar based on average stand age, composition, and size. They ranged in area from 22 to 46 acres, in age from 30 to 60 years old, and in pretreatment basal area from 143 to 188 square feet per acre. The study sites were mixed pine-hardwood forests, dominated by loblolly (*Pinus taeda*), with lesser amounts of Virginia (*P. virginiana*) and shortleaf (*P. echinata*) pine. Upland oaks were common and included chestnut (*Quercus prinus*), white (*Q. alba*), northern red (*Q. rubra*), scarlet (*Q. coccinea*), black (*Q. velutina*), and southern red (*Q. falcata*) oaks. Other hardwoods included yellow-poplar (*Liriodendron tulipifera*), red maple (*Acer rubrum*), and black cherry (*Prunus serotina*), among others.

Part of the original goal of the study was to test varying levels of management disturbances for their effectiveness at shifting the species mix toward greater hardwood dominance, particularly oak. The nine treatment levels (eight treatments and one control) were the result of a 3 by 3 factorial design incorporating two disturbance factors at three levels each. The three thinning levels (mechanical thinning from below) were no thinning, light thinning [target residual basal area (BA) of 75 square feet per acre], and heavy thinning (target residual BA of 50 square feet per acre). Prescribed burns occurred at three frequencies: no burns, infrequent burns (every 9 years), and frequent burns (every 3 years). Only the infrequent and frequent burn treatments are relevant for this analysis, hence six treatments were included in the data used here.

Four replications (blocks) were initiated from 2006 through 2008; each block consisted of nine stands, one for each treatment level. Each stand was surveyed and marked with five permanent measurement plots. Thinned stands received a single thinning treatment to the desired residual BA prior to burning. A total of 38 landscape-scale prescribed fires, usually encompassing multiple treatment stands, were conducted over 12 years. The low- to moderate-severity prescribed fires all took place during the dormant season from January through early April and used backing fires or strip head fires to ensure that only surface fires occurred. For details, see Schweitzer and others (2016).

Data Collection

Dependent variable: maximum thermocouple temperature

Fire temperature data were recorded for most prescribed fires. We followed methods from Iverson and others (2004) for distributing six to eight HOBO data recorders (HOBO U12 Series Datalogger) and thermocouple probes (TCPs) (HOBO TCP6-12 Probe Thermocouple Sensor, Onset Computer Corporation, Cape Cod, MA) per measurement plot per burn. Dataloggers were buried approximately 6 inches deep, and litter and duff layers were replaced over mineral soil after burial. The thermocouple probes were installed in a vertical orientation with the tips approximately 10 inches above the litter layer. Temperatures were recorded together with date-time stamps every 2 seconds from the morning before through the night following the burn. For this analysis, only the maximum recorded temperatures were used, and no data was used if the maximum temperature did not exceed 90 °F. Although we did not view the burn temperatures as precise measurements of fire intensity, in the sense of heat transfer, we maintain that they provided a useful index for comparing fire intensity among locations or among different fires. Thermocouple-based data also carry further advantages, which are addressed in more detail in the Discussion.

Covariates: stand data

After stands were assigned to treatments, each was surveyed using five permanent measurement plots. For each plot, soil type, aspect, slope percentage, and elevation were recorded.

Covariates: weather data

Hourly weather data for each burn date and 7 days prior to each burn were obtained from the MesoWest weather database maintained by University of Utah (2022). Relevant data fields included air temperature (°F), relative humidity (RH, percent), wind speed (miles per hour), wind direction (degrees), wind gust maximum speed (miles per hour), precipitation accumulation (inches), fuel temperature (°F), and fuel moisture (percent). The preburn date air temperature, relative humidity, and precipitation variables were averaged over varying time windows (e.g., 24 hours, 48 hours, 72 hours, etc.) prior to ignition to investigate the effect that weather preceding the burn date would have on the fire temperatures. Burn date weather was matched to the fire temperature data according to the weather time stamp most closely matching fire ignition time.

Covariates: fuels data

In each permanent measurement plot, fuel loads were measured after leaf fall and before the prescribed burn for that cycle. To collect these data, two 50-foot planar intercepts were established in each plot (Brown 1974). For each transect, two distances were randomly selected. At each distance, two 1-foot square samples were marked, one at 3 feet to the left, the other at 3 feet to the right of the planar intercept. In these 1-square-foot sample areas, all down woody debris was collected and separated into the following categories: bark, fruit, leaves, duff, 1-hour fuels, and 10-hour fuels. At the lab, samples were oven dried in paper bags at 175 °F for 48 hours (72 hours for duff); immediately after drying, samples were weighed to 0.00035 ounces (0.01 grams) using an Ohaus 810-g precision scale. The fuels variables were invariably and highly right skewed; therefore, model building used the natural log-transformed versions of these measures.

Independent variables of primary interest: vegetation data

The primary research interest lay in determining associations between vegetation data and prescribed fire intensity. All vegetation data were collected prior to thinning and then in the summer before each prescribed burn. Regeneration data (understory reproduction, i.e., seedlings and stems <1.5 inches diameter at breast height, d.b.h.) were tallied within a 1/100-acre subplot with center at plot center. Tallies were recorded by species and size class (this analysis uses only the aggregate counts across all the reproduction size classes and divides the total counts by 1,000). Midstory trees (d.b.h. ≥1.5

inches, but <5.6 inches) were tagged and counted within a 1/40-acre subplot with center at plot center. Overstory trees (d.b.h. ≥5.6 inches) were tagged and counted within a 1/5-acre subplot with center at plot center. For the midstory and overstory, species and d.b.h. were recorded for every tallied tree in the respective subplots. For this analysis, species were aggregated into four groups: oak, pine, red maple, and other species. An index, the importance value (IV), was used to aggregate the high dimensional vegetation data for the midstory and overstory. The IV is an equally weighted average of the relative density (stems per acre) and relative dominance (BA per acre) of each species group (Cottam and Curtis 1956, Crow and others 2002).

This analysis utilizes data through four burn cycles for all four blocks; thus, frequently burned stands received four fires, while infrequently burned stands received two fires. Further background on the study and analysis on other research questions may be found elsewhere (Craycroft and Schweitzer 2023; Schweitzer and others 2016, 2019).

Statistical Analysis

Although we had a high number of temperature measurements, the data were not independent. There was spatial correlation, with plots nested within stands. There was also temporal correlation, because stands within the same blocks were generally ignited under the same weather conditions. And there were repeated measures over time. Hence it was critical to not treat the data as a simple random sample. We therefore employed a mixed effects model approach to address these various sources of dependence, as specified in the following model statement:

$$\ln T_{ijk} = X_{ijk}\beta + \alpha_k + \alpha_{j|k} + \epsilon_{ijk}$$

where

ln is the natural log operator

T_{ijk} is the maximum TCP temperature for observation i, in plot j, in stand k

X_{ijk} is the vector of covariate values associated with observation ijk

β is the vector of coefficients capturing the relationships between the fixed effects and the outcome

α_k is the random intercept for stand k

$\alpha_{j|k}$ is the random intercept for plot j nested within stand k

ϵ_{ijk} is the true noise term for each observation

We treated stand (the combination of treatment and block) and plot nested within stand as random effects, while all other covariates plus year, day of year, and time of day were treated as fixed effects.

To develop this mixed effects model, we used the protocol described by Zuur and others (2009). First, we estimated a model with all possible covariates, plus all reasonable interaction terms. Using that model, we sought the optimal random structure using restricted maximum likelihood and Akaike's Information Criteria (AIC). Third, given the optimal random structure, we sought the optimal fixed structure using maximum likelihood to compare nested models and removed nonsignificant covariates. A threshold of $\alpha = 0.05$ was used for assessing statistical significance. In the final model validation step for the best fitting model, plots of residuals against predicted values, against included covariates, and against excluded covariates were all examined for systematic unexplained variation. Analysis was conducted in R Version 4.1.1 (R Core Team 2021).

RESULTS

The 38 prescribed fires over the course of the four burn cycles included in this analysis generated 1,389 individual temperature time series. Average maximum TCP temperature was 344 °F [±166 °F, standard deviation (SD)]. The majority of the maximum temperatures were in the 200–500 °F range, but some reached above 800 °F (fig. 2). Distributions of maximum burn temperatures did not vary greatly among the different treatments (fig. 3).

The best fitting mixed effects model included the covariates summarized in table 1. The only vegetation variables that displayed a statistically significant association with the maximum TCP

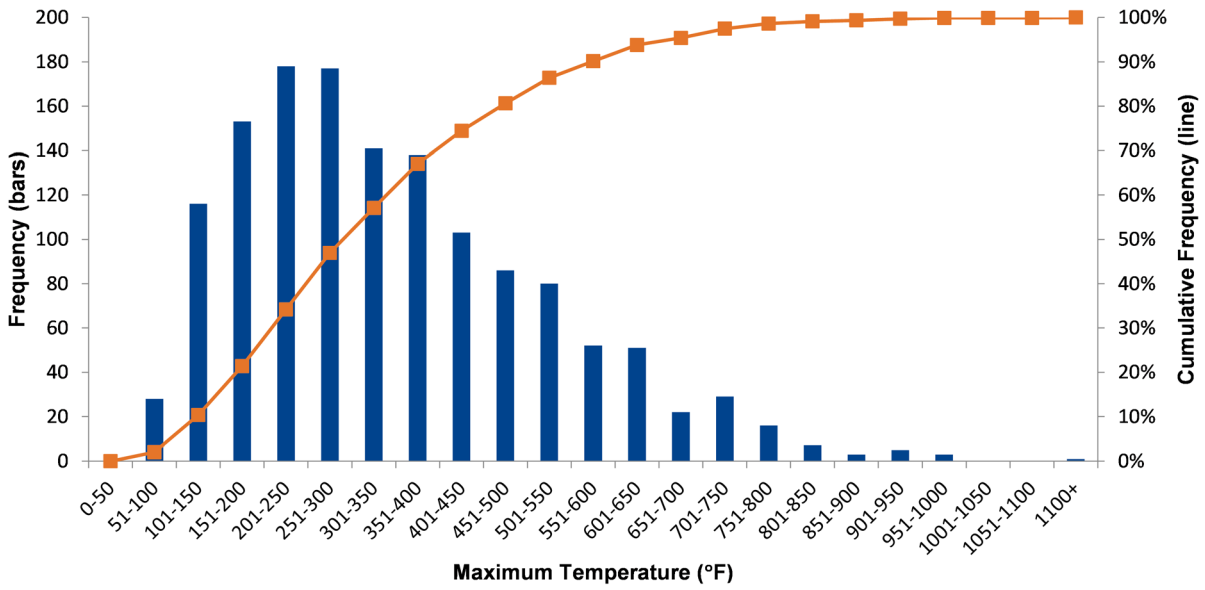

Figure 2—Histogram of maximum thermocouple probe (TCP) temperatures for 38 dormant season prescribed fires on the William B. Bankhead National Forest, AL. Fires included one or more study stands, each stand contained five measurement plots, and each plot received from six to eight probes; this resulted in 1,389 maximum TCP values. The bars in the histogram represent the frequency (left axis) of maximum TCP values within 50-degree intervals, while the line on the graph illustrates the cumulative frequency (right axis).

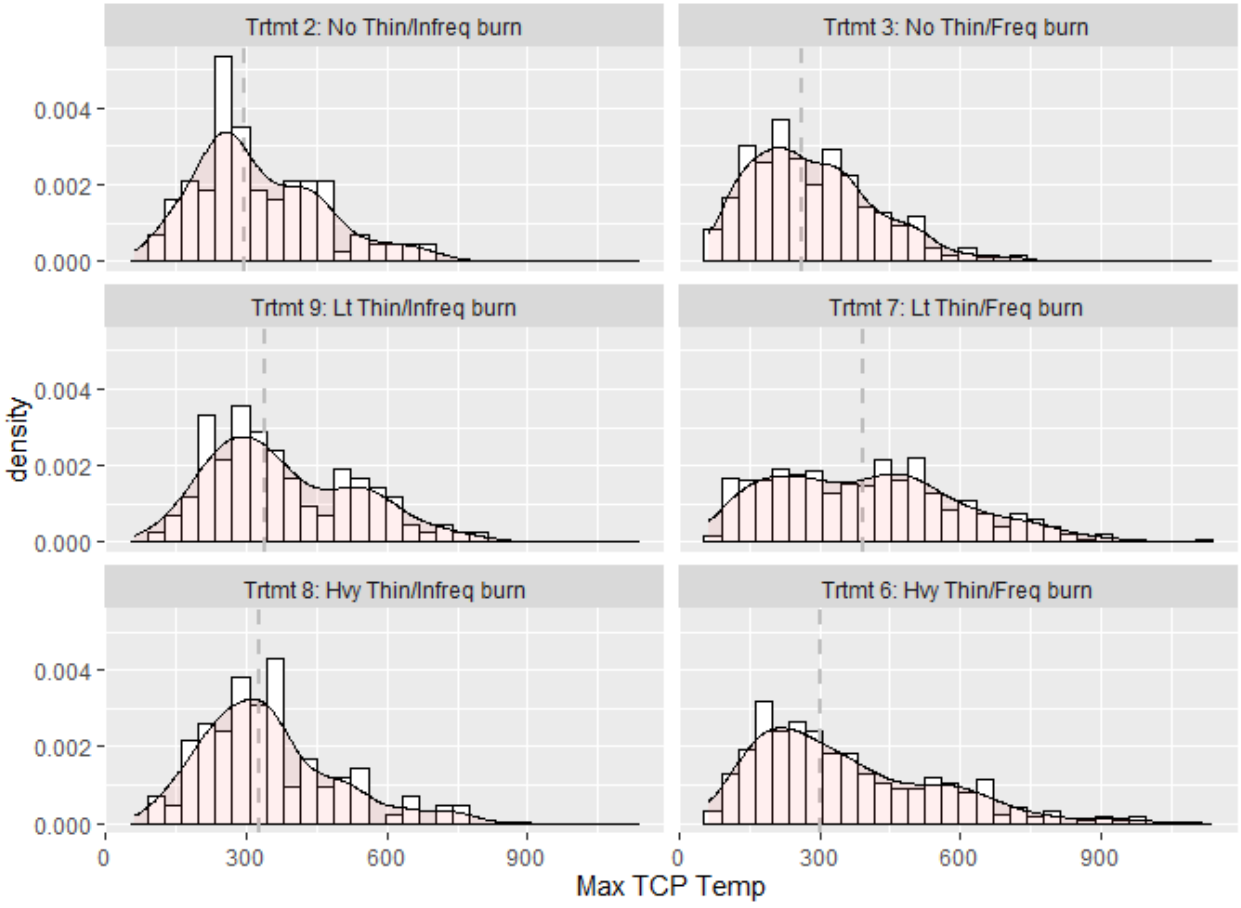

Figure 3—Maximum thermocouple probe (TCP) temperature density histograms by treatment for 38 dormant season prescribed fires on the William B. Bankhead National Forest, AL. Trtmt: treatment. Infreq: one burn every 9 years. Freq: one burn every 3 years. Lt Thin: thinning to residual basal area (BA) of 75 square feet per acre. Hvy Thin: thinning to residual BA of 50 square feet per acre. We used density, not frequency, histograms because the infrequent burn treatments (left column) contain roughly half as many observations as the frequent burn treatments (right column); plotting frequencies would cause the infrequent burn histograms to be squashed, making comparisons between the columns more difficult. Dashed vertical lines indicate the median for each treatment.

temperatures were the IVs for oaks and pines, and the seedling counts for pines and others. Other statistically significant covariates included time of day, day of year, averages of relative humidity and air temperature for the 24 hours preceding ignition, bark mass, duff mass, fuel temperature and fuel moisture, year, and three interaction terms. Table 2 organizes the statistically significant fixed effects by their relative importance in explaining variation in the maximum temperatures.

The model was built using the log-transformed outcome to address the right-skewed temperature distribution and the heteroscedasticity in the data; consequently, the estimated coefficients all describe effects on the log scale. To interpret covariate effects on the original scale of burn temperature, we assumed changes of one SD of each fixed effect term in turn (holding all other covariates constant), and then back-transformed the resulting impact from the log scale to the original scale. For example, a one SD increase in the IV for oak was associated with a 23 °F increase in predicted maximum burn temperature, holding all other covariates constant (table 3).

Table 1—Estimated coefficients and significance levels for fixed effects model used to predict maximum thermocouple probe temperature as a surrogate for fire intensity for 38 prescribed fires on the William B. Bankhead National Forest, AL

Variable category	Variable	Coefficient	*P*
Vegetation	Importance value, oak	0.0052	0.006
Vegetation	Importance value, pine	0.0038	0.004
Vegetation	Total seedlings, pine[1]	0.0115	0.045
Vegetation	Total seedlings, other species[1]	-0.0145	<0.001
Fuels	Bark weight (ln)	0.2495	0.024
Fuels	Duff weight (ln)	-0.1359	<0.001
Weather	Air temperature[2]	0.0656	<0.001
Weather	Relative humidity[2]	0.0371	0.010
Weather	Fuel temperature	-0.0163	0.048
Weather	Fuel moisture	-0.1429	0.034
Weather	Start time	-2.8137	0.002
Weather	Julian day of burn	-0.0355	<0.001
Weather	Year[3]	–	<0.001
Interaction	Air temperature x Relative humidity	-0.0009	0.001
Interaction	Fuel temperature x Fuel moisture	0.0022	0.017
Interaction	Time x Julian day of burn	0.0645	<0.001

ln = natural log .

[1] Seedling counts were divided by 1,000 to obtain more meaningful coefficients.

[2] Averages of air temperature (°F) and relative humidity for the 24 hours prior to burn start time.

[3] Year was treated as a factor variable; hence there was a different coefficient (not shown) for every value other than 2007, the reference year.

Table 2—Fixed effects model variables ranked by relative importance in explaining variation in maximum thermocouple probe temperature for 38 prescribed fires on the William B. Bankhead National Forest, AL

Category	Variable name
Weather	Julian day of burn
Weather	Air temperature, 24 hours pre-ignition
Weather	Time of day
Weather	Relative humidity, 24 hours pre-ignition
Vegetation	Importance value, pine
Vegetation	Importance value, oak
Vegetation	Total seedlings, other species
Fuels	Duff weight
Weather	Fuel moisture
Weather	Fuel temperature
Vegetation	Total seedlings, pine
Fuels	Bark weight

Relative importance was determined by *P*-values, model AIC changes, and considering if a variable was also included in interaction terms or not.

Table 3—Effect on maximum thermocouple probe temperature (original scale) of a one standard deviation change in four vegetation variables in the mixed effects model

	Importance value		Total seedlings	
Factor	Oak	Pine	Pine	Other species
Model coefficient	0.00525	0.00382	0.01150	-0.01447
1 standard deviation (SD)	13.8	19.7	3.1	4.2
1 SD change (log scale)	0.0724	0.0752	0.0359	-0.0609
Change[1]	23.5	24.5	11.5	-18.4

[1] Change in median maximum thermocouple probe temperature at original scale (°F).

DISCUSSION

A long-held view of fire behavior is that intensity is a product of weather, topography, and fuels (Rothermel and Deeming 1980, Varner and others 2015). Each of these three components are broad concepts comprising many individual elements; ascertaining the relative importance of the concepts to fire behavior has been a focus of much research (Bessie and Johnson 1995, Hély and others 2001, Wenk and others 2011). The main question driving this analysis was not which component had the highest relative importance, but rather, after controlling for weather and fuels, which vegetation variables, if any, had a statistically significant association with maximum TCP temperature. We view maximum TCP temperature as a useful index for fire intensity even if it does not fully describe the heat transfer effects that occur. Teasing out the influence of vegetation requires conditioning models on those other factors. Indeed, weather variables played the largest role in explaining maximum TCP temperatures in this model (table 2). We highlight that the averages of air temperature and relative humidity over the 24 hours prior to ignition were more strongly associated with the outcome than air temperature or relative humidity strictly from the burn start time. Many studies reporting weather conditions prevailing during prescribed burns present those conditions only at ignition time or averaged over the duration of the burn (e.g., Iverson and others 2004, Wenk and others 2011). In our model building process, we also tested 48-, 72-, 96-, and 120-hour prior averages; while relationships were found for several of these, the strongest association here and best improvement in overall model fit was achieved with the 1-day prior averages. These results suggest that in reporting on fire research, it may be most useful to present weather conditions for 24 or more hours preceding the burn.

With respect to vegetation variables, none of the midstory importance values were significant in this model. Dormant season fires primarily consume leaf litter and thus it is likely that the overstory size class simply provides so much more leaf litter that the effect on burn temperatures from the overstory overwhelms any effect of the midstory as related to species distribution. The overstory species groups that showed significant associations were oaks and pines, with higher IVs in each case implying higher maximum TCP temperatures. The IV for red maple was not significant; it is conjectured, however, that an increase in IV for red maple would be accompanied by a decrease in IV for oaks or pines or both, and hence would tend to lead to a decrease in maximum TCP temperatures. The IV for the other species group was omitted as a candidate variable (linear dependence among the independent variables would result if all four IVs for a size class were included together). For the understory, seedling totals for the other species group and for pines were statistically significant, with higher pine seedling counts associated with higher maximum TCP temperatures, and higher seedling counts for the other species group (i.e., nonoaks, pines, or red maples) associated with lower temperatures.

Of the six categories of dead fuels tested in this analysis (bark, fruit, leaves, duff, 1-hour, and 10-hour), only bark and duff were statistically significant. We have found that the category of fuel that is most consistently and completely consumed by these prescribed fires is leaves (Craycroft and Schweitzer, unpublished), which is somewhat different from the experience of another recent study involving prescribed fire and thinning in the South (Vander Yacht and others 2019). Nevertheless, in this analysis, the mass of leaves in the aggregate does not appear to have a significant effect on fire temperatures after controlling for other variables. Therefore, it may be of interest to quantify fuel loads not just by size and type, but also by species; this may also enable assessment of how similar combustion and flammability results in the field are compared to lab experiments (Kreye and others 2023, Varner and others 2015). This study, however, did not capture that aspect of fuel loads.

With fire being such a dynamic phenomenon and acknowledging that feedback loops influencing fire behavior are complex, it is interesting that this model suggests associations between maximum TCP temperature and several vegetation variables, even after conditioning on weather and fuels. Also, certain other covariates such as slope steepness and aspect, while significant in models where those were the only

independent variables, were not statistically significant after conditioning on the other covariates included here. Meanwhile, the random components of the model suggest that there is relatively high variation yet unexplained by the model. This is understandable given the complex nature of fire as well as of the general categories of independent variables (vegetation, fuels, weather) and the consequent difficulty of fully capturing these constructs with a relatively small number of individual variables.

Several issues have been raised regarding the utility of TCPs for capturing fire behavior. Experiments have demonstrated lags between actual heating and when higher temperatures are recorded (Iverson and others 2004), and between the removal of heat and when lower temperatures are recorded (McGranahan 2020). Furthermore, data measurements depend upon the physical properties of the materials making up particular probes (Bova and Dickinson 2005). The orientation in which TCPs are installed also affects fire behavior estimates (Coates and others 2018).

Nevertheless, TCPs offer several important practical benefits. Compared to temperature sensitive paints, TCPs enable a time series of temperature measurements, rather than a single maximum temperature estimate. The data are measured on a continuous scale, rather than comparatively wide-ranged discrete temperature intervals, as with paints. TCPs provide ambient temperature readings and so can be used to compute duration of heating above ambient. More technologically advanced radiometers can more accurately measure actual energy flux but are more expensive than TCPs. The utility of TCP data means that a better approach than forsaking such data is treating it more as an index rather than an exact expression of fire intensity.

Finally, comparisons across burns, or across locations during the same burn, may provide valid and insightful results when care is taken to standardize the calibration and installation of the devices. Utilizing the full temperature-time profile provided by these devices, rather than the single metric of maximum TCP temperature, is the object of the next phase of this study.

CONCLUSIONS

Restoring fire as a disturbance and process in mixed pine-hardwood forests is an important step towards creating desired ecosystem functions and structures. However, effectively applying prescribed fire requires understanding how multiple factors, such as fire behavior, forest characteristics, and fuel composition interrelate in feedback loops. This understanding can best be achieved via long-term studies that comprehensively examine all aspects of these loops.

The ongoing research study at the BNF is an example of such a study, valuable due to both its longevity and its broad scope of data collection. After implementing 38 landscape-scale prescribed fires over 12 years, representing four cycles of frequent burning and two cycles of infrequent burning, we were able to couple data across weather, fuels, and vegetation to discern variables driving maximum TCP temperature as a surrogate for fire intensity. Using a mixed effects model, we controlled for factors affecting fire intensity and found that maximum TCP temperature was related to the IV of oaks and pines, bark and duff fuel loads, and weather. Continuing research both in this and in similar studies will serve to further refine our understanding of the dynamic feedback loops involving fires and forests.

LITERATURE CITED

Bessie, W.; Johnson, E. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology. 76: 747–762. https://doi.org/10.2307/1939341.

Bova, A.S.; Dickinson, M.B. 2005. Linking surface-fire behavior, stem heating, and tissue necrosis. Canadian Journal of Forest Research. 35: 814–822. https://doi.org/10.1139/x05-004.

Brose, P.H.; Dey, D.C.; Phillips, R.J.; Waldrop, T.A. 2013. A meta-analysis of the fire-oak hypothesis: does prescribed burning promote oak reproduction in eastern North America? Forest Science. 59: 322–334. https://doi.org/10.5849/forsci.12-039.

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. https://www.fs.usda.gov/research/treesearch/28647.

Coates, T.A.; Chow, A.T.; Hagan, D.L. [and others]. 2018. Thermocouple probe orientation affects prescribed fire behavior estimation. Journal of Environmental Quality. 47: 170–176. https://doi.org/10.2134/jeq2017.02.0055.

Cottam, G.; Curtis, J.T. 1956. The use of distance measures in phytosociological sampling. Ecology. 37: 451–460. https://doi.org/10.2307/1930167.

Craycroft, J.; Schweitzer, C. 2023. Profiling overstory survival trends following varying thinning and burning disturbance regimes in a mixed pine-hardwood forest in the US South. Forest Ecology and Management. 531: 120796. 9 p. https://doi.org/10.1016/j.foreco.2023.120796.

Crow, T.R.; Buckley, D.S.; Nauertz, E.A.; Zasada, J.C. 2002. Effects of management on the composition and structure of northern hardwood forests in Upper Michigan. Forest Science. 48: 129–145. https://www.fs.usda.gov/research/treesearch/13262.

Hély, C.; Flannigan, M.; Bergeron, Y.; McRae, D. 2001. Role of vegetation and weather on fire behavior in the Canadian mixedwood boreal forest using two fire behavior prediction systems. Canadian Journal of Forest Research. 31: 430–441. https://doi.org/10.1139/x00-192.

Iverson, L.R.; Yaussy, D.A.; Rebbeck, J. [and others]. 2004. A comparison of thermocouples and temperature paints to monitor spatial and temporal characteristics of landscape-scale prescribed fires. International Journal of Wildland Fire. 13: 311–322. https://doi.org/10.1071/WF03063.

Kreye, J.K.; Kane, J.M.; Varner, J.M. 2023. Multivariate roles of litter traits on moisture and flammability of temperate northeastern North American tree species. Fire Ecology. 19: 1–12. https://doi.org/10.1186/s42408-023-00176-5.

McGranahan, D.A. 2020. An inconvenient truth about temperature–time data from thermocouples. Plant Ecology. 221: 1091–1104. https://doi.org/10.1007/s11258-020-01064-7.

R Core Team. 2021. R: A language and environment for statistical computing. Version 4.1.1. Vienna, Austria: R Foundation for Statistical Computing. https://www.R-project.org.

Rothermel, R.C.; Deeming, J.E. 1980. Measuring and interpreting fire behavior for correlation with fire effects. Gen. Tech. Rep. INT-93. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 3 p.

Schweitzer, C.J.; Dey, D.C.; Wang, Y. 2016. Hardwood-pine mixedwoods stand dynamics following thinning and prescribed burning. Fire Ecology. 12: 85–104. https://doi.org/10.4996/fireecology.1202085.

Schweitzer, C.J.; Dey, D.C.; Wang, Y. 2019. White oak (*Quercus alba*) response to thinning and prescribed fire in Northcentral Alabama mixed pine–hardwood forests. Forest Science. 65: 758–766. https://doi.org/10.1093/forsci/fxz031.

University of Utah. 2022. MesoWest. https://mesowest.utah.edu/cgi-bin/droman/download_api2.cgi?stn=BHFA1. [Date accessed: March 15, 2022].

U.S. Department of Agriculture Forest Service [USDA Forest Service]. 2004. Revised land and resource management plan: National Forests in Alabama. Manage. Bull. R8-MB 112A. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region. 330 p.

Vander Yacht, A.L.; Keyser, P.D.; Kwit, C. [and others]. 2019. Fuel dynamics during oak woodland and savanna restoration in the Mid-South USA. International Journal of Wildland Fire. 28: 70. https://doi.org/10.1071/WF18048.

Varner, J.M.; Kane, J.M.; Kreye, J.K.; Engber, E. 2015. The flammability of forest and woodland litter: a synthesis. Current Forestry Reports. 1: 91–99. https://doi.org/10.1007/s40725-015-0012-x.

Wenk, E.S.; Wang, G.G.; Walker, J.L. 2011. Within-stand variation in understorey vegetation affects fire behaviour in longleaf pine xeric sandhills. International Journal of Wildland Fire. 20: 866–875. https://doi.org/10.1071/WF10087.

Zuur, A.F.; Ieno, E.N.; Walker, N.J. [and others]. 2009. Mixed effects models and extensions in ecology with R. New York: Springer. 574 p. https://doi.org/10.1007/978-0-387-87458-6.

PRESCRIBED FIRE BARRIERS AND SOLUTIONS FOR MANAGING SUSTAINABLE FORESTS: FORUM DISCUSSION

Callie Schweitzer, Dan Dey, and Andrea Brandon

EXTENDED ABSTRACT

A virtual forum for land managers to provide feedback on barriers to, and opportunities for, the increased use of prescribed fire in the United States was held in June 2021. The session included a series of case studies illustrating barriers and opportunities in different ecosystems as well as key findings from recent research on prescribed fire use. In Eastern U.S. forests, where fire has long been excluded from multiple fire-adapted forest types including oak-pine (*Quercus* spp.-*Pinus* spp.) forests, woodlands, and savannas, ongoing research demonstrates that prescribed fire can be implemented while retaining timber value of commercially important hardwoods (Ryan and others 2013). In the Western United States, where prescribed fire is a critical restoration tool in fire-adapted forests, shrublands, and grasslands, the "modeling-monitoring-messaging" framework has been used to increase public acceptance of smoke impacts from prescribed fire, particularly in contrast to wildfire smoke impacts (Stotts and others 2018). In grasslands of the southern Great Plains, where fire exclusion is associated with encroachment of cedars (*Juniperus* spp.) and woody shrubs, ongoing research is demonstrating that the expansion of prescribed burn windows into the growing season can enhance desirable fire effects including improved forage productivity and increased cedar mortality (Weir and Scasta 2017). Currently, success stories in prescribed fire management often occur at local levels and are seemingly random, dependent on local personalities and relationships. The fire message can be confusing, with language targeting "fighting" fire, and the public is not well educated on the ecological role of fire in forested ecosystems.

For the forum, nearly 100 individuals from more than 20 diverse agencies and organizations participated in a day-long event. Participants included individuals representing Federal and State agencies, local units of government, Tribal governments, nongovernmental organizations, academia, other research institutions, and industry. Fields of forestry, fire ecology, wildlife, biology, economics, and others were represented. In addition, approximately 47 written responses were collected. Participants

Callie Schweitzer, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Huntsville, AL 35801

Dan Dey, Assistant Station Director for Research, U.S. Department of Agriculture, Forest Service, Northern Research Station, Columbia, MO 65211

Andrea Brandon, Science Delivery Specialist, U.S. Department of Agriculture, Forest Service, Northern Research Station, Parsons, WV 26241

were asked to respond to questions about four key topics. Questions addressed 1) delineating the barriers to planning, implementation, and monitoring the use of prescribed fire at the desired pace, scale, and seasonality; 2) proposed solutions to the identified barriers; 3) additional information needed to implement proposed solutions; and 4) new barriers likely to arise to proposed solutions.

During the feedback portion of the session, managers identified prevalent barriers to prescribed fire use. These included narrow burn windows, limited resources and capacity (both for planning and implementation), and the complexity associated with large-scale burning. Much discussion centered on a "culture of burning" that is often lacking, in both leadership and management, along with a limited understanding of fire among nonfire staff; however, it was noted that southern forest managers understood the application of fire and were talented at using fire as a tool to meet management goals. A general lack of incentives and limited capacity to work across ownership boundaries was also identified as a common barrier, along with the ubiquitous lack of funding and other resources. Less commonly identified barriers varied among regions and land ownership and included air quality restrictions, lack of confidence in consistent leadership support, social license, suppression culture, and conflicts with timber and other resources.

Opportunities to overcome these barriers included resource sharing and leveraging multistakeholder partnerships, the potential to create larger scale prescribed burning organizations, and building trusted relationships among practitioners and the public. Collaborations among many partners in prescribed fire is crucial in all aspects of planning, implementation, monitoring, and science. Incorporating different voices, ideas, and perspectives are important and contribute to better plans and decision making and increases capacity by sharing of resources. Tools used to promote and form collaborations may include all types of agreements (cost-share, participating, interagency, cooperative) and Federal authorities such as the Wyden Agreement and Good Neighbor Authority. Existing entities such as fire councils and associations and fire-learning networks help facilitate partnerships in science delivery collaborations. Learn-to-burn workshops, organized by a conglomerate of extension agencies, State forestry agencies, local resource and conservation councils, and others, bring landowners together as active participants in learning how to plan and implement prescribed fire on their land. Other noted collaborations focused on using prescribed fire as a resource tool include those with professional societies such as the Society of American Foresters and The Wildlife Society, and nongovernmental organizations including The Nature Conservancy, National Wild Turkey Federation, The Rocky Mountain Elk Foundation, American Forest Foundation, and Quail Forever.

To further define and implement proposed solutions, participants highlighted the need for research to incorporate social, physical, and ecological approaches. Participating managers emphasized the need for additional information to implement proposed solutions effectively. They highlighted the importance of tailored research and products specific to

prescribed fire, including improved fire behavior and effects models. They also expressed the need for information on climate change impacts, fire history across ecosystems, fire effects on various resources, and trends in fire frequency. To better educate the public on the benefits of prescribed fire, programs need to investigate how people prefer to be informed and design training and education that target those preferences. Increased disciplinary integration in education, training, and certification programs for fire, silviculture, wildlife, hydrology, and other resources would encourage greater acceptance and use of prescribed fire.

Figure 1—A prescribed fire on the William B. Bankhead National Forest, AL, conducted during the dormant season, has short flame lengths and creeps along in the understory of the forest, consuming mostly leaf litter. (USDA Forest Service photo by Callie Schweitzer)

Ecological modeling that incorporates scenarios and outcomes for various disciplines, including timber, regeneration, wildlife, recreation, water, and soils, that are germane to specific forested systems have tremendous utility and are often lacking, requiring adaptation from dissimilar systems which results in divergent outcomes. Predicting where prescribed fire would result in the most desired outcomes and what kind of tradeoffs in terms of resource benefits are to be expected is ongoing research with much anticipated outcomes.

Managers expressed a need for landscape models to help identify priority areas to be treated by prescribed fire and to target deployment of resources. Identifying the best use of prescribed fire on the landscape, in the context of maintenance burning to reduce wildfire risk and maintaining this fire return interval, or moving to untreated areas, was a key issue of concern. Although information regarding the implementation of, and science behind, prescribed fire, abounds, there remains a lack of a repository for these resources. Additionally, as prescribed fire science and use increases, incorporating new research and outcomes into planning will continue to be a challenge. Communicating anticipated fire, ecological, and smoke impacts from a range of fire scenarios will remain a struggle, and resource availability may limit broad-scale applications.

Overall, the forum provided valuable insights into the barriers and opportunities for increased use of prescribed fire in the United States. Land managers expressed the need for collaborative efforts, improved resources, and increased awareness to effectively overcome these barriers and implement prescribed fire as a restoration tool.

LITERATURE CITED

Ryan, K.C.; Knapp, E.E.; Varner, J.M. 2013. Prescribed fire in North American forests and woodlands: history, current practice, and challenges. Frontiers in Ecology and the Environment. 11: e15–e24. https://doi.org/10.1890/120329.

Stotts, C.; Lahm, P.; Standish, C. 2018. Resources for predicting and mitigating smoke impacts of wildland fires. ERI Working Paper No. 40. Flagstaff, AZ: Ecological Restoration Institute, Northern Arizona University. 11 p.

Weir, J.R.; Scasta, J.D. 2017. Vegetation responses to season of fire in tallgrass prairie: a 13-year case study. Fire Ecology. 13: 137–142. https://doi.org/10.4996/fireecology.130290241.

LINKING FIRE BEHAVIOR TO EFFECTS: USE OF PYROMETERS AND THERMOCOUPLES

Mohammad M. Bataineh, Brant Portner, Matthew Pelkki, and Robert Ficklin

ABSTRACT

Linking fire behavior and heating to plant responses is needed to generalize fire effects. Temperature recording devices, such as thermocouples and pyrometers, have been deployed in several studies. Use of these relatively unsophisticated devices is likely to continue as spatiotemporally explicit methods for quantifying temperature variability become more accessible. We compared pyrometer and thermocouple device temperatures placed at ground level and 30 cm aboveground to determine pyrometer suitability as a rapid deployment device for capturing variability within and among four burns conducted in a frequently (every 3 years) burned upland oak-hickory (*Quercus-Carya*) forest of the Ozark Mountains, AR. Pyrometer temperatures better reflected thermocouple temperatures for burns with higher fire line intensities and heat per unit area. Placement of pyrometers 30 cm aboveground appeared to be more advantageous in recording fire temperatures in upland oak-hickory forests such as those in this study. However, metal (aluminum tag) pyrometer deployment in prescribed burns with an anticipated very low to low fire behavior may not be suitable.

INTRODUCTION

Despite a plethora of fire effect studies, generalizations of wildland fire effects on fuel and vegetation remain difficult to establish (Butler and Dickinson 2010, Pyne and others 1996). Inconsistencies among study methodologies continue to present challenges in providing a comprehensive understanding of fire effects (Brose and others 2013, Pyne and others 1996). Perhaps the most significant of limitations is our inability to directly link fire behavior and heat transfer to vegetation and fuel responses. Linking of fire behavior and heating patterns to individual plant and species responses at fine spatial scales is needed to generalize results from fire effect studies by addressing the inherent variability in fire behavior within and among studies (Bataineh and others 2022, Brose and others 2013, Butler and Dickinson 2010, Rothermel and Deeming 1980). This linkage is the foundation in first order fire effects models that use a mix of species and fire-related attributes to predict immediate impacts (Lutes and others 2012). Plant mortality, tissue injury, and postfire resprouting and regeneration are all localized, neighborhood-level processes that are influenced by variability in fuel accumulation and consumption along with heat transfer at fine spatial scales.

Operational use of prescribed burning to reduce hazardous fuel loads, restore disturbance regime and woodland structures, and promote oak reproduction requires a better understanding of fire effects at relevant scales, if meaningful burning guidelines are to be developed. Traditionally, fire behavior is described using standardized metrics estimated or measured at the flaming front, such as rate of spread and flame length (Byram 1959, Rothermel and Deeming 1980). Fireline intensity, which cannot be directly measured, is defined as energy release rate from a linear section of the fire front and is estimated by multiplying rate of spread by heat yield and

Mohammad M. Bataineh, Associate Professor, College of Forestry, Agriculture, and Natural Resources, University of Arkansas at Monticello, Monticello, AR 71656

Brant Portner, Graduate Student, College of Forestry, Agriculture, and Natural Resources, University of Arkansas at Monticello, Monticello, AR 71656

Matthew Pelkki, Professor, College of Forestry, Agriculture, and Natural Resources, University of Arkansas at Monticello, Monticello, AR 71656

Robert Ficklin, Professor, College of Forestry, Agriculture, and Natural Resources, University of Arkansas at Monticello, Monticello, AR 71656.

available and consumed fuel (Byram 1959). Flame length, distance between average flame tip and base at midpoint of the active flaming front, correlates with fireline intensity and may be used in its estimation (Alexander and Cruz 2012, Rothermel and Deeming 1980). Although fireline intensity, rate of spread, and flame length are useful metrics, their operational use in reporting fire behavior is limited. Even when these metrics are reported, the values represent limited locations along the active flaming front of a burn unit. These metrics are also only suited for surface fires in uniform, porous fuelbeds and cannot be used to describe crowning or fire behavior in clumped fuelbeds (Rothermel and Deeming 1980).

Fire temperature has been commonly used to provide a linkage between fire behavior and effects, albeit not without limitations (Breuer 1965, Fons 1946, Van Wagner and Methven 1978, Walker and Stocks 1968). Use of temperature to characterize fire behavior has been criticized as a coarse resolution metric requiring fast-reacting sensors that integrate data across the whole energy field of view (Van Wagner and Methven 1978). Fire temperature alone cannot be mechanistically linked to effects, nor is it an adequate descriptor of fire intensity (McGranahan 2020, Smith and others 2016). However, temperature is needed in the calculations of fire's radiative heat flux density and convective heat transfer (Bova and Dickinson 2008). Fire temperature recording devices such as metal or ceramic pyrometers with temperature-sensitive paint, calorimeters, and thermocouples have been deployed or tested in several studies (Bataineh and others 2022, Iverson and others 2004, Kennard and others 2005). Although temperature from these devices reflects device temperature (Bova and Dickinson 2008, Kennard and others 2005), use of these relatively unsophisticated devices is likely to continue in the near future as spatiotemporally explicit methods for quantifying temperature variability, such as thermal imagers, become more accessible and affordable (Dickinson and others 2016). It is also unrealistic to expect abandonment of relatively simple to use, low labor, and low-cost devices when accurate direct measurements or predictions of fire heat flux remain out of reach (Bova and Dickinson 2008, Kremens and others 2010). Furthermore, device temperature correlates highly with the fire's heat flux, reflecting an integration of fire-induced heating magnitude and duration, and allowing for accurate estimation of fuel consumption and flame residence times (Bova and Dickinson 2008, Kennard and others 2005). Pyrometer and thermocouple deployment in prescribed burns is far better than mere qualitative characterization of fires (e.g., season or severity-of-burn comparisons). Thermocouple deployment with data loggers allows for time-temperature integration and consequently the calculation of various fire behavior metrics reflecting the magnitude and duration of heating effects such as residence time above a specific threshold or the arithmetic product of degree times seconds (Bova and Dickinson 2008, Kennard and others 2005, McGranahan 2020). While acknowledging the limitations of fire temperature measuring devices and use of temperature in characterizing fire behavior, here we compare pyrometer and thermocouple device temperatures to determine the suitability of pyrometers as rapid deployment devices for capturing variability within and among four prescribed burns conducted in a frequently burned (every 3 years) upland oak-hickory forest of the Ozark Mountains, AR. Temperature devices were compared for ground level and 30 cm aboveground placement of pyrometers and thermocouples.

MATERIALS AND METHODS

Study Area

The study was located within the University of Arkansas Savoy Experimental Forest in Washington County, AR (Bataineh and others 2022). Located within the Ozark Highlands ecoregion, the area is an upland oak-hickory forest with a gently rolling topography. Elevation range is 300–400 m above sea level. Annually, the region receives 117 cm of rainfall and 16 cm of snowfall. Annual frost-free period is 180–194 days. Mean annual temperature is 14 °C; mean minimum and mean maximum temperatures are 8 °C and 20 °C, respectively. Soils consist mostly of the Clarksville-Nixa-Baxter and Captina-Nixa-Pickwick associations which are well-drained, level to gently sloping soils formed mainly

under hardwoods (Harper and others 1969). Predominant soil series of the area are the Nixa (loamy-skeletal, siliceous, active, mesic Glossic Fragiudults) with 3 to 8 percent slopes and the less prevalent Clarksville (loamy-skeletal, siliceous, semiactive, mesic Typic Paleudults) with 12 to 60 percent slopes (Harper and others 1969). The area of 142 ha was high graded in 1996–1998 removing all merchantable trees >31 cm in diameter at breast height (Bataineh and others 2022). The preharvest stand originated following exploitative cuts in the early 1900s. Postharvest residual overstory ranged from 324 to 348 trees per ha and was composed of low commercial quality white (*Quercus alba*), black *(Q. velutina*), and post (*Q. stellata*) oaks as well as blackgum (*Nyssa sylvatica*), hickory (*Carya* spp.), and low proportions of other red oaks and red maple (*Acer rubrum*). Stand-level basal area was around 14 m^2ha^{-1} with a quadratic mean diameter of 23.4 cm, representing understocked conditions, below the B-line, based on Gingrich's stocking charts for upland hardwoods (Gingrich 1967). Site index was around 18 m for white oak using a base age of 50 years. Advance reproduction of red maple, blackgum, black cherry (*Prunus serotina*), and flowering dogwood (*Cornus florida*) represented 50–60 percent of total number of stems, whereas oaks represented roughly 25 percent (Bataineh and Pelkki 2018).

Study Design and Measurements

As part of a long-term study initiated in 2004, this study used repeated burning applications to compare device temperatures. The long-term study aimed to evaluate prescribed burning, herbicide application, and combinations of burning and herbicide for improving the composition and structure of high-graded oak-hickory stands (Bataineh and Pelkki 2018, Morris 2007). With initial burning occurring in 2005, repeat burns were conducted in 2008, 2011–2012, 2014–2015, and 2017. Repeat burns scheduled as part of the long-term study in 2017, and four treatment units, each 0.8 ha, were used in this study.

Within each burn unit, 10 to 15 wildland fuel cells (each 9.3 m^2) with uniform fuel abundance and distribution were established (Hiers and others 2009). Fuel cells were selected randomly from a sampling frame of 720 grid cells systematically dividing each burn unit, excluding a 5-m buffer around each unit. Over a 3-day period in April 2017, burns were conducted within each unit. Ignition pattern for all but one unit was circular (ring) ignition (burn units 6, 13, and 17) without spot fires in the center of the unit. Due to higher wind speeds and air temperature, burn unit 19 ignition pattern was strip heading with an average strip width of 20–25 m. Fire behavior parameters were recorded during each burn, and weather conditions were recorded prior to and during burning (table 1). Air temperature, relative humidity, eye-level wind speed, and wind direction from origin were recorded using a handheld Kestrel® 3000 weather meter. Ranging wooden poles coated with fire resistant paint with alternate gradations at 30 cm intervals were used as sighting reference within each burn unit. Poles were spaced in a 10- by 10-m grid across each burn unit. Timing of flame front passage from one pole to another was recorded and rate of spread calculated. Flame height and average flame angle were estimated using ranging pole gradations as reference. Flame length was calculated from flame height and average angle. Heading, backing, and flanking fires flame length and rate of spread were estimated separately, and recorded in the field as an average across each burn unit. Fireline intensity was calculated from flame length estimates for each burn unit (Byram 1959, Rothermel and Deeming 1980). Heat per unit area was calculated from fireline intensity and rate of spread estimates (Byram 1959, Rothermel and Deeming 1980).

As part of an accompanying study examining first-order fire effects on oak and maple reproduction (Bataineh and others 2022), two relatively thick (4.8 mm diameter) HOBO® (HOBO®, Bourne, MA) type-K thermocouple probes and two pyrometers, Tempilaq® (Tempilaq®, Holdahl, Minneapolis, MN) painted aluminum tags, were used within each wildland fuel cell. Thermocouple probes and pyrometers were placed in proximity prior to ignition at ground level and 30 cm aboveground. Ground-level probes were placed horizontally directly under the litter layer. Aboveground probes were placed vertically, perpendicular to ground surface. Thermocouple data loggers were buried outside each fuel cell to reduce disturbance and

Table 1—Weather, mean fuel moisture, mean soil moisture, and fire behavior characteristics for four dormant season prescribed burns conducted as part of a frequent burning regime at 3-year intervals in upland oak-hickory forests of the Arkansas Ozarks

	Burn unit			
Variable	6	13	17	19
Burn date (M/D/Y)	4/7/2017	4/7/2017	4/8/2017	4/9/2017
Start time (H:M)	12:46	18:40	13:53	14:21
Days since last rain	4	5	6	6
Soil moisture (percent)	21 (4)	19 (2)	18 (2)	19 (2)
Litter moisture (percent)	46 (14)	17 (14)	9 (9)	24 (13)
Duff moisture (percent)	71 (18)	36 (20)	37 (31)	33 (14)
Air temperature (°C)	20	21	28	27
Relative humidity (percent)	30	37	36	50
Wind speed (kph)	2.1	1.9	1.6	5.6
Wind direction from origin	E	NW	SE	SW
Flame lengths (m)	0.6	0.3	0.9	1.0
Rate of spread (m/min)	1.2	0.6	2.4	1.8
Fireline intensity (kW/m)	88	20	217	252
Heat per unit area (kJ/m^2)	4,402	1,956	5,430	8,414

Soil, litter, and duff moisture are means followed by standard deviation in parentheses. Relative humidity was measured around time of ignition.
Source: Bataineh and others (2022).

were set to record at 1-second intervals starting 30 to 60 minutes before ignition. Pyrometers had 22 strips of temperature-indicating paint ranging from 79 °C to 427 °C, in increments of 14 °C for 79 °C to 316 °C and 28 °C increments for 343 °C to 427 °C. Pre and postburn fuel loadings were estimated using two transects (4.3 m each) following the procedure outlined in Brown (1974). Fuel loading transects intersected at the center of each fuel cell. Dead and downed woody debris was tallied in four timelag-size classes, namely: woody 1-hour (0.1–0.6 cm diameter at intersection), 10-hour (0.6–2.5 cm), 100-hour (2.5–7.6 cm), and 1,000-hour (>7.6 cm). Woody 1-hour and 10-hour classes were tallied along the first 0.9 m of each transect, whereas the 100-hour class was tallied along the first 1.8 m of each transect. Larger fuel sources in the 1,000-hour class were tallied along the entire length of each transect and were classified as sound or rotten. Litter (including freshly fallen leaves, dead herbs, dead shrubs, and acorns) and duff (organic soil horizons including fermentation and humus layers) were collected separately from 30 cm^2 quadrats, placed near but outside each wildland fuel cell and reflected fuel conditions of these cells. Litter and duff samples were weighed fresh before burning, oven dried to a constant weight at 60 °C for 3 days, and reweighed dry to estimate fuel moisture.

Analytical Approach

To compare device temperatures between pyrometers and thermocouples, peak or instantaneous maximum temperature from thermocouples was correlated to maximum temperature reached for pyrometers as indicated by the melting of paint strips. Simple linear regression was also used to attain proportion of variability in thermocouple temperature that is explained by pyrometers. Fuel consumption was calculated by subtracting post from preburn fuel loads for each fuel cell. Fuel loads for downed wood were calculated using the formulas presented in Brown (1974). Downed wood, litter, and duff loads were converted to unit area estimates. Subsequently, fuel consumption was calculated for the four downed

wood timelag-size classes, litter, and duff. Fuel consumption estimates by component were correlated to temperatures of each thermocouple and pyrometer. Preburn litter and duff moisture content estimates were calculated as the difference between fresh and dry weight in proportion to oven dry weight. All analyses were performed using the R base package version 3.0.2 (R Development Core Team 2013).

RESULTS AND DISCUSSION

Four burns were conducted in April 2017 that varied in fireline intensities, flame length, and rate of spread (table 1). Burns for units 17 and 19 had higher fireline intensities and heat per unit area compared to fires in units 6 and 13. Rates of spread and flame lengths were also higher for units 17 and 19 compared to 6 and 13. Fire behavior ranged from very low to low for units 6 and 13 and low-moderate for units 17 and 19 based on flame lengths and rate of spread (Scott and Burgan 2005). Variation in fire behavior was a reflection of differences in weather and fuel moisture conditions among the four burns (table 1), rather than fuel loads (Bataineh and others 2022). Unit 17 burn was conducted during a period of relatively higher air temperature, lower relative humidity, lower southerly wind speeds, and lower litter moisture relative to the remaining burns. Compared to unit 17, similar air temperatures but higher southerly wind speeds, higher relative humidity, and higher litter moisture content resulted in lower rate of spread but higher fireline intensity, heat per unit area, and flame length for unit 19. Ignition pattern in unit 19 with strip heading fire of 20–25 m in width had a substantial impact on fire behavior that modulated rate of spread. Duff and litter moisture contents were highest for unit 6, which had the second lowest fireline intensity and flame length, with moisture levels at 71 ± 18 (standard deviation) and 46 ± 14 percent, respectively (table 1). Moisture content for duff and litter was obtained from samples collected before burning and was likely higher than that at ignition time, early afternoon for unit 6.

Device temperature correlation depended on burn characteristics and device placement at ground level or 30 cm aboveground (figs. 1 and 2). For 30 cm aboveground placement, burns with higher fireline intensities and heat per unit area (units 17 and 19) had high and statistically significant correlation coefficients compared to those with lower fireline intensities and heat per unit area (fig. 1). Regardless of burn, pyrometers at 30 cm aboveground explained 54 percent of the variability (P <0.01) in thermocouple instantaneous maximum temperature similar to results elsewhere (Kennard and others 2005). Poor performance of temperature-indicating paint (Kennard and others 2005) was also noted in this study. Low fireline intensity burns (units 6 and 13) had fewer pyrometers with melted paints that could be reliably recorded. Poor melting required that at least two observers score each pyrometer (Wally and others 2006), with only distinct melting recorded. For ground-level placement, device temperature correlation coefficients were low and statistically nonsignificant, showing greater discord between thermocouple instantaneous maximum temperature and maximum temperature reached for pyrometers (fig. 2). Regardless of burn, no significant linear relationship was found between thermocouples and pyrometers at ground level (P = 0.87). Placement of pyrometers 30 cm aboveground appears to be more advantageous in recording fire temperatures in upland oak-hickory forest of this study and elsewhere (Kennard and others 2005). Metal (aluminum tag) pyrometer deployment in prescribed burns with an anticipated very low to low fire behavior (Scott and Burgan 2005) may not be suitable for recording fire temperature and fire characterization, however.

Device temperatures did not correlate strongly with fuel components consumption (table 2). The general lack of linear relationships between device maximum temperatures and fuel consumption highlights the limitation of using temperature as a descriptor of fire intensity (McGranahan 2020, Smith and others 2016) and indicates that estimates of fuel consumption based on device maximum temperatures alone may not be adequate. A larger margin of error is expected when thermocouple instantaneous maximum temperature is used, rather than time-temperature integration, to predict fuel consumption (Bova and Dickinson 2008). Fuel consumption for the 1-hour and 10-hour components, however, showed statistically

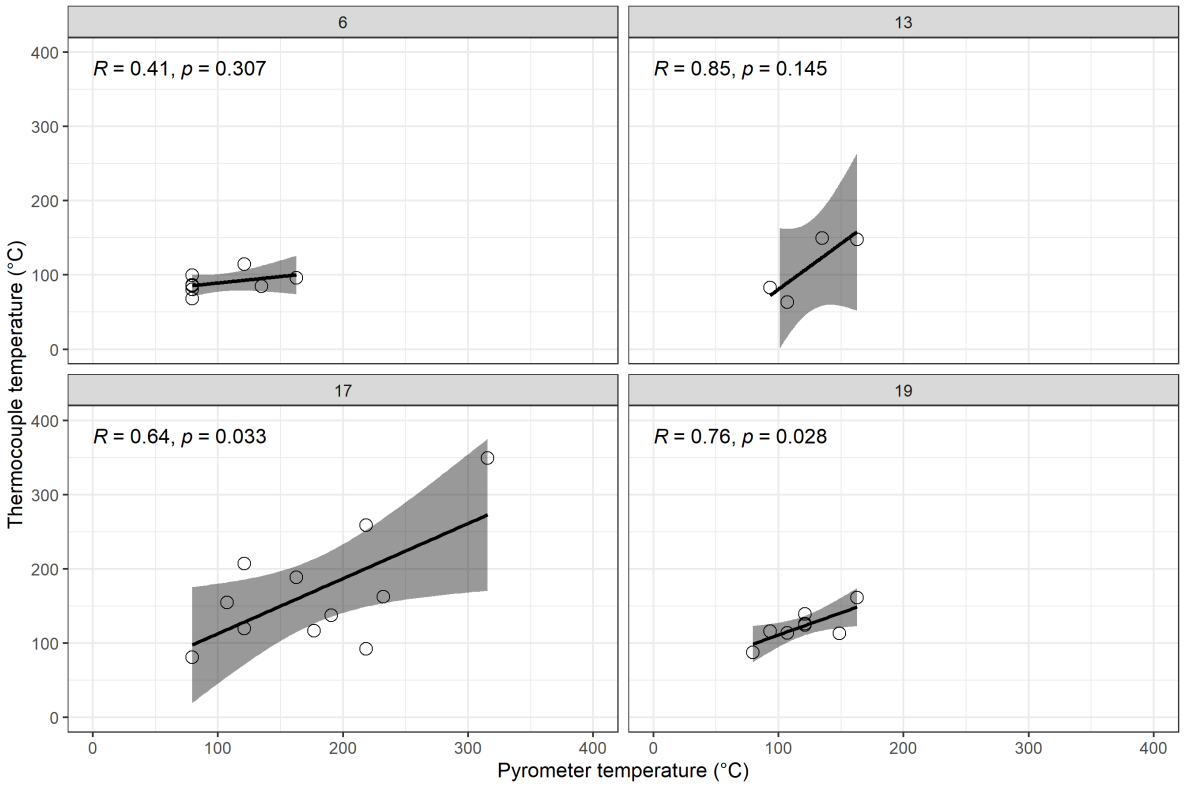

Figure 1—Scatter plots, with Pearson correlation coefficients and *P*-values, depicting the relationship between device temperatures, thermocouples and metal pyrometers, placed 30 cm aboveground for each of four burns conducted as part of a frequent burning regime (every 3 years) in upland oak-hickory forest of the Ozark Mountains, AR.

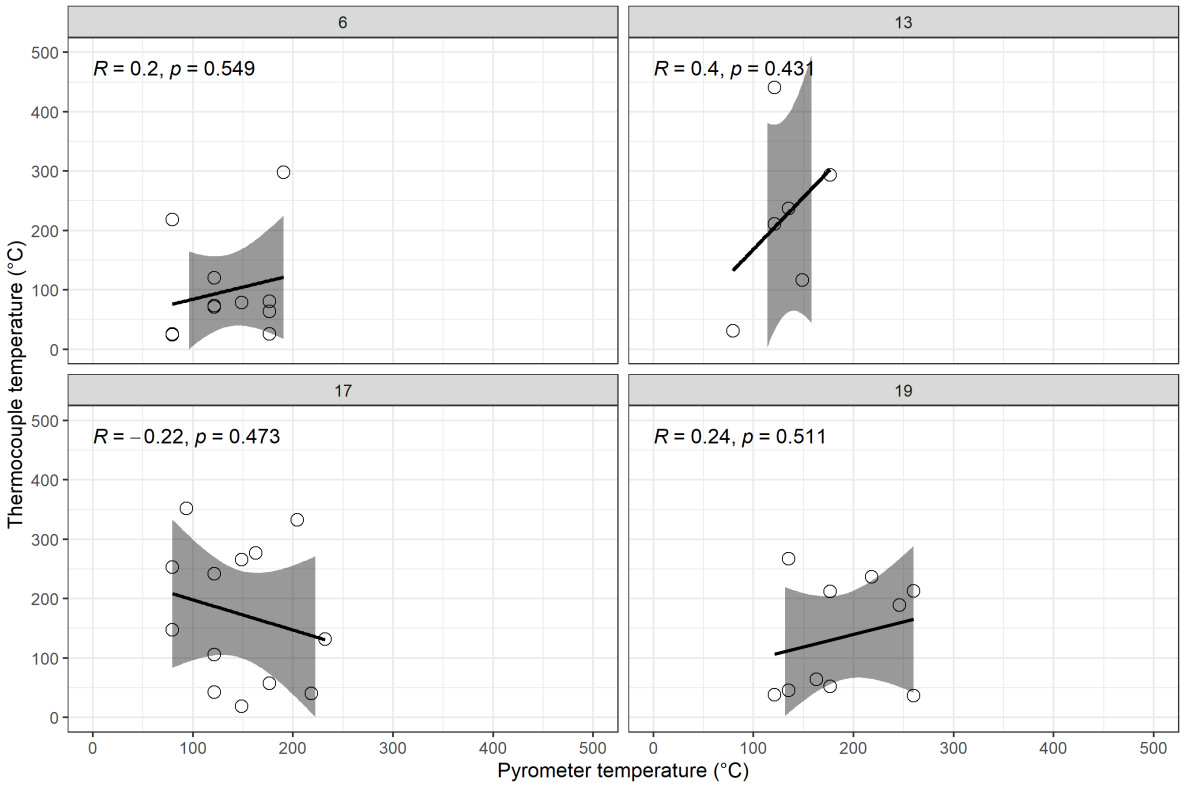

Figure 2—Scatter plots, with Pearson correlation coefficients and *P*-values, depicting the relationship between device temperatures, thermocouples and metal pyrometers, placed at ground level for each of four burns conducted as part of a frequent burning regime (every 3 years) in upland oak-hickory forest of the Ozark Mountains, AR.

Table 2—Pearson correlation coefficients (*P*-values in parenthesis) for temperatures recorded by pyrometers and thermocouples in relation to fuel components consumption during four prescribed burns conducted in a frequently (every 3 years) burned upland oak-hickory forest of the Ozark Mountains, AR

Temperature	Distance from ground	Burn unit	Litter	Duff	1-hour fuel	10-hour fuel	100-hour fuel
Thermocouple (°C)	**30 cm aboveground**	6	0.19 (0.51)	-0.24 (0.42)	0.05 (0.88)	-0.03 (0.92)	-0.11 (0.67)
		13	0.27 (0.34)	-0.14 (0.63)	0.04 (0.89)	-0.10 (0.74)	-0.32 (0.23)
		17	0.24 (0.42)	0.29 (0.31)	-0.10 (0.72)	-0.15 (0.62)	-0.06 (0.83)
		19	-0.12 (0.75)	-0.10 (0.79)	-0.45 (0.19)	***-0.83 (<0.01)***	-0.19 (0.61)
	At ground level	6	0.02 (0.95)	0.10 (0.73)	-0.09 (0.76)	-0.37 (0.19)	-0.06 (0.84)
		13	0.17 (0.56)	0.09 (0.78)	-0.30 (0.30)	-0.25 (0.39)	0.29 (0.32)
		17	-0.12 (0.67)	0.39 (0.15)	-0.21 (0.45)	0.16 (0.57)	-0.15 (0.59)
		19	-0.49 (0.15)	0.01 (0.99)	-0.38 (0.28)	-0.30 (0.40)	0.29 (0.42)
Pyrometer (°C)	**30 cm aboveground**	6	-0.06 (0.90)	-0.45 (0.26)	-0.19 (0.65)	-0.45 (0.27)	-0.30 (0.48)
		13	0.64 (0.36)	-0.71 (0.29)	***0.98 (0.02)***	0.23 (0.78)	-0.38 (0.62)
		17	-0.15 (0.66)	-0.19 (0.57)	-0.41 (0.21)	***-0.67 (0.02)***	-0.23 (0.49)
		19	0.25 (0.56)	0.51 (0.20)	-0.56 (0.15)	***-0.86 (<0.01)***	0.03 (0.95)
	At ground level	6	0.25 (0.46)	0.10 (0.77)	0.36 (0.28)	***-0.71 (0.02)***	0.24 (0.48)
		13	-0.21 (0.70)	-0.30 (0.57)	0.36 (0.48)	0.26 (0.62)	0.78 (0.07)
		17	0.26 (0.39)	0.13 (0.67)	0.34 (0.25)	-0.38 (0.20)	0.15 (0.63)
		19	0.39 (0.26)	-0.06 (0.86)	***-0.79 (<0.01)***	-0.62 (0.06)	-0.54 (0.10)

Statistically significant values are in bold italics.

significant negative relationships with device temperatures associated with burns of low-moderate fire behavior, namely units 17 and 19 (Scott and Burgan 2005).

CONCLUSIONS

Fire temperatures from thermocouples and pyrometers placed at ground level and 30 cm aboveground were compared to determine the suitability of pyrometers in characterizing fires, and thus providing an empirical linkage between fire behavior and effects. Metal (aluminum tag) pyrometers placed 30 cm aboveground correlated better and explained about half of the variability in thermocouple instantaneous maximum temperature compared to placement of pyrometers at ground level. Pyrometer temperatures better reflected thermocouple temperatures for burns with higher fireline intensities and heat per unit area, which may have reflected better performance of paint at higher temperatures or easier scoring of pyrometers by observers. Placement of pyrometers 30 cm aboveground appears to be more advantageous in recording fire temperatures in upland oak-hickory forests like those in this study. However, using metal (aluminum tag) pyrometers in prescribed burns with an anticipated very low to low fire behavior may not be suitable.

ACKNOWLEDGMENTS

Thanks to the Arkansas Forestry Division, especially Dennis Spear and his prescribed burn crew for conducting and supervising the burns. This project was assisted by Dalton Weatherly, Billy Sanders, Phillip Duncan, and Alan Jones of the University of Arkansas at Monticello. Special thanks to Lynne Thompson, John Dennis, and Tyler Corbin for reviewing an earlier version of this paper.

LITERATURE CITED

Alexander, M.E.; Cruz, M.G. 2012. Graphical aids for visualizing Byram's fireline intensity in relation to flame length and crown scorch height. Forestry Chronicle. 88(2):185-190. https://doi.org/10.5558/tfc2012-035.

Bataineh, M.M.; Pelkki, M. 2018. Rehabilitation of cutover oak-hickory stands using prescribed burning and herbicide application: decadal responses of vegetation. In: Kirschman, J.E., comp. Proceedings of the 19th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-234. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 347–354. https://doi.org/10.2737/SRS-GTR-234.

Bataineh, M.M.; Portner, B.; Pelkki, M.; Ficklin, R. 2022. Prescribed fire first-order effects on oak and maple reproduction in frequently burned upland oak-hickory forests of the Arkansas Ozarks. Forests. 13: 1865. 16 p. https://doi.org/10.3390/f13111865.

Bova, A.S.; Dickinson, M.B. 2008. Beyond "fire temperatures": calibrating thermocouple probes and modeling their response to surface fires in hardwood fuels. Canadian Journal of Forest Research. 38(5): 1008–1020. https://doi.org/10.1139/X07-204.

Breuer, E.H. 1965. Thermocouples for forest fire research. Res. Note INT-31. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 7 p.

Brose, P.H.; Dey, D.C.; Phillips, R.J.; Waldrop, T.A. 2013. A meta-analysis of the fire-oak hypothesis: does prescribed burning promote oak reproduction in Eastern North America? Forest Science. 59(3): 322–334. http://dx.doi.org/10.5849/forsci.12-039.

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.

Butler, B.W.; Dickinson, M.B. 2010. Tree injury and mortality in fires: developing process–based models. Fire Ecology. 6(1): 55–79. https://doi.org/10.4996/fireecology.0601055.

Byram, G.M. 1959. Combustion of forest fuels. In: Davis, K.P., ed. Forest fire control and use. New York: McGraw-Hill Book Company: 61–89.

Dickinson, M.B.; Hudak, A.T.; Zajkowski, T. [and others]. 2016. Measuring radiant emissions from entire prescribed fires with ground, airborne and satellite sensors–RxCADRE 2012. International Journal of Wildland Fire. 25(1): 48–61. http://dx.doi.org/10.1071/WF15090.

Fons, W.L. 1946. Analysis of fire spread in light forest fuels. Journal of Agricultural Research. 72(13): 93–121.

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

Harper, M.D.; Phillips, W.W.; Haley, G.J. 1969. Soil Survey of Washington County, Arkansas. Washington, DC: U.S. Department of Agriculture, Soil Conservation Service and Forest Service in cooperation with Arkansas Agricultural Experiment Station. 94 p.

Hiers, J.K.; O'Brien, J.J.; Mitchell, R.J. [and others]. 2009. The wildland fuel cell concept: an approach to characterize fine-scale variation in fuels and fire in frequently burned longleaf pine forests. International Journal of Wildland Fire. 18(3): 315–325. https://doi.org/10.1071/WF08084.

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(3): 311–322. https://doi.org/10.1071/WF03063.

Kennard, D.K.; Outcalt, K.W.; Jones, D.; O'Brien, J.J. 2005. Comparing techniques for estimating flame temperature of prescribed fires. Fire Ecology. 1(1): 75–84. https://doi.org/10.4996/fireecology.0101075.

Kremens, R.L.; Smith, A.M.S.; Dickinson, M.B. 2010. Fire metrology: current and future directions in physics-based measurements. Fire Ecology. 6(1): 13–35. https://doi.org/10.4996/fireecology.0601013.

Lutes, D.C.; Keane, R.E.; Reinhardt, E. 2012. FOFEM 6.0 user guide. Missoula, MT: U.S. Department of Agriculture Forest Service, Forest Service, Rocky Mountain Research Station. 65 p.

McGranahan, D.A. 2020. An inconvenient truth about temperature-time data from thermocouples. Plant Ecology. 221(11): 1091–1104. https://doi.org/10.1007/s11258-020-01064-7.

Morris, T.W. 2007. Initial effects of prescribed fire and herbicide treatments in upland oak forests of the Arkansas Ozarks. Monticello, AR: University of Arkansas at Monticello. 143 p. M.S. thesis.

Pyne, S.J.; Andrews, P.L.; Laven, R.D. 1996. Introduction to wildland fire. 2nd ed. Revised. Hoboken, NJ: Wiley. 808 p.

R Development Core Team. 2013. R: a language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. https://www.R-project.org/. [Date accessed: October 30, 2023].

Rothermel, R.C.; Deeming, J.E. 1980. Measuring and interpreting fire behavior for correlation with fire effects. Gen. Tech. Rep. INT-93. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 4 p.

Scott, J.H.; Burgan, R.E. 2005. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. Gen. Tech. Rep. RMRS-153. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 72 p. https://doi.org/10.2737/RMRS-GTR-153.

Smith, A.M.S.; Talhelm, A.F.; Kolden, C.A. [and others]. 2016. The ability of winter grazing to reduce wildfire size and fire-induced plant mortality was not demonstrated: a comment on Davies et al (2015). International Journal of Wildland Fire. 25(4): 484–488. http://dx.doi.org/10.1071/WF15163.

Van Wagner, C.E.; Methven, I.R. 1978. Two recent articles on fire ecology. Canadian Journal of Forest Research. 8(4): 491–492. https://doi.org/10.1139/x78-075.

Walker, J.D.; Stocks, B.J. 1968. Thermocouple errors in forest fire research. Fire Technology. 4(1): 59–62. https://doi.org/10.1007/BF02588607.

Wally, A.L.; Menges, E.S.; Weekley, C.W. 2006. Comparison of three devices for estimating fire temperatures in ecological studies. Applied Vegetation Science. 9(1): 97–108. https://doi.org/10.1111/j.1654-109X.2006.tb00659.x.

INFLUENCES OF OVERSTORY PYROPHYTIC AND MESOPHYTIC TREES AND UNDERSTORY SOLAR IRRADIANCE ON LEAF LITTER AND WOODY DEBRIS FUEL MOISTURE RETENTION

Luiza G. Lazzaro, Heather D. Alexander, Jeffrey B. Cannon, and Michael J. Aspinwall

EXTENDED ABSTRACT

Following decades of fire exclusion, open-canopied, pyrophytic oak (*Quercus* spp.) and pine (*Pinus* spp.) landscapes across the central and Eastern United States are shifting to closed-canopied forests with a dense midstory occupied by shade-tolerant, often fire-sensitive species (i.e., mesophytes) (Alexander and others 2021). Encroaching species may create an understory with increased moisture due to lower solar irradiance beneath the canopy (Alexander and others 2021), and increased small leaf litter, which may compact fuel beds and decrease water evaporation (Cornwell and others 2015, Varner and others 2015), and increase woody debris (MacMillan 1988).

The primary objective of this study was to understand how fine fuel moisture dynamics beneath overstory trees varies by functional groups (pyrophytic pine; pyrophytic oaks; mesophytic oak) and may impact prescribed fire effectiveness in systems with varying levels of mesophytic tree encroachment. We hypothesized that the leaf litter and woody debris of pyrophytic pines (*Pinus palustris*) and pyrophytic oaks (*Quercus margaretta* and *Q. laevis*) will dry faster compared to a mesophytic oak (*Q. laurifolia*) because of their leaf traits (e.g., thickness, specific leaf area, curliness, and volume), which cause lack of fuel compaction, and lower woody density. We also hypothesized that under the canopy of pyrophytic pines and oaks, higher vapor pressure deficit (VPD) would hasten moisture loss of fuels compared to mesophytic oak, and hence influence flammability.

To understand the influence of overstory trees on fuel-drying environments, we conducted a fuel-drying experiment at The Jones Center at Ichauway, GA, which consisted largely of open stands of second-growth

Luiza G. Lazzaro, Graduate Student, College of Forestry, Wildlife and Environment, Auburn University, Auburn, AL 36849

Heather D. Alexander, Associate Professor, College of Forestry, Wildlife and Environment, Auburn University, Auburn, AL 36849

Jeffery B. Cannon, Landscape Ecologist, The Jones Center at Ichauway, Newton, GA 39870

Michael J. Aspinwall, Affiliate Faculty, College of Forestry, Wildlife and Environment, Auburn University, Auburn, AL 36849

longleaf pine woodlands and associated communities. We selected 15 focal trees, 5 in each of the studied functional groups. In summer 2022, we implemented five 1-day-long fuel-drying experiments. Each day, we fully hydrated 30 bags of leaf litter (10 per functional group) and 30 bags of 10-hour woody debris (10 per functional group) and placed them under each focal tree (fig. 1). To measure moisture loss, we collected and weighed two bags of each fuel every 2 hours for 10 hours. We also deployed microclimate sensors (Cannon and others 2022) to measure microclimate conditions beneath the focal trees.

Figure 1—Thirty soaked litter bags and 30 soaked 10-hour woody debris bags were placed under 15 trees from the bole to the edge of the crown. (Photo courtesy of Luiza Lazzaro)

Our preliminary results suggest that response time, that is, the time it takes fuels to reach 63 percent of relative moisture loss (Kreye and others 2012, Nelson and Hiers 2008) of the leaf litter, varied by functional group. Pyrophytic pine dried faster than pyrophytic oak and mesophytic oak leaf litter, respectively. We also found that response time also varied with the composition of the overstory. Leaf litter beneath pyrophytic pines dried faster than pyrophytic oak and mesophytic oak overstories, respectively. The response time of woody debris did not vary among functional groups, woody fuels dried faster beneath pyrophytic pine overstories. VPD was lowest beneath mesophytic oak compared to pyrophytic pines and pyrophytic oaks.

As we hypothesized, leaf litter of mesophytic oak had greater response time compared to pyrophytic pines and pyrophytic oaks, which could be explained by leaf litter traits, where their small leaves created a compacted fuel bed and decreased the loss of moisture (Kreye and others 2013, McDaniel and others 2021). We also observed that mesophytic

oak had a lower loss of moisture on leaf litter and woody debris, which could be explained by the denser canopy, decreased light incidence and consequently lower VPD (Alexander and Arthur 2010, Babl and others 2020, Canham and others 1994, Siegert and Levia 2011). The lack of differences in woody debris response time among functional groups could be explained by the time of experiment, for the density of the material woody debris should take longer to dry than leaf litter. Thus, we propose a longer experimental measurement period for future studies.

Our findings show that mesophytic species play a significant role in increasing fuelbed moisture levels by not only changing litter traits but through altering the microclimate and drying conditions of fuel beds. Our work also helps to classify the understudied species sand post oak (*Q. margaretta*) and laurel oak (*Q. laurifolia*) into the mesophytic spectrum. By improving our knowledge of fire, climate, and vegetation feedbacks, we can develop more effective conservation management strategies, such as defining more precise windows for prescribed fires in mixed forests.

LITERATURE CITED

Alexander, H.D.; Arthur, M.A. 2010. Implications of a predicted shift from upland oaks to red maple on forest hydrology and nutrient availability. Canadian Journal of Forest Research. 40: 716–726. https://doi.org/10.1139/X10-029.

Alexander, H.D.; Siegert, C.; Brewer, J.S. [and others]. 2021. Mesophication of oak landscapes: evidence, knowledge gaps, and future research. BioScience. 71: 531–542. https://doi.org/10.1093/biosci/biaa169.

Babl, E.; Alexander, H.D; Siegert, C.M.; Willis, J.L. 2020. Could canopy, bark, and leaf litter traits of encroaching non-oak species influence future flammability of upland oak forests? Forest Ecology and Management. 458: 117731. 11 p. https://doi.org/10.1016/j.foreco.2019.117731.

Canham, C.D.; Finzi, A.C; Pacala, S.W.; Burbank, D.H. 1994. Causes and consequences of resource heterogeneity in forests: interspecific variation in light transmission by canopy trees. Canadian Journal of Forest Research. 24: 337–349. https://doi.org/10.1139/x94-046.

Cannon, J.B.; Warren, L.T., Ohlson, G.C. [and others]. 2022. Applications of low-cost environmental monitoring systems for fine-scale abiotic measurements in forest ecology. Agricultural and Forest Meteorology. 321: 108973. 11 p. https://doi.org/10.1016/j.agrformet.2022.108973.

Cornwell, W.K.; Elvira, A.; van Kempen, L. [and others]. 2015. Flammability across the gymnosperm phylogeny: the importance of litter particle size. New Phytologist. 206 (2): 672–681. https://doi.org/10.1111/nph.13317.

Kreye, J.K.; Varner, J.M.; Hiers, J.K.; Mola, J. 2013. Toward a mechanism for eastern North American forest mesophication: differential litter drying across 17 species. Ecological Applications. 23: 1976–1986. https://doi.org/10.1890/13-0503.1.

Kreye, J.K.; Varner, J.M.; Knapp, E.E. 2012. Moisture desorption in mechanically masticated fuels: effects of particle fracturing and fuelbed compaction. International Journal of Wildland Fire. 21: 894–904. https://doi.org/10.1071/WF11077.

MacMillan, P.C. 1988. Decomposition of coarse woody debris in an old-growth Indiana forest. Canadian Journal of Forest Research. 18: 1353–1362. https://doi.org/10.1139/x88-212.

McDaniel, J.K.; Alexander, H.D.; Siegert, C.M.; Lashley, M.A. 2021. Shifting tree species composition of upland oak forests alters leaf litter structure, moisture, and flammability. Forest Ecology and Management. 482: 118860. 11 p. https://doi.org/10.1016/j.foreco.2020.118860.

Nelson, R.M., Jr.; Hiers, J.K. 2008. The influence of fuelbed properties on moisture drying rates and timelags of longleaf pine litter. Canadian Journal of Forest Research. 38: 2394–2404. https://doi.org/10.1139/X08-078.

Siegert, C.M.; Levia, D.F. 2011. Stomatal conductance and transpiration of co-occurring seedlings with varying shade tolerance. Trees. 25: 1091–1102. https://doi.org/10.1007/s00468-011-0584-4.

Varner, J.M.; Kane, J.M.; Kreye, J.K.; Engber, E. 2015. The flammability of forest and woodland litter: a synthesis. Current Forestry Reports. 1: 91–99. https://doi.org/10.1007/s40725-015-0012-x.

Hardwoods

ADVANCING ESTABLISHED NATURAL WHITE OAK (*QUERCUS ALBA*) REPRODUCTION THROUGH A MIDSTORY REMOVAL

Canaan J. Dugger and Wayne K. Clatterbuck

ABSTRACT

A midstory release was used to enhance naturally established white oak (*Quercus alba*) seedlings that resulted from a previous bumper acorn crop at the Forest Resources Research and Education Center in Oak Ridge, TN, during the summer of 2020. Total number of species, height classes, and photosynthetically active radiation were collected on the plot level and height and root collar diameter (RCD) were collected on the subplot level over two complete growing seasons. White oak seedlings receiving the release treatment showed significant differences in percent height and RCD growth on the subplot level as well as significant changes in height classes on the plot level following each growing season and from the first to last year of the study compared to the control. The percentage of light received following the midstory removal was not a significant predictor of percent height growth after two growing seasons.

INTRODUCTION

White oak (*Quercus alba*) is a keystone species within our eastern forest ecosystems. A myriad of benefits, including hard mast and habitat for wildlife species, valuable lumber to society, and vital ecosystem services such as soil stabilization, air purification, and the sequestration of carbon are provided by white oak (Cavender-Bares 2016). Oak ecosystems spread across 46 million hectares (Loftis 2004) which makes up over half of eastern forests (Dey 2014b). However, the ability of white oak to successfully regenerate and recruit into the upper canopy is currently threatened.

Due to the vast shifts from historically frequent disturbance regimes over the last century, forest microclimates have been drastically transformed to foster many of the major shade-tolerant oak competitor species. Without the occurrence of frequent disturbances, dense midstory canopies are formed which obstruct light from reaching the forest floor. This has resulted in the competitiveness of white oak regeneration across its historically dominated sites to be drastically reduced.

White oak possesses many physiological adaptations in response to its relationship to disturbance. These adaptions should be utilized to implement successful silvicultural treatments. Decades of scientific studies have documented that adequate size and number of advance reproduction is necessary to regenerate oak (Clatterbuck 2019, Loftis 1990, Sander 1972). A midstory release can be a practical option to recruit seedlings into the future upper canopy (Hackworth and others 2020). By providing these suitable levels of diffuse light to the forest floor, white oak seedling competitiveness should increase. This study investigates growth and development of advance reproduction of white oak from a midstory removal for two growing seasons following a bumper acorn year.

METHODS

Study Site

This study was initiated in the summer of 2020 across the grounds of the Forest Resources Research and Education Center (FRREC) in Oak Ridge, TN. The forest within the study area was

Canaan J. Dugger, Graduate Student, University of Tennessee, School of Forest Resources, Knoxville, TN 37996
Wayne K. Clatterbuck, Professor, University of Tennessee, School of Forest Resources, Knoxville, TN 37996

two-aged and the upper canopy was dominated by white oak. The average diameter at breast height (d.b.h.) for these overstory white oaks ranged from 51 to 90 cm. Their heights ranged from 26 to 37 m. Basal area ranged from 6.5 to 7.8 m^2/ha across all sites.

Species such as hickory (*Carya* spp.), chestnut oak (*Quercus montana*), northern red oak (*Quercus rubra*), and shortleaf pine (*Pinus echinata*) also composed the upper canopy class. The intermediate and midstory canopy classes consisted of American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), sweetgum (*Liquidambar syraciflua*), yellow-poplar (*Liriondendron tulipifera*), black cherry (*Prunus serotina*), eastern redbud (*Cercis canadensis*), dogwood (*Cornus florida*), sourwood (*Oxydendrum arboreum*), and black gum (*Nyssa sylvatica*).

In addition to the substantial amounts of white oak advanced reproduction, mostly from the 2019 acorn bumper crop, there were also seedlings present of species such as maple, hickory, dogwood, oak, sassafras (*Sassafras albidum*), American beech, sweetgum, yellow-poplar, black cherry, shortleaf pine, American holly (*Ilex opaca*), blackgum, eastern redbud, umbrella magnolia (*Magnolia tripetala*), locust (*Robinia* spp.), and elm (*Ulmus* spp.). Estimated ground coverage in these closed canopy stands was sparse, <20 percent. Heavy leaf litter, rocky topsoil, and coarse woody debris were all present on the forest floor. Soils were predominately composed of well-drained Fullerton-Pailo complex with varying slopes ranging from 5 to 35 percent (USDA NRCS 2023). All plots were homogenous in their soil profiles, vegetation and tree species compositions, abundance of white oak advance reproduction, and site productivities.

Study Installation and Design

Plots were subjectively placed in areas containing large quantities of white oak advance reproduction in the summer of 2020. A total of twenty-six 2-m by 2-m plots and fifty-two 0.5-m by 0.5-m subplots were installed prior to receiving the designation of control or release treatment. Three to four plots were installed at four release treatment locations while one to four were installed at seven control locations, but each plot always contained two subplots within it. Both plots and subplots were evenly divided into control and release treatment groups. The total area representing all 26 plots was 104 m^2 and the subplot total area was 13 m^2.

The four plot location areas received a midstory removal treatment in the summer of 2020 that averaged 0.04 to 0.08 ha. Midstory tree species that produce copious amounts of seed annually such as maple, American beech, yellow-poplar, and black cherry were targeted by felling and/or girdling with a chainsaw. In addition, stumps were treated with a 50-percent glyphosate solution to deter sprouting. One to four dominant or codominant white oak trees were retained at release areas to provide future seed source as well as cast some shade on seedlings. The basal area remained within the range of 6.5 to 7.8 m^2/ha in both the release and control treatments, suggesting that the midstory release did not remove overstory basal area. Age, diameter, and height of overstory trees remaining after the midstory treatment were measured.

Data Collection

Growth and light measurements taken in June to July of 2020 served as a baseline to compare each of the two complete growing seasons (2020–2021 and 2021–2022). Woody species were counted within each plot. Seedlings within the plot were further categorized into height classes (<30.5, 30.5 to <70, 70 to <91.4, and 91.4 to 122 cm). Nonwoody ground vegetation composed less than 5 percent of the ground coverage and was not measured.

An aluminum tag with a unique identifier was attached with galvanized wire at the base of each seedling within the subplots. Seedling heights were measured from the ground to the apical bud using a meter stick. Digital calipers were used to measure root collar diameter (RCD) where the base of the seedling stem met the soil. These measurements were taken prior to the application of the release treatment in June through July of 2020 and were repeated at the end of the next two growing seasons in September through October. Subplot species with missing tags were assumed to be either missing or dead and removed from the data.

An AccuPAR model LP-80 ceptometer (Meter Group®, Pullman, WA) was implemented to collect photosynthetically active radiation (PAR) measurements on control and release plots. These measurements were taken twice a year from 2020 to 2022 during July through August on two overcast-free days, with the exception of control plot location 11, which was not installed until the fall following the summer release cut.

The ceptometer was calibrated prior to the start of each year's data collection and light measurements were taken at sternum height (1 m) above the ground during solar noon from 12:00 to 2:00 p.m. Eastern Time. Prior to taking measurements at each plot location, a baseline PAR reading was taken consistently in an exposed, open area of full sunlight at the FRREC adjacent to plots. A total of eight light readings were taken on each plot to order to ensure consistency over 2 days. Four of these PAR measurements were taken from plot center facing outward. The remaining four measurements were taken from each corner facing inward to the plot center.

Data Analysis

Data were analyzed in the R statistical package (R Core Team 2012) with the two complete growing seasons as well as between the first year's baseline measurements and last year (2022). A Wilcoxon test was applied to examine the subplot level effects of the midstory release, comparing average height and RCD between treatments within each growing season as well as the first to the last year, regardless of species. In addition, the average percent height and RCD growth of white oak and maple seedlings were compared between treatments by growing season as well as the first to last year. Seedlings that were missing a tag or were otherwise dead, were removed from the dataset across all years for analysis involving average percent growth, but not for comparisons of average measurements. A *P*-value <0.05 indicated a statistically significant difference. A chi-square test of independence compared plot-level proportional abundance between white oak and maple seedlings by treatment and growing season as well as from the first year to the last year. Proportional changes in white oak and maple seedling height classes between control and release treatments by growing seasons and from first to last year was also analyzed on the plot level. A *P*-value >0.05 indicated proportions were statistically similar.

RESULTS

The average age of white oaks was 122 years with an average height and d.b.h. of 30.5 m and 60 cm, respectively. The site index was estimated to be 21 m at 50 years (USDA NRCS 2023). White oaks composed greater than 70 percent of the total seedlings in the control and release plots initially (2020), maples 18 percent, and miscellaneous species 8 percent (fig. 1). Red maple and sugar maple were grouped together into a maple category for the analyses. Through subsamples, red maple composed approximately 70 percent, and sugar maple 30 percent, of the maple category.

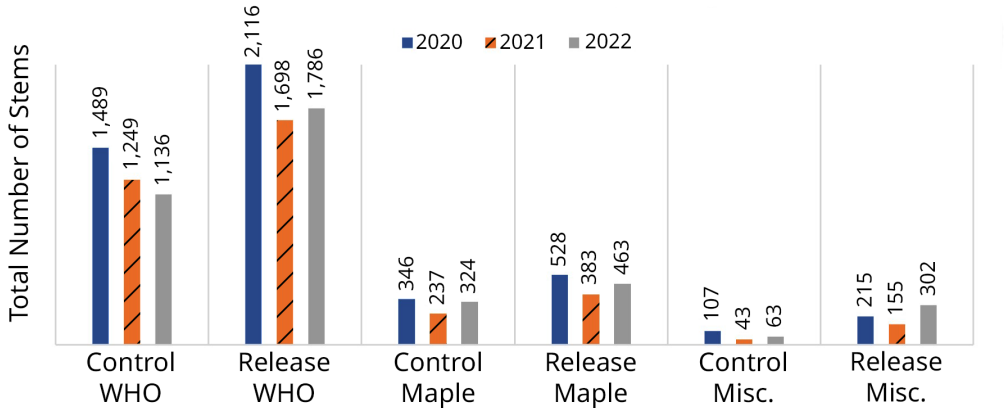

Figure 1—Plot level abundance by treatment and year for white oak (WHO) and maple. Miscellaneous species included chestnut oak, black cherry, hickory, sassafras, beech, northern red oak, black oak (*Quercus velutina*), black gum, dogwood, sweetgum, elm spp. , locust spp., and sourwood.

Only white oak and maple were considered in this research due to their abundance. A total of 782 seedlings were tagged across 52 subplots during the summer of 2020. At the end of the growing season, in the fall of 2022, there was a total of 557 tagged woody seedlings remaining (table 1). A total of 4,801 seedlings were counted across all 26 plots in the summer of 2020. By the conclusion of this study, in the fall of 2022, there were 4,074 seedlings counted (fig. 1).

Response of White Oak

A total of 624 white oak seedlings were tagged at the start of this study in the summer of 2020. At the conclusion of this study, in the fall of 2022, there were 492 seedlings (table 1). The midstory removal treatment observed proportional changes in white oak seedlings that advanced into the taller height classes on the plot level over the two complete growing seasons as well as from the first to last year (table 2). From 2020 to 2021 (P <0.0001) and 2021 to 2022 (P <0.0001), release treatment height classes did not show similar distribution proportions. This same trend was also observed in the first (P = 0.0112) and second (P = 0.0031) complete growing season in the control treatment plots. Changes in height class distributions between the first base year and last year were also not proportionally similar in control (P <0.0001) or release (P <0.0001) treatment plots. Changes in plot-level height classes were proportionally similar between release and control treatments in 2021 but not in 2022 (P <0.0001).

Table 1—Total number of tagged white oak and maple seedlings by treatment and year on the subplot level

Treatment and species	2020 (baseline)	2021	2022
Release	462	334	312
White oak	354	273	265
Maple	78	47	33
Control	320	275	245
White oak	270	251	227
Maple	30	14	11
Total	782	609	557

The midstory release treatment appeared to affect the average percent growth of white oak height and RCD on the subplot level following each growing season as well as from the first to last year of this study (table 3). When comparing control to release subplot level treatments between 2020 and 2021 (P <0.0001), between 2021 and 2022 (P <0.0001), and between 2020 and 2022 (P <0.0001), the percent height growth of white oak seedlings had statistically significant differences between all of them. Similar effects were also observed between release and control treatments after each growing season (P <0.0001, P <0.0224, respectively) as well as from the first to last year (P <0.0001), on the percent growth of RCD. When each growing season was directly compared, average percent changes of white oak RCD growth were statistically different in both release and control subplot treatments (P <0.0001, P <0.0001, respectively). These same statistically significant differences were also observed in white oak average precent height growth between

Table 2—Number of plot-level white oak seedlings in each height class and percent total by treatment and year

		2020 (baseline)		2021		2022	
Treatment and height class		Number	Percent	Number	Percent	Number	Percent
Release	<30.5 cm	2,011	95	1,532	90	1,470	82
	30.5–70 cm	105	5	164	9	282	15
	70–91 cm	0	0	2	1	33	3
	91–122 cm	0	0	0	0	1	1
Control	<30.5 cm	1,410	95	1,152	92	1,006	88
	30.5–70 cm	79	5	97	8	127	11
	70–91 cm	0	0	0	0	3	1

There were significant (P <0.05) proportional differences after each growing season for both the release and control treatments. Treatments were significantly different from base year (2020) through 2022.

Table 3—Percent height and root collar diameter growth of white oak seedlings by treatment and year

Year and treatment		Height	Root collar diameter
		~~ *percent* ~~~~~~~	
2020–2021	Control	16.5	12.6
	Release	27.2	50.0
2021–2022	Control	10.2	35.2
	Release	16.5	57.1
2020–2022	Control	27.8	59.5
	Release	48.6	98.3

Sample size is 227 white oak seedlings in the control subplots and 265 seedlings in the release subplots.

There was a significant (P <0.05) seedling percent growth increase for released seedlings in height and root collar diameter for each growth period and treatment.

the two growing seasons for both release and control subplot treatments (P <0.0001, P = 0.0002, respectively).

Response of Maple

There was a total of 108 maple seedlings tagged in the summer of 2020, 30 in control subplots and 78 in release subplots (table 1). In 2021 there were 61 seedlings: 14 control and 47 release. By the end of the growing season in 2022 there were a total of 44 maple seedlings left across all 52 subplots: 11 control and 33 release.

Maple also experienced changes in population distributions within height classes from the midstory removal treatment (table 4). Maple receiving the release treatment had statistically significant changes in population distributions of height classes between 2020 and 2021 (P = 0.0285) and between 2021 and 2022 (P = 0.0044) on the plot level. No significant difference in proportional changes of control height classes was observed when comparing 2020 to 2022. However, there was a significant change in release treatment height classes from 2020 to 2022 (P <0.0001). Control maple did not have any significant changes in proportions in height class distributions after each growing season or from the first to last year. There were also no proportional differences in maple height class populations when comparing control to release treatments in both 2021 and 2022.

No significant differences were observed when average percent height and RCD growth were compared between control or release treatments over the two complete growing seasons as well as between the first and last year (table 5). However, when comparing the first growing season to the second, there were significant differences in release treatment height (P = 0.0015) and RCD (P = 0.0019).

Table 5—Percent height and root collar diameter growth of maple seedlings by treatment and year

Year and treatment		Height	Root collar diameter
2020–2021	Control	80.8	39.5
	Release	44.3	75.4
2021–2022	Control	16.8	38.2
	Release	20.8	44.2
2020–2022	Control	76.8	94.8
	Release	127.7	134.3

Sample size is 11 maple seedlings in the control subplots and 33 seedlings in the release subplots.

There was a significant (P <0.05) seedling percent growth increase for released seedling treatment in height and root collar diameter after two growing seasons (2020–2022).

Table 4—Number of plot-level maple seedlings in each height class and percent total by treatment and year

		2020 (baseline)		2021		2022	
Treatment and height class		Number	Percent	Number	Percent	Number	Percent
Release	<30.5 cm	511	97	358	93	299	92
	30.5–70 cm	17	3	25	7	24	7
	70–91 cm	0	0	0	0	1	1
Control	<30.5 cm	326	94	221	93	299	92
	30.5–70 cm	20	6	16	7	24	7
	70–91 cm	0	0	0	0	1	1

There was a significant (P <0.05) proportional difference in the <30.5-cm height class after each growing season for the release treatment only. The release treatment was significantly (P <0.05) different from base year through 2022.

Photosynthetically Active Radiation (PAR)

Light was not found to be a significant predictor of white oak seedling height in either control or release treatment groups after two growing seasons. Figure 2 depicts the range of percent height growth and percent light received for both control and release treatments. White oak seedlings in the release treatment had a much wider range of percent light received and percent height growth compared to the control group. The majority of control seedlings were clustered within the 3- to 10-percent light received and 2- to 42-percent height growth range and release seedlings were clustered within the 5- to 20-percent light received and 10- to 30-percent height growth.

DISCUSSION

The results of this study support the existing literature which suggests, with adequate accumulation of advanced white oak reproduction, a midstory removal treatment can provide a positive stimulus on abundance, vigor, and growth of seedlings (Dillaway and Stringer 2007, Lockhart and others 2000, Lorimer and others 1994, Motsinger and others 2010). White oak held an advantage across the duration of this study on both the plot and subplot level over its only major competitor, maple (fig. 1, table 1) which was likely attributed to a previous large bumper acorn crop season. This level of abundance is a positive indicator that white oak is in a position to maintain its current advantage over local competitors. If these levels of white oak abundance continue to increase or are

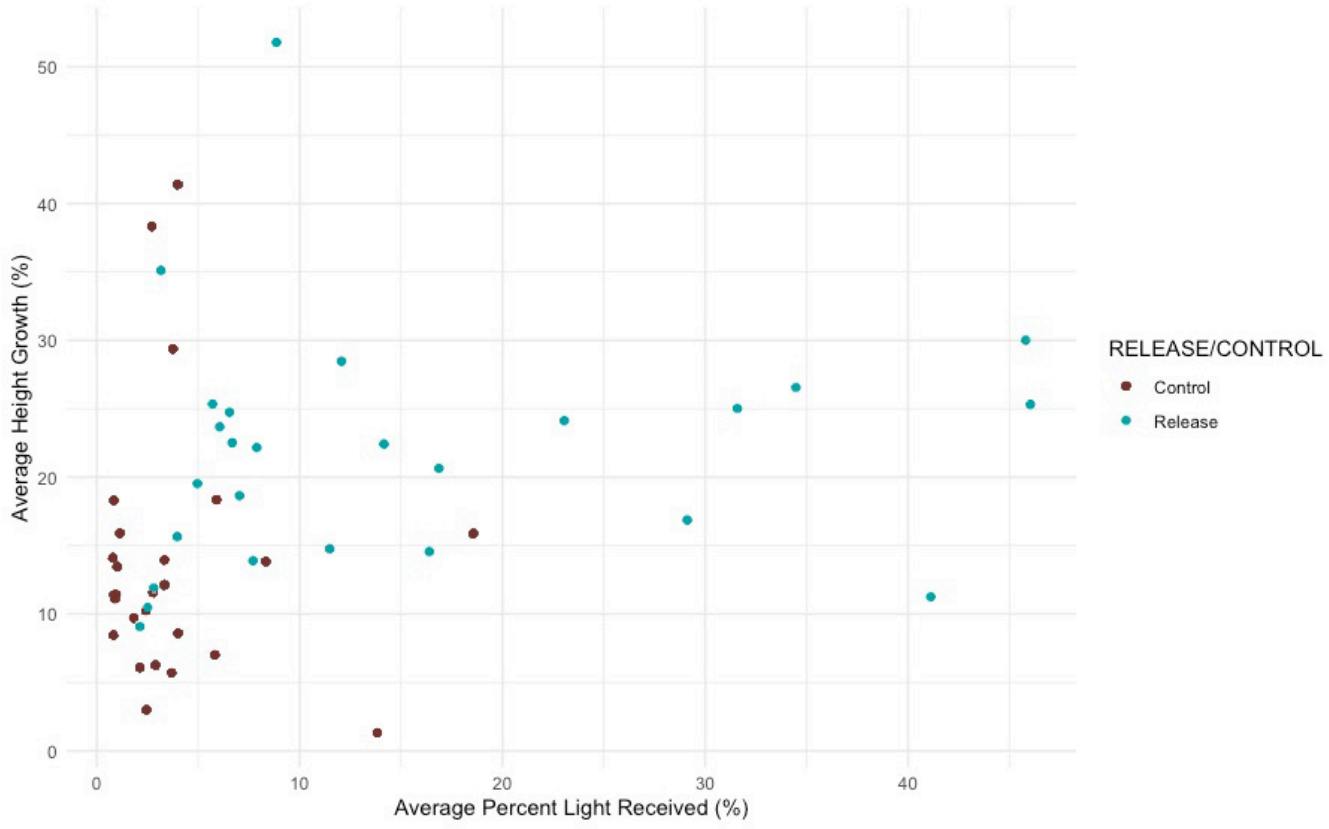

Figure 2—Scatterplot of control and release white oak seedling height growth in response to percent light received. Each point represents average height growth of the two subplots for each plot and the photosynthetically active radiation average of four center plot measurements on one date in 2021 and 2022 (n = 52 plots; 26 control and 26 release).

maintained, unimpeded by competition, the sheer quantity of established white oak seedlings is on track to meet the existing recommendations of Clatterbuck and Hodges (1988), Sander (1972), Sander and others (1976), and Stringer (2016), who suggested having 100–200 large seedlings per 0.4 ha to yield 40–60 mature trees in the future.

The method for plot placement likely caused discrepancies in reported abundance as well. The placement and installation of plots followed the "subjective without preconceived bias" approach described by Clatterbuck (1996), Mueller-Dombois and Ellenburg (1974), and Pfister and Arno (1980) in which the placement of plots was subjectively chosen to sample the landscapes, compositions, and canopy structures present in our study area. This approach allowed areas with abundant white oak advanced reproduction to be sampled and to determine how they would respond to a midstory release. Basal area range measured before and after midstory release and in control treatment were similar, suggesting the midstory release did not create any large gaps in the overhead canopy.

Strong intraspecific competition occurred between advance white oak reproduction (fig. 1). Usually, density of oak saplings is much lower than seedlings, which indicates a bottleneck is occurring between these stages due to competition (Crow 1988, Nowacki and others 1990). Even when white oak is ample in both the overstory and understory, the absence of saplings in the midstory prevents the potential of ingrowth of white oak into the overstory (Clatterbuck 2019, Dey 2014a, Signell and others 2005). A study conducted by Beck (1970) demonstrated that woody and herbaceous competition following an overstory removal cut was not detrimental for the first couple of years but within five growing seasons red oak seedlings were overtopped. A midstory removal or crop tree release could be an effective method to slowly introduce the transmittance of diffuse light through the canopy to continue ascendance of oak into the overstory.

Maple abundance and growth has been observed to quickly catch up to white oak seedlings in response to increasing levels of diffuse light created by a midstory removal. Hartman and others (2005) demonstrated that maple with high abundance overtopped oak in as little as 10 years under a shelterwood system. A study in Kentucky evaluating responses of oak and competitors to a midstory removal reported that red maple had a mean 6-year relative height growth of 459 percent over other oak species and made up 55 percent of seedlings in release treatment plots (Craig and others 2014). Little information is available from the literature on the influence of bumper acorn crops on the success of a midstory release treatment for the growth of white oak advance reproduction and associated competitors. This study provides an opportunity to understand more on the processes involved with successfully culturing large quantities of established white oak reproduction resulting from a bumper acorn crop on average productivity sites. Intraspecific competition among the numerous white oak advance reproduction and the fewer maple may impact maple growth.

For two complete growing seasons since a midstory release, significant shifts in height class distributions of white oak seedlings were observed, which suggests a positive response to the midstory removal treatment (table 2). All plots that received the release treatment had greater relative proportions in the taller height classes after the conclusion of each growing season. Control plots also had increasing height class distribution shifts after each growing season; however, it was less than the release. This is likely due to white oak's intermediate shade tolerance, allowing it to maintain height growth even in lower light levels.

These results are in accordance with the recommendations of Sander (1972), who states that successful oak management can be achieved with an abundance of white oak seedlings with heights >130 cm. After two complete growing seasons no white oak seedlings were in this recommended 130-cm height class. However, if height growth continues at this rate, the 130-cm height class could be achieved in a few years. Percent height and RCD growth of release white oak seedlings on the subplot level were observed to have a response to the increased diffuse light levels created by the midstory removal. There were statistically significant differences in percent height and RCD growth between

treatments of white oak seedlings following each growing season as well as from the first to the last year (table 3).

There are limited studies investigating RCD and its influences on successful management. White oak seedlings have a judicious growth strategy when allocating resources below ground during the first year prior to aboveground response (Dillaway and Stringer 2007). In this study as well as a study by (Parrott and others 2012), greater ground line diameters were observed in release treated seedlings. Previous research reports that a midstory removal has direct positive influences on both height and RCD growth, vigor, and abundance of upland oak species for up to 10 years following the treatment (Frank and others 2018, Lhotka and Loewenstein 2013, Lockhart and others 2000, Loftis 1990, Miller and others 2017).

CONCLUSIONS

The response of white oak seedling indicates that there was a positive influence from the increase in light levels created by the midstory removal. In contrast to several other similar studies (Dillaway and Stringer 2007, Parrott and others 2012) investigating the effects of a midstory removal, maple was limited comparatively and did not appear to have serious effects on the competitive success of white oak seedlings after two growing seasons. This suggests that with an abundant accumulation of advanced seedlings provided from a large bumper acorn crop season, white oak can be competitive and find success on average productivity upland sites when diffuse light levels are increased to support increased height growth. Intraspecific competition among the numerous white oak seedlings can also have positive and negative effects on seedling growth.

This research provides a valuable opportunity to understand the preliminary processes involved with bumper acorn crop advance reproduction and the influences that a subsequent midstory removal can have over its recruitment success. White oak was observed to have a positive response in abundance and growth following the midstory removal. Maple growth was also observed to have a positive response to the increased light levels in the release treatments. However, white oak seedling abundance overcompensates the number of maple seedlings by a ratio of 4:1. The midstory release response of maple, though much fewer numbers, was greater than white oak and may necessitate competition control in the future if a crop tree release or shelterwood harvest is implemented.

Alternatively, the overabundance of white oak advance reproduction may discourage the growth of the smaller maple component and influence growth among adjacent white oak stems. Many of the maple present at the plot locations were established prior to the bumper acorn crop season. In addition, maple was only present on a fraction of the plots and subplots. Future studies should investigate both the longer-term growth effects of increased light on large accumulation of bumper acorn advanced white oak reproduction in relation to competitor species such as maple.

ACKNOWLEDGMENTS

Thanks to Kevin Hoyt and the wonderful staff at the University of Tennessee Arboretum at Oak Ridge for being flexible and accommodating with our schedules as well as allowing us to conduct a multiple-year study on their grounds.

LITERATURE CITED

Beck, D.E. 1970. Effect of competition on survival and height growth of red oak seedlings. Res. Pap. SE-56. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 7 p.

Cavender-Bares, J. 2016. Diversity, distribution and ecosystem services of the North American oaks. International Oaks. 27: 37–48.

Clatterbuck, W.K. 1996. A community classification system for forest evaluation: development, validation, and extrapolation. Environmental Monitoring and Assessment. 39: 299–321. https://doi.org/10.1007/BF00396151.

Clatterbuck, W.K. 2019. What do we know about oaks? Keystones of oak silviculture. In: Clark, S.L.; Schweitzer, C.J., eds. Oak symposium: sustaining oak forests in the 21st century through science-based management. e-Gen. Tech. Rep. SRS-237. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 3–12. https://doi.org/10.2737/SRS-GTR-237.

Clatterbuck, W.K.; Hodges, J.D. 1988. Development of cherrybark oak and sweet gum in mixed, even-aged bottomland stands in central Mississippi, U.S.A. Canadian Journal of Forest Research. 18: 12–18. https://doi.org/10.1139/x88-003.

Craig, J.M.; Lhotka, J.M.; Stringer, J.W. 2014. Evaluating initial responses of natural and underplanted oak reproduction and a shade-tolerant competitor to midstory removal. Forest Science. 60: 1164–1171. https://doi.org/10.5849/forsci.13-602.

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

Dey, D.C. 2014a. Oak regeneration ecology and dynamics. In: Waldrop, T.A., ed. Wildland fires in the Appalachians: discussions among managers and scientists. Gen. Tech. Rep. SRS-199. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 3–11. https://doi.org/10.2737/SRS-GTR-199.

Dey, D.C. 2014b. Sustaining oak forests in eastern North America: regeneration and recruitment, the pillars of sustainability. Forest Science. 60: 926–942. https://doi.org/10.5849/forsci.13-114.

Dillaway, D.; Stringer, J. 2007. Effects of shade on the growth of natural and artificially established white oak (*Quercus alba* L.) regeneration. In: Buckley, D.S.; Clatterbuck, W.K., eds. Proceedings of the 15th central hardwood forest conference. e-Gen. Tech. Rep. SRS-101. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 638–643. https://doi.org/10.2737/SRS-GTR-101.

Frank, G.S.; Rathfon, R.A.; Saunders, M.R. 2018. Ten-year responses of underplanted northern red oak to silvicultural treatments, herbivore exclusion, and fertilization. Forests. 9: 571. 15 p. https://doi.org/10.3390/f9090571.

Hackworth, Z.J.; Lhotka, J.M.; Stringer, J.W. 2020. Midstory removal facilitates growth but reduces competitiveness of oak reproduction prior to and after shelterwood establishment cutting. Forest Science. 66: 371–381. https://doi.org/10.1093/forsci/fxz083.

Hartman, J.P.; Buckley, D.S.; Sharik, T.L. 2005. Differential success of oak and red maple regeneration in oak and pine stands on intermediate-quality sites in northern Lower Michigan. Forest Ecology and Management. 216: 77–90. https://doi.org/10.1016/j.foreco.2005.05.041.

Lhotka, J.M.; Loewenstein, E.F. 2013. Development of three underplanted hardwood species 7 years following midstory removal. Southern Journal of Applied Forestry. 37(2): 81–90. https://doi.org/10.5849/sjaf.12-001.

Lockhart, B.R.; Hodges, J.D.; Gardiner, E.S. 2000. Response of advance cherrybark oak reproduction to midstory removal and shoot clipping. Southern Journal of Applied Forestry. 24(1): 45–50. https://doi.org/10.1093/sjaf/24.1.45.

Loftis, D.L. 1990. Predicting post-harvest performance of advance red oak reproduction in the southern Appalachians. Forest Science. 36: 908–916.

Loftis, D.L. 2004. Upland oak regeneration and management. In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions, and sustainability. Gen Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 163–167.

Lorimer, C.G.; Chapman, J.W.; Lambert, W.D. 1994. Tall understorey vegetation as a factor in the poor development of oak seedlings beneath mature stands. Journal of Ecology. 82: 227–237. https://doi.org/10.2307/2261291.

Miller, G.W.; Brose, P.H.; Gottschalk, K.W. 2017. Advanced oak seedling development as influenced by shelterwood treatments, competition control, deer fencing, and prescribed fire. Journal of Forestry. 115: 179–189. https://doi.org/10.5849/jof.16-002.

Motsinger, J.R.; Kabrick, J.M.; Dey, D.C. [and others]. 2010. Effect of midstory and understory removal on the establishment and development of natural and artificial pin oak advance reproduction in bottomland forests. New Forests. 39: 195–213. https://doi.org/10.1007/s11056-009-9164-5.

Mueller-Dombois, D.; Ellenberg, H. 1974. Aims and methods of vegetation ecology. New York: John Wiley. 547 p.

Nowacki, G.J.; Abrams, M.D.; Lorimer, C.G. 1990. Composition, structure, and historical development of northern red oak stands along an edaphic gradient in north-central Wisconsin. Forest Science. 36: 276–292. https://doi.org/10.1093/forestscience/36.2.276.

Parrott, D.L.; Lhotka, J.M.; Stringer, J.W.; Dillaway, D.N. 2012. Seven-year effects of midstory removal on natural and underplanted oak reproduction. Northern Journal of Applied Forestry. 29(4): 182–190. https://doi.org/10.5849/njaf.12-001.

Pfister, R.D.; Arno, S.F. 1980. Classifying forest habitat types based on potential climax vegetation. Forest Science. 26: 52–70. https://doi.org/10.1093/forestscience/26.1.52.

R Core Team. 2012. R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. http://www.R-project.org/.

Sander, I.L. 1972. Size of oak advance reproduction: key to growth following harvest cutting. Res. Pap. NC-79. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 6 p.

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

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. https://doi.org/10.1016/j.foreco.2005.07.006.

Stringer, J.W. 2016. Oak regeneration challenges. In: Keyser, P.D.; Fearer, T.; Harper, C.A., eds. Managing oak forests in the Eastern United States. Boca Raton, FL: CRC Press: 80–91. https://doi.org/10.1201/b19076.

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2023. Web soil survey. https://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx. [Date accessed: April 23, 2023].

TIMING OF DORMANT SEASON INUNDATION IS CRUCIAL TO FLOOD EFFECTS ON BOTTOMLAND RED OAKS

Benjamin A. Babst, Richard D. Sample, and James Cook

EXTENDED ABSTRACT

Since flooding in winter and early spring is an integral part of bottomland hardwood ecosystems, moderately flood-tolerant red oaks, such as Nuttall oak (*Quercus texana*), should be well adapted to this dormant season inundation. However, altered hydrologic regimes and unusual weather patterns are now more frequently exposing bottomlands to flooding at times of the year when trees are transitioning into or out of dormancy, which may have physiological impacts on these red oak seedlings.

For example, Sample and Babst (2020) found that soil flooding in fall (from September 20 through December 7) of Nuttall oak seedlings resulted in lower fine root biomass and decreased nitrogen accumulation during flooding and reduced overwinter survival after flooding ended. In unflooded seedlings, there appeared to be much less uptake of some nutrients (e.g., sulfur, phosphorus, and potassium) during the fall compared to nitrogen, and so fall flooding had less impact on accumulation of those nutrients. On the other hand, accumulation of iron, manganese, and sodium greatly increased during soil flooding (Sample and Babst 2020).

To expand upon this work, we examined how the duration of dormant season flooding may affect the performance of seedlings of Nuttall oak and the moderately flood-intolerant Shumard oak (*Q. shumardii*) (Sample and others 2023). We flooded dormant Nuttall and Shumard oak seedlings for 0, 1, 2, or 3 months. Bareroot 1-0 seedlings were potted in January and were in the field with full sun. Pots were placed in tanks to administer flooding to 2.5 cm above the soil line, and all flooding was initiated on February 18. The 1-month flood treatment ended when bud break began, and the 2- and 3-month flood treatments extended 1 and 2 months beyond bud break, respectively.

Flooding that was restricted to the winter dormancy period (i.e., 1-month flood treatment) had no effect on Nuttall oak but reduced the growth of Shumard oak seedlings in the spring. Leaf, root, and whole seedling

Benjamin A. Babst, Associate Professor, Arkansas Forest Resources Center, and University of Arkansas at Monticello, College of Forestry, Agriculture and Natural Resources, Monticello, AR 71656

Richard Sample, Forest Ecologist, U.S. Department of Agriculture, Forest Service, Bedford, IN 47421

James Cook, Graduate Student, University of Arkansas at Monticello, College of Forestry, Agriculture and Natural Resources, Monticello, AR 71656; Present address: Crew Leader, Bartlett Tree Experts, White Stone, VA 22578

biomass of winter-flooded Shumard oak seedlings remained less than unflooded seedlings at the end of the growing season.

Flooding that extended beyond budbreak (i.e., 2- and 3-month flood treatments) resulted in reduced leaf area and root biomass accumulation for both species. Flooding did not result in increased Nuttall oak seedling mortality, but there was 52 percent and 94 percent mortality in Shumard oak seedlings when flooding extended 1 month and 2 months beyond budbreak, respectively.

Without further flooding during summer, Nuttall oak seedlings in all flood treatments produced leaf area and fine root mass similar to unflooded seedlings by the end of the growing season, indicating the recovery of their ability to take up water and nutrients. The less flood-tolerant Shumard oak seedlings did not fully recover from even the winter-only flood treatment. As a result of slower recovery of resource acquisition tissues, only those Nuttall oak seedlings that had been flooded into late spring (i.e., 3-month flood treatment) had reduced biomass at the end of summer, primarily due to reduced taproot mass. Hence, flooding that extends several weeks beyond budbreak before it subsides will probably have little impact on Nuttall oak but flooding that extends more than 1 month beyond budbreak may cause some stress and reduce the size of the storage organ (i.e., taproot). Such a reduction may have consequences for overwintering or for the ability to tolerate other stresses that may occur over the summer.

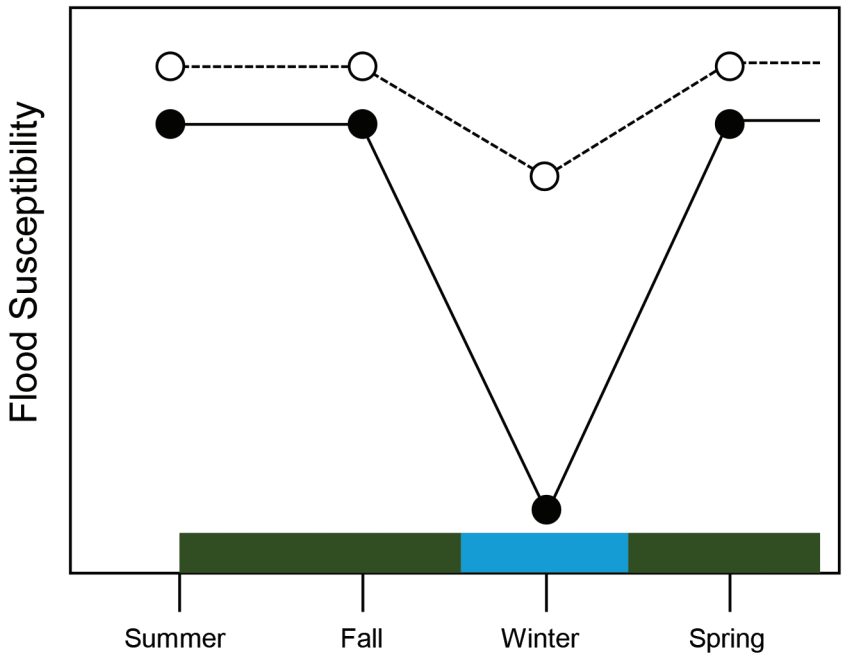

Figure 1—Conceptual model of flood susceptibility in moderately flood-tolerant red oaks, such as Nuttall oak and willow oak (filled symbols, solid line), and flood-intolerant red oaks, such as Shumard oak (open symbols, dashed line). Red oaks do not tolerate prolonged flooding during the summer and fall. The moderately flood-tolerant red oaks become less susceptible to flood only during periods of low activity during winter, whereas the intolerant red oaks do not tolerate prolonged flooding even during the winter. The light blue bar across the bottom indicates the time of year when flooding would be most appropriate in a greentree reservoir to retain red oaks.

To reduce risk of stress and mortality, our results suggest that managers can try to drain flood waters from greentree reservoirs by 1 month after budbreak, where regeneration of Nuttall oak is desired. Bottomland afforestation projects may want to avoid planting Nuttall oak seedlings in locations that are likely to remain flooded in late spring and restrict Shumard oak plantings to higher elevations that typically do not flood.

ACKNOWLEDGMENTS

We would like to thank the Division of Forestry of the Arkansas Department of Agriculture for donating the *Quercus* seedlings used in this experiment. Thanks also to Alyssa Sanders, Zachary Reed, Jason Lam, Will Tietz, Roberto Bernal, Lesly Jean-Francois, Colby Mohler, Ethan Russell, Jacob Schwantz, Gabby Sherman, Richard Vaerewyck, Dalton Weatherly, and Stacy Wilson for technical assistance. This project was supported in part by the National Science Foundation's Research Infrastructure Improvement Award OIA-1457888 the CASE project (Center for Advanced Surface Engineering) of the Arkansas EPSCoR Program, and by the U.S. Department of Agriculture, National Institute of Food and Agriculture, McIntire-Stennis project number 1009319.

LITERATURE CITED

Sample, R.; Babst, B.A. 2020. Autumn flooding disrupts seasonal nitrogen storage and impacts spring growth in *Quercus texana* seedlings. Trees. 34: 813–823. https://doi.org/10.1007/s00468-020-01960-5.

Sample, R.D.; Cook, J.; Babst, B.A. 2023. Resiliency of Nuttall oak but not Shumard oak to winter and spring flood: dormancy alone does not confer flood tolerance. Trees. 37: 1121–1136. https://doi.org/10.1007/s00468-023-02411-7.

THE RESPONSE OF WILLOW OAK (*QUERCUS PHELLOS*) SEEDLING ROOTS TO FALL ENVIRONMENTAL CUES AND WINTER FLOODING

Benjamin A. Babst, Jonathan M. Kressuk, Tucker Collins, Emile S. Gardiner, and Mohammad M. Bataineh

EXTENDED ABSTRACT

The timing of seasonal flooding in bottomlands relative to seedling dormancy is a crucial determinant of how flooding affects seedlings (Sample and Babst 2020, Sample and others 2023). Alterations to seasonal flood regimes likely plays a role in poor regeneration of bottomland red oak species such as willow oak (*Quercus phellos*) in the Lower Mississippi Alluvial Valley, especially in greentree reservoirs (GTRs). For example, natural waterlogging tends to occur, as well as sporadic flooding, from mid-December through May, and it is assumed that dormancy confers greater resistance to flooding (Broadfoot and Williston 1973, Guttery and Ezell 2006, Therrell and Bialecki 2015). However, artificial flooding is typically initiated earlier in GTRs, often in November, but in some cases as early as October (Guttery and Ezell 2006, Wigley and Filer 1989).

This study investigated 1) whether roots of willow oak seedlings enter dormancy during winter, 2) if cold soil temperatures and other environmental cues reduce root activity in willow oak during winter, and 3) if reduced root activity increased root tolerance to winter soil flooding. We employed an experimental approach to study first-year willow oak seedlings in a greenhouse and a correlative approach to sample first-year, naturally seeded willow oak seedlings in the field. In the greenhouse experiment, soil temperature of potted seedlings was manipulated using large water baths that maintained temperatures within ±0.5 °C of set points (fig. 1).

Benjamin A. Babst, Associate Professor, Arkansas Forest Resources Center, and University of Arkansas at Monticello, College of Forestry, Agriculture and Natural Resources, Monticello, AR 71656

Jonathan M. Kressuk, Graduate Student, University of Arkansas at Monticello, College of Forestry, Agriculture and Natural Resources, Monticello, AR 71656; Present address: Graduate Student, North Carolina State University, Raleigh, NC 27695

Tucker Collins, Graduate Student, University of Arkansas at Monticello, College of Forestry, Agriculture and Natural Resources, Monticello, AR 71656; Present address: Wildlife Habitat Biologist, Arkansas Game and Fish Commission, Jonesboro, AR 72404

Emile S. Gardiner, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Center for Bottomland Hardwoods Research, Stoneville, MS 38776

Mohammad M. Bataineh, Associate Professor, Arkansas Forest Resources Center, and University of Arkansas at Monticello, College of Forestry, Agriculture and Natural Resources, Monticello, AR 71656; Present address: Research Biological Scientist, U.S. Department of Agriculture, Forest Service, Southern Research Station, Pineville, LA 71360

Figure 1—Willow oak seedlings in soil temperature control tanks. Pots in leak-proof plastic bags were placed in tanks to maintain soil temperature within ±0.5 °C. (Photo courtesy of Benjamin A. Babst)

In the greenhouse and field, willow oak leaves were semipersistent during winter. Retention of green leaves through the winter implies that carbon fixation and carbohydrate supply to the roots continued through both studies. In the greenhouse, root growth and respiration decreased when soil temperature was lowered. However, at 10 °C, which is close to the average soil temperature in the Lower Mississippi Alluvial Valley, root growth continued at a moderate rate. This suggests that willow oak seedling roots may not enter dormancy during winter. Temperature effects on root respiration were direct and immediately reversible, i.e., the effects of temperature were not due to acclimation by the roots. Though soil temperature clearly had a strong influence on root respiration, respiration also decreased over the winter months if soil temperature was held constant at 15 °C. Hence, something other than soil temperature caused reduced root activity in the winter but not to the same extent as low soil temperatures.

In the field, willow oak seedlings exhibited decreased root respiration immediately following frost events, but respiration rebounded as air temperatures increased. Root respiration was much more variable in the field than in the greenhouse and was positively correlated to average soil temperature and maximum daily air temperature. Root respiration also had weaker inverse correlations with photosynthetically active radiation and quantum yield of photosystem II.

Soil flooding initiated in December immediately halted root growth but did not reduce root respiration. Survival and spring growth were high for seedlings in all treatments. Seedlings that experienced warmer winter soil temperatures showed greater morphological indication of flooding stress, which included presence of hypertrophied lenticels and adventitious root growth to the surface of floodwater.

Our results indicate that soil temperature has a major influence on root growth and respiration, but daytime high air temperatures may also be important for root respiration. Root growth could continue during typical winter soil temperatures in the Lower Mississippi Alluvial Valley. This active tissue growth could be more vulnerable to hypoxia, but flooding stress on willow oak seedlings appears relatively low during the winter even when soil temperature is not low enough to stop root growth. It is not clear whether willow oak seedlings that retain leaves will be at a greater risk of damage or mortality due to overtopping by flood waters, which is common in bottomland forests and GTRs.

Because fall and spring flooding have negative impacts on willow oak seedlings (Sample and Babst 2020, Sample and others 2023), waiting to flood GTRs until root activity declines in early to mid-winter, and draining flood waters in early spring may decrease the stress experienced by willow oaks in these forests.

ACKNOWLEDGMENTS

We would like to thank Shelley Griffin, Jimmy Cook, Parikrama Sapkota, Joshua Forrest, Jacob Schwantz, and Chuck Walker for technical assistance. This project and preparation of the manuscript were funded in part by the Pittman-Robertson Wildlife Restoration Program (Grant #AR-W-F l4AF00290) of the U.S. Fish and Wildlife Service through an agreement with the Arkansas Game and Fish Commission, and by the U.S. Department of Agriculture, National Institute of Food and Agriculture, McIntire-Stennis project number 1009319.

LITERATURE CITED

Broadfoot, W.M.; Williston, H.L. 1973. Flooding effects on southern forests. Journal of Forestry. 71: 584–587.

Guttery, M.R.; Ezell, A.W. 2006. Characteristics of a bottomland hardwood forest under greentree reservoir management in east central Arkansas. In: Connor, KF., 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: 409–411.

Sample, R.; Babst, B.A. 2020. Autumn flooding disrupts seasonal nitrogen storage and impacts spring growth in *Quercus texana* seedlings. Trees. 34: 813–823. https://doi.org/10.1007/s00468-020-01960-5.

Sample, R.D.; Cook, J.; Babst, B.A. 2023. Resiliency of Nuttall oak but not Shumard oak to winter and spring flood: dormancy alone does not confer flood tolerance. Trees. 37: 1121–1136. https://doi.org/10.1007/s00468-023-02411-7.

Therrell, M.D.; Bialecki, M.B. 2015. A multi-century tree-ring record of spring flooding on the Mississippi River. Journal of Hydrology. 529: 490–498. https://doi.org/10.1016/j.jhydrol.2014.11.005.

Wigley, T.B., Jr.; Filer, T.H., Jr. 1989. Characteristics of greentree reservoirs: a survey of managers. Wildlife Society Bulletin. 17: 136–142.

FACTORS AFFECTING NATURAL OAK REGENERATION IN BOTTOMLAND HARDWOOD FORESTS IN NORTHERN MISSOURI: LIGHT LEVELS, ACORN LOSSES, OR LACK OF ACORNS?

Isaac Hayford and Benjamin O. Knapp

ABSTRACT

Successful oak (*Quercus* spp.) regeneration in bottomland forests can be limited by several factors, including number of acorns produced, acorn viability and predation, microsite conditions for germination, and early seedling survival. These factors could influence the development of large oak advance reproduction needed for successful oak regeneration. We designed an experiment to test the importance of some of these factors on oak regeneration in bottomland forests in northern Missouri. We found that low acorn production and high rates of acorn predation were prevalent throughout the stands. Buried acorns showed higher germination rates than acorns placed on the surface of the soil. Our results also indicated that germination of acorns was unaffected by light availability. Thus, our findings suggest that midstory release will not increase germination and therefore should not be expected to increase the initial establishment of new seedlings. However, midstory release could potentially be useful in enhancing development of seedlings following germination.

INTRODUCTION

Natural regeneration is regarded as a more cost-effective option than artificial means of forest establishment (Löf and others 2019). The process of natural regeneration involves several stages that include seed production, seed dispersal, seed germination, seedling survival, and seedling growth or recruitment into the overstory. The success of natural regeneration is contingent on the suitability of species-specific environmental conditions, including site factors and disturbance regimes. Failure at any stage of the regeneration process can result in failure in the regeneration of a species. Understanding species-specific environmental and regeneration needs is important to successfully manage them (Grubb 1977).

Acorn production varies greatly between years and among species, individuals, and locations (Rose and others 2012). The interannual variation in acorn production is usually associated with synchronous acorn production among individuals within a population. However, acorn production levels could also fluctuate depending on the number of fruiting trees or the number of fruits per tree (Greenberg and Parresol 2000). There are contrasting findings regarding the influence of age and size of trees on acorn production. Goodrum and others (1971) found that intrinsic features such as age and size affect acorn production whilst Greenberg (2000) found that measurable tree characteristics such as size do not provide sufficient information to identify individual trees of superior acorn production ability. Other factors, including stand density, genetics, and weather events such as late spring freezes or heavy precipitation that interferes with oak flower survival or pollen dispersal, have been implicated to affect acorn production (Healy 1997, Johnson and others 2002).

Acorns are an important source of food for a wide range of animals including insects, birds, and mammals such as rodents, suids, and cervids due

Isaac Hayford, Assistant Professor, Department of Agriculture and Environmental Sciences, Lincoln University of Missouri, Jefferson City, MO 65101

Benjamin O. Knapp, Associate Professor, School of Natural Resources, University of Missouri, Columbia, MO 65211

to their palatability, nutritiousness, widespread occurrence, and availability during the critical fall and winter seasons (Janzen 1971, Johnson 1994). White-tailed deer (*Odocoileus virginianus*), eastern chipmunks (*Tamias striatus*), and mice (*Mus* spp.) consume a large percentage of fallen acorns on the ground, whereas tree squirrels (*Sciurus* spp.) consume them both on the ground and before they fall (Haas and Heske 2005). Acorn predation by vertebrates or insects could destroy up to 90 percent of acorns, leading to substantial reduction in oak reproduction (Crawley and Long 1995). Masting by oaks is a defensive strategy that allows a higher proportion of acorns to escape predation by satiating acorn predators with excess during mast years and reducing predators during nonmast years (Janzen 1971, Kelly and Sork 2002). This phenomenon, termed as the predator satiation hypothesis, has been demonstrated by studies where relatively lower rates of predation were recorded in high mast production years as opposed to higher rates of predation in low mast production years (Goodrum and others 1971, Kelly and Sork 2002).

Species vary greatly in their requirements for germination, with external factors such as temperature, substrate, moisture, or light as possible germination cues. Plants can be classified into three groups based on their response to light for germination, and these include those that require light to germinate, those that require shade to germinate, and those that have a large percentage of their seeds neutral to light (Flores and others 2016). Acorn germination is not constrained by light availability and can germinate even under a closed canopy (Dey 2002, Moorhead and others 1991). The position of acorns in the soil has also been shown to affect rates of acorn germination. Previous studies have reported higher germination rates for buried acorns compared to acorns placed on top of the soil (Guo and others 2009, Nilsson and others 1996). Acorns require variable times of cold stratification to enhance their germination rates. Pin oak (*Quercus palustris*) acorns, for example, require stratification of 30–45 days at 0–5 °C to break dormancy (Burns and Honkala 1990).

Because successful oak regeneration is dependent upon an adequate number of large oak advance reproduction, poor seed crops, high rates of acorn predation, and poor germination can have substantial impacts on the ability of oaks to compete with other species. Most mature oak stands usually have several hundred to thousands of oak seedlings per hectare present on the forest floor (Hayford and others 2023). The absence of adequate oak seedlings on the forest floor in our study sites was particularly worrying in the quest of maintaining or increasing the abundance of oaks. The main goal of this study was to investigate the cause of natural regeneration failures of bottomland oaks in our study sites. Was it due to inadequate acorn production, acorn losses through acorn predation, the effect of light regime, or the effect of the position of acorns in the soil on acorn germination? The specific objectives were to determine 1) the effect of oak density on acorn production; 2) the effects of the position of the acorn in the soil (buried versus not buried), acorn predator exclosures, and acorn production level (low, medium, high) on acorn removal; and 3) the effects of the position of the acorn in the soil (buried versus not buried) and light levels on acorn germination.

We hypothesized that acorn predation would be higher in the open plots than in plots that restricted access to small mammals like squirrels and mice, and in complete exclosure plots that excluded all mammals. It was expected that the complete exclosure plots would restrict access to all known acorn predators and hence retain all or most acorns placed in these plots to ensure higher germination rates compared to the other experimental plots. We also hypothesized that buried acorns would have higher germination rates than acorns that were not buried. We were not expecting acorn germination to be different among the light levels.

METHODS

Study Site

The study was conducted on a 220-ha mixed bottomland hardwood forest at Deer Ridge Conservation Area (DRCA) (40°10’30”N, 91°48’0”W) in Lewis County of northeastern Missouri. DRCA has one of the largest tracts of remnant bottomland forest in northeastern Missouri and is adjacent to a channelized segment

of the North Fabius River. This site contained contiguous bottomland forest blocks >20 ha, and State ownership facilitated long-term access and control of management activities. The soils of the study area were dominated by Blackoar silt loam (fine-silty, mixed, superactive, mesic Fluvaquentic Hapludolls), Fatima silt loam (fine-silty, mixed, superactive, mesic Fluvaquentic Hapludolls), and Kickapoo fine sandy loam (coarse-loamy, mixed, superactive, nonacid, mesic Typic Udifluvents); each were formed in alluvium but varied in drainage class. All soil series were associated with slopes between 0 and 2 degrees. The site was prone to annual flooding events, with patches of low-lying areas permanently inundated with water. The mean annual temperature and precipitation for the area over a 30-year period (1991–2020) were 11.7 °C and 1,007 mm, respectively (PRISM Climate Group, www.prism.oregonstate.edu). The tree species in the area were predominantly silver maple (*Acer saccharinum*), American elm (*Ulmus americana*), American sycamore (*Platanus occidentalis*), green ash (*Fraxinus pennsylvanica*), boxelder (*Acer negundo*), eastern cottonwood (*Populus deltoides*), pin oak, bur oak (*Quercus macrocarpa*), and swamp white oak (*Quercus bicolor*) (Rives and others 2020).

Experimental Design

All treatments were established in stands with homogenous hydrology. This study used a factorial split-split-plot design. The whole plot treatments included three levels of light availability, and the split-plot treatments were a factorial combination of three levels of acorn predator exclosures and three levels of acorn production. The position of acorns in the soil (buried versus not buried) was the split-split-plot factor.

The three light levels were:

- High light—a uniform, area-based approach was used to reduce basal area of midstory trees in a previously unharvested stand by 65 percent, leaving an understory light level of 18 percent of full sunlight.
- Medium light—a uniform, area-based approach was used to reduce basal area of midstory trees in a previously unharvested stand by 30 percent, leaving an understory light level of 15 percent of full sunlight.
- Low light—no treatment was applied and light level was approximately 5 percent of full sunlight.

The three levels of acorn predator exclosures were installed in August 2020. Each of the acorn predator exclosure experimental plots was 1.52 m by 0.61 m. Exclosures were situated 1 m from each other.

- Complete exclosures prohibited all known acorn predators such as mice, squirrels, and deer from access to the plot. The exclosures consisted of 0.01-m mesh galvanized steel hardware cloth. The exclosure walls were 0.61 m tall, covered at the top, and buried 0.15 m under the soil surface to prevent small mammals from burrowing under the walls.
- Squirrel-mouse plot exclosures restricted access by vertebrates larger than squirrels such as deer. They were constructed with the same wire mesh as complete exclosures, but the hardware cloth was raised 0.15 m above ground level to allow access by mice and squirrels.
- Open plot controls were marked by four rebars where small and large predators had unrestricted access to the experimental plot.

In a 14-year study by McQuilkin and Musbach (1977), the highest number of mature acorns produced in a particular year in 32- to 46-year-old pin oak stands in southeastern Missouri was 492,700/ha, and hence we selected 40 acorns to represent high production in a 0.92 m^2 area. Buried acorns were intended to simulate caches of gray squirrels (*Sciurus carolinensis*). Caches were made by depositing acorns horizontally into a 0.02-m-deep depression and covering them with soil and litter. In the fall of 2020, each of the exclosure and control treatments received one of the following acorn production and position treatments:

- High acorn production—20 acorns were buried halfway in simulated caches in a 4 by 5 grid with each acorn spaced 0.15 m apart. An additional 20 acorns were marked with blue paint and deposited on the soil surface underneath the leaf litter.

- Medium acorn production—15 acorns were buried halfway in simulated caches in a 3 by 5 grid with each acorn spaced 0.15 m apart. An additional 15 acorns were marked with blue paint and placed on the soil surface underneath the leaf litter.
- Low acorn production—10 acorns were buried halfway in simulated caches in a 2 by 5 grid with each acorn spaced 0.15 m apart. An additional 10 acorns were marked with blue paint and placed on the soil surface underneath the leaf litter.

The nine treatment combinations were replicated three times in each of the three light levels for a total of 81 experimental units (fig. 1). The pin oak acorns were purchased from the George O. White State Forest Nursery in Licking, MO. The acorns were collected during the preceding autumn and screened for soundness, stratified, and stored in accordance with standard nursery practices. Acorns were refrigerated after receipt from the nursery for about 1 month prior to field deployment to prevent them from heating up or drying out and to break dormancy. Prior to refrigeration, the acorns meant to be placed on the soil surface were marked with blue paint to differentiate them from those to be buried.

Acorn Crop Measurement

Sixteen wire mesh cone traps, 0.42 m^2 in size, were arrayed in a 4 by 4 grid in each of the nine plots where the acorn exclosures were constructed, with traps spaced 10 m apart (144 traps total, 60.48 m^2 total sample area). This approach was chosen rather than collecting acorns from individual trees because of the variability in acorn production among individuals and the difficulty associated with collection from individuals under closed canopy conditions. The traps did not restrict access to predators.

Data Collection

The remaining acorns or seedlings were counted 8, 21, 43, and 307 days after placement in the exclosure plots on November 4, 2020. We found no acorn germination until the 307th day check. Acorn traps were checked at approximately 2-week intervals from late August until the

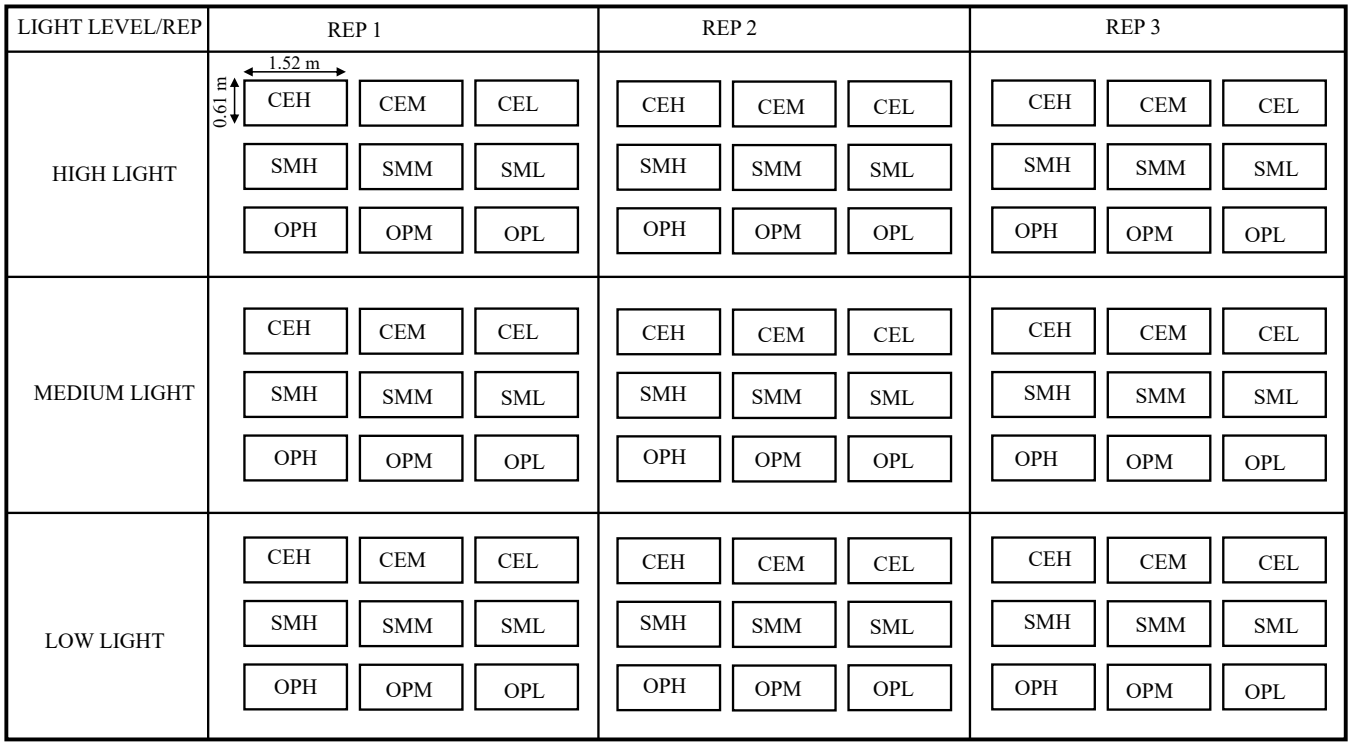

Figure 1—Field layout of nine acorn predator exclosure experimental units replicated three times within each of three light levels. CE = complete exclosure; SM = squirrel-mouse plot; OP = open plot; H = high acorn production; M = medium acorn production; and L = low acorn production.

completion of acorn drop in mid-December in both 2020 and 2021. All mature acorns collected were identified to species, counted, and removed during each visit. We counted all well-developed acorns without distinguishing between insect-damaged and sound acorns or between different acorn sizes.

Data Analysis

We did not have enough data to test the relationship between oak density and acorn production due to low acorn production. All the acorns in the open and squirrel-mouse plots were completely removed within 8 to 21 days after acorn placement in exclosures (table 1). As a result, acorn removal rates could only be tested in the complete exclosure plots. We used logistic regression to test the effects of acorn production level (low, medium, high) and acorn burial status (buried, nonburied) on acorn removal within the complete exclosure plots. We used a generalized linear mixed model with a binomial distribution, a logit link function, and a random intercept that used stand as the subject, using GLIMMIX procedure of SAS (version 9.3, SAS Institute, Inc., Cary, NC). We could not include days of acorn removal in the model because that created convergence issues. Thus, we pooled all data across time periods. The same analysis framework was used to test the effects of light level, acorn burial status, and their interactions on acorn germination at the day 307 sampling. Each acorn that germinated was coded as 1 and nongerminated acorns coded as 0. Germination analysis was conducted on only remnant acorns (i.e., acorns that were not removed by predators) in the complete exclosure plots because all acorns in the open and squirrel-mouse plots were removed by predators. When significant differences were detected among treatments, Tukey's honestly significant difference adjustment was used to evaluate pairwise comparisons.

RESULTS

In total, 2,430 acorns were monitored for removal by predators during the 1-year study period within the exclosure plots. All acorns (buried and not buried) were removed from the open and squirrel-mouse plots by the 21st day following placement in the exclosures. For the complete exclosure plots, the proportion of acorns removed was 0.39 by the last sample date (table 1). In most cases, the predators could not access the acorns in the complete exclosure plots. However, in the few instances where the predators were able to access the complete exclosure plots, probably due to faulty construction, most or all acorns were removed (fig. 2). The simulated acorn production level affected the rate of acorn removal within the complete exclosure plots (P = 0.02, fig. 3). The rate of acorn removal was lower within the complete

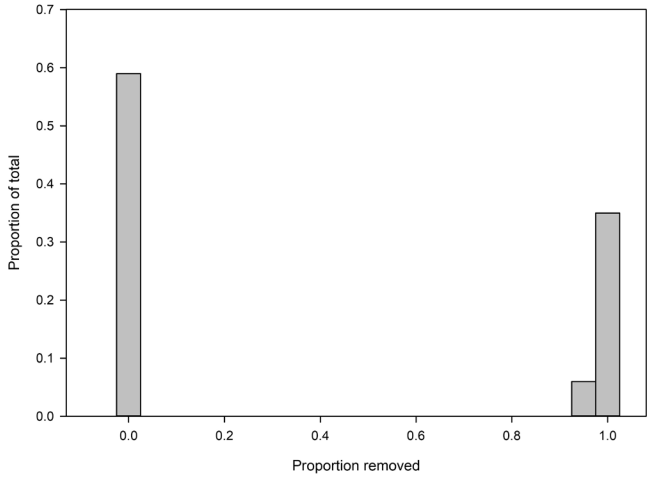

Figure 2—Histogram showing the rate of acorn removal within the complete exclosure plots at final day (day 307) of monitoring at Deer Ridge Conservation Area, Lewis County, MO.

Table 1—Proportion of acorns removed from predator exclosures by number of days

	Not buried acorns			Buried acorns		
Days	Open	Squirrel-mouse	Complete	Open	Squirrel-mouse	Complete
0	0.00	0.00	0.00	0.00	0.00	0.00
8	1.00	0.99	0.14	0.99	0.97	0.10
21	1.00	1.00	0.37	1.00	1.00	0.36
43	1.00	1.00	0.37	1.00	1.00	0.37
307	1.00	1.00	0.39	1.00	1.00	0.39

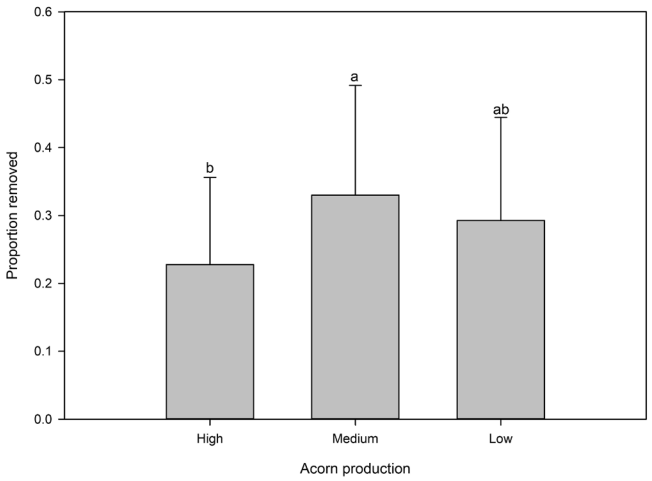

Figure 3—Acorn production level effect on average proportion of acorns removed within the complete exclosure plots at Deer Ridge Conservation Area, Lewis County, MO, 2020–2021. Different letters indicate significant differences among acorn production levels. Error bars represent one standard error from the average.

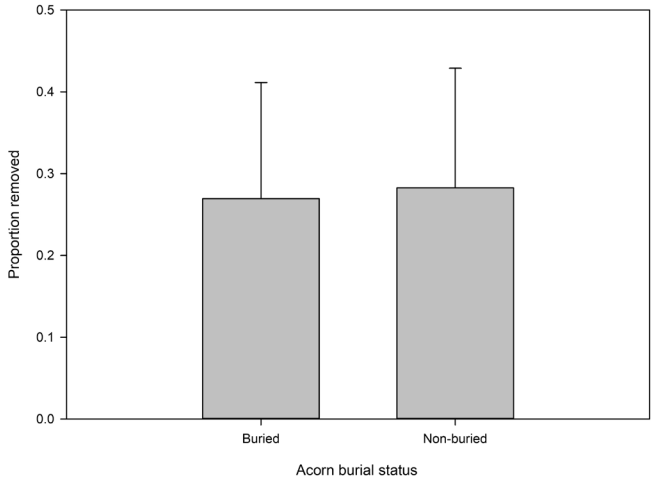

Figure 4—Average proportion of acorns removed by predators from complete exclosure plots by burial status at Deer Ridge Conservation Area, Lewis County, MO, 2020–2021. Error bars represent standard error from the average.

exclosure plots that received high acorn production than those that received medium acorn production. The rate of acorn removal within the low acorn production level treatment, however, was not different from the medium and high acorn production. There was no significant difference in proportion of removal between acorns that were buried and not buried in the complete exclosure plots (P = 0.52, fig. 4).

There was a significant effect of acorn burial status on proportion of germination (P = 0.03, fig. 5), where greater proportion of buried acorns that remained in the complete exclosure plots germinated than acorns that were not buried. Germination of acorns was not different among the three light levels (P = 0.59, fig. 5). The mean proportion of germination across the three light levels was 0.32. We did not have sufficient data to test the relationship between oak density and acorn production due to low acorn production. In total, we found only 54 acorns during the 2-year study period, and these were found in 3 of 144 acorn traps, and in 1 of 9 plots.

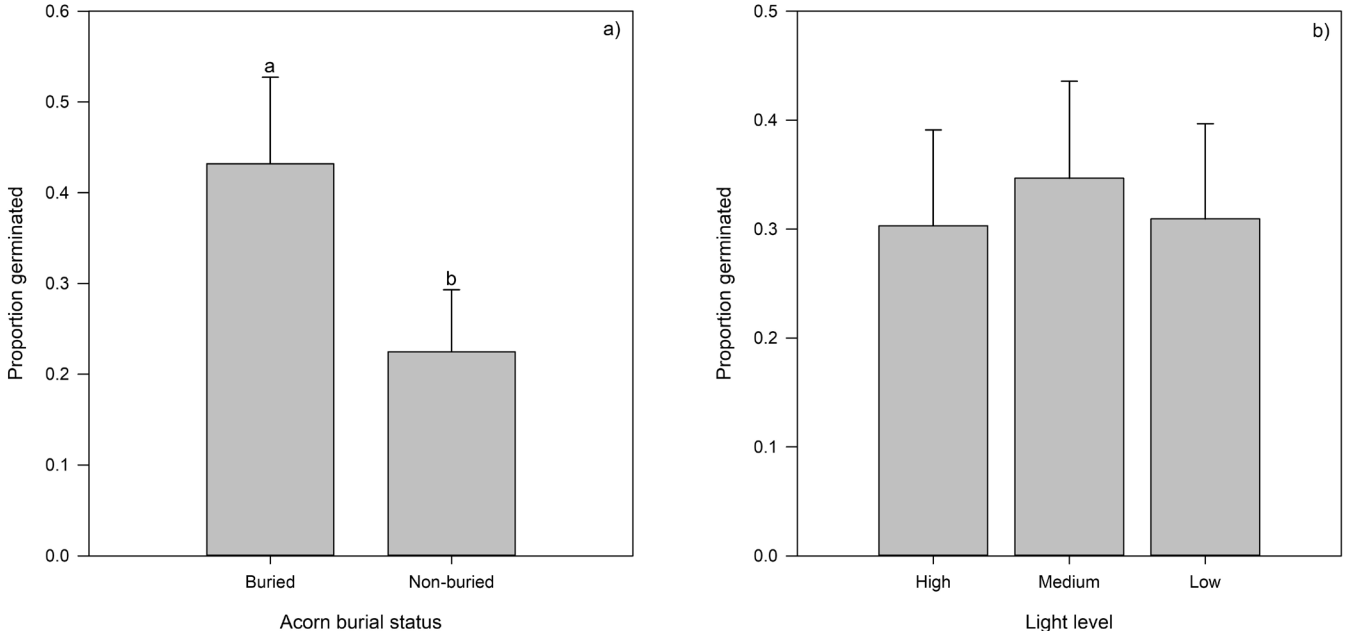

Figure 5—Average proportion of acorns that germinated by (a) burial status and (b) light level at Deer Ridge Conservation Area, Lewis County, MO, 2020–2021. Error bars represent standard error from the average. Different letters indicate significant differences according to pairwise comparisons.

DISCUSSION

Results indicated low acorn production and high acorn predation. Acorn removal was extremely rapid, with complete removals within 8 to 21 days after acorn placement in the open plots that allowed access by all mammals, and squirrel-mouse plots that allowed access by small mammals. Although we did not find any bottomland studies with this finding, some upland studies in China have reported instances where acorns deployed on the ground were rapidly removed within a few hours to a few days (Li and Zhang 2003, Wang and others 1999). In our study, low acorn production might have contributed to rapid removal rates within the open and squirrel-mouse experimental plots. Our results also suggest that small mammals, rather than deer, were probably the most important acorn predators. Other studies have reported that large animals like deer do not substantially contribute to acorn predation because mice, squirrels, and other small-sized animals compensate for their effect when they are excluded (Borchert and others 1989, Greenberg and Zarnoch 2018). Acorns removed by deer are consumed and have no further chance of germination, whereas acorns moved by squirrels and other small animals may be cached for later germination (Kellner and others 2014).

We found that acorn removal did not differ between buried and nonburied acorns. Gómez and others (2003) reported that burial did not increase acorn survival from predation. This probably occurred because acorns were consumed by several predators that could detect buried acorns with very high efficiency. Other studies have reported that burying acorns reduced acorn predation (Haas and Heske 2005, Kollmann and Schill 1996). However, the acorns were buried much deeper in those studies (burial depth ranged from 2 to 8 cm) than in our study, where acorns were buried halfway, suggesting that caching by animals may protect acorns from further predation.

Our results indicated that acorn burial improved germination. Although Barnett (1977) reported that burial of white oak (*Quercus alba*) acorns had no effect on germination, other studies have found that acorn burial increased germination (Guo and others 2009, Haas and Heske 2005, Kollmann and Schill 1996, Nilsson and others 1996). Buried acorns are exposed to a moist soil environment that prevents seed desiccation and may reduce damage from drying, intense sunlight, low temperature, or fungi (Kollmann and Schill 1996, Nilsson and others 1996).

Germination of acorns was unaffected by light availability. Acorn germination is not constrained by light availability because the seed is relatively large and supplies most of the carbohydrates for growth until seed reserves are depleted (Dey 2002). According to Moorhead and others (1991) acorns can germinate under a closed canopy. Since the midstory release treatments used in our study did not increase germination, they should not be expected to increase the establishment of new seedlings. However, after germination, the midstory release could be used to enhance development of the seedlings as reported in other bottomland studies (Lockhart and others 2000).

CONCLUSIONS

Our results suggested that predators have an important influence on the proportion of acorns that eventually have a chance of germination. We found that removal of acorns was extremely rapid if small mammals had access to acorns. The low acorn production in our stands might have contributed to rapid acorn removal rates observed due to a lack of acorn availability in the area. Our results also suggested that burial provided the necessary microclimate that enhanced germination of remnant acorns compared to surface acorns that might be exposed to damage by drying, intense sunlight, low temperature, or fungi. We found that germination occurred irrespective of light availability. Our results suggest the importance of mast years, in which high levels of acorn production may be necessary to satiate small mammal predation, for natural regeneration of pin oak in bottomland forests. While midstory release treatments may not be expected to increase germination rates, the greater light availability provided through well-timed release may improve early seedling

survival and growth following germination. Alternatively, due to the low acorn production and high rates of acorn predation in our study sites, artificial regeneration might be useful for establishing seedlings rather than relying on natural regeneration.

ACKNOWLEDGMENTS

We gratefully acknowledge Missouri Department of Conservation for funding this project. We thank Trystan Harpold, Jacob Worsham, Hope Fillingim, and Ian Vaughan for assisting with fieldwork for this study, as well as numerous other individuals for contributions to project establishment and maintenance over the years.

LITERATURE CITED

Barnett, R.J. 1977. The effect of burial by squirrels on germination and survival of oak and hickory nuts. American Midland Naturalist. 98: 319–330. https://doi.org/10.2307/2424983.

Borchert, M.I.; Davis, F.W.; Michaelsen, J.; Oyler, L.D. 1989. Interactions of factors affecting seedling recruitment of blue oak (*Quercus douglasii*) in California. Ecology. 70: 389–404. https://doi.org/10.2307/1937544.

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

Crawley, M.J.; Long, C.R. 1995. Alternate bearing, predator satiation and seedling recruitment in *Quercus robur* L. Journal of Ecology. 83: 683–696. https://doi.org/10.2307/2261636.

Dey, D. 2002. The ecological basis for oak silviculture in eastern North America. In: McShea, W.J.; Healy, W.M., eds. Oak forest ecosystems. Baltimore, MD: John Hopkins University Press: 60–79.

Flores, J.; González-Salvatierra, C.; Jurado, E. 2016. Effect of light on seed germination and seedling shape of succulent species from Mexico. Journal of Plant Ecology. 9: 174–179. https://doi.org/10.1093/jpe/rtv046.

Gómez, J.M.; García, D.; Zamora, R. 2003. Impact of vertebrate acorn- and seedling-predators on a Mediterranean *Quercus pyrenaica* forest. Forest Ecology and Management. 180: 125–134. https://doi.org/10.1016/S0378-1127(02)00608-4

Goodrum, P.D.; Reid, V.H.; Boyd, C.E. 1971. Acorn yields, characteristics, and management criteria of oaks for wildlife. Journal of Wildlife Management. 35: 520–532. https://doi.org/10.2307/3799707.

Greenberg, C.H. 2000. Individual variation in acorn production by five species of southern Appalachian oaks. Forest Ecology and Management. 132: 199–210. https://doi.org/10.1016/S0378-1127(99)00226-1.

Greenberg, C.H.; Parresol, B.R. 2000. Acorn production characteristics of southern Appalachian oaks: a simple method to predict within-year acorn crop size. Res. Pap. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 14 p.

Greenberg, C.H.; Zarnoch, S.J. 2018. A test of the predator satiation hypothesis, acorn predator size, and acorn preference. Canadian Journal of Forest Research. 48: 237–245. https://doi.org/10.1139/cjfr-2017-0381.

Grubb, P.J. 1977. The maintenance of species-richness in plant communities: the importance of the regeneration niche. Biological Reviews. 52: 107–145. https://doi.org/10.1111/j.1469-185x.1977.tb01347.x.

Guo, C.-R.; Lu, J.-Q.; Yang, D.-Z.; Zhao, L.-P. 2009. Impacts of burial and insect infection on germination and seedling growth of acorns of *Quercus variabilis*. Forest Ecology and Management. 258: 1497–1502. https://doi.org/10.1016/j.foreco.2009.05.044.

Haas, J.P.; Heske, E.J. 2005. Experimental study of the effects of mammalian acorn predators on red oak acorn survival and germination. Journal of Mammalogy. 86: 1015–1021. https://doi.org/10.1644/1545-1542(2005)86[1015:esoteo]2.0.co;2.

Hayford, I.; Knapp, B.O.; Kabrick, J.M. 2023. Responses of natural and artificial pin oak (*Quercus palustris*) reproduction to a sequence of silvicultural release treatments in bottomland hardwood forests in southern Missouri. Forest Ecology and Management. 529: 120699. 11 p. https://doi.org/10.1016/j.foreco.2022.120699.

Healy, W.M. 1997. Thinning New England oak stands to enhance acorn production. Northern Journal of Applied Forestry. 14: 152–156. https://doi.org/10.1093/njaf/14.3.152.

Janzen, D.H. 1971. Seed predation by animals. Annual Review of Ecology and Systematics. 2: 465–492. https://doi.org/10.1146/annurev.es.02.110171.002341.

Johnson, P.S. 1994. How to manage oak forests for acorn production. TB-NC-1. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. https://www.nrs.fs.usda.gov/pubs/tb/tb1/techbrf1.html.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2002. The ecology and silviculture of oaks. Cambridge, MA: CABI Publishing. 503 p.

Kellner, K.F.; Riegel, J.K.; Swihart, R.K. 2014. Effects of silvicultural disturbance on acorn infestation and removal. New Forests. 45: 265–281. https://doi.org/10.1007/s11056-014-9409-9.

Kelly, D.; Sork, V.L. 2002. Mast seeding in perennial plants: why, how, where? Annual Review of Ecology and Systematics. 33: 427–447. https://doi.org/10.1146/annurev.ecolsys.33.020602.095433.

Kollmann, J.; Schill, H.-P. 1996. Spatial patterns of dispersal, seed predation and germination during colonization of abandoned grassland by *Quercus petraea* and *Corylus avellana*. Vegetatio. 125: 193–205. https://doi.org/10.1007/BF00044651.

Li, H.-J.; Zhang, Z.-B. 2003. Effect of rodents on acorn dispersal and survival of the Liaodong oak (*Quercus liaotungensis* Koidz.). Forest Ecology and Management. 176: 387–396. https://doi.org/10.1016/S0378-1127(02)00286-4.

Lockhart, B.R.; Hodges, J.D.; Gardiner, E.S. 2000. Response of advance cherrybark oak reproduction to midstory removal and shoot clipping. Southern Journal of Applied Forestry. 24: 45–50. https://doi.org/10.1093/sjaf/24.1.45.

Löf, M.; Castro, J.; Engman, M. [and others]. 2019. Direct seeding to restore oak (*Quercus* spp.) forests and woodlands. Forest Ecology and Management. 448: 474–489. https://doi.org/10.1016/j.foreco.2019.06.032.

McQuilkin, R.A.; Musbach, R.A. 1977. Pin oak acorn production on green tree reservoirs in southeastern Missouri. Journal of Wildlife Management. 41: 218–225. https://doi.org/10.2307/3800598.

Moorhead, D.J.; Hodges, J.D.; Reinecke, K.J. 1991. Silvicultural options for waterfowl management in bottomland hardwood stands and greentree reservoirs. In: Coleman, S.S.; Neary, D.G., comps. and eds. Proceedings of the sixth biennial southern silvicultural research conference. Gen. Tech. Rep. SE-70. Vol. 2. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 710–721.

Nilsson, U.; Gemmel, P.; Löf, M., Welander, T. 1996. Germination and early growth of sown *Quercus robur* L. in relation to soil preparation, sowing depths and prevention against predation. New Forests. 12: 69–86. https://doi.org/10.1007/BF00029983.

Rives, R.G.; Knapp, B.O.; Olson, M.G. [and others]. 2020. Regenerating mixed bottomland hardwood forests in north Missouri: effects of harvest treatment on structure, composition, and growth through 15 years. Forest Ecology and Management. 475: 118371. 9 p. https://doi.org/10.1016/j.foreco.2020.118371.

Rose, A.K.; Greenberg, C.H.; Fearer, T.M. 2012. Acorn production prediction models for five common oak species of the eastern United States. Journal of Wildlife Management. 76: 750–758. https://doi.org/10.1002/jwmg.291.

Wang, W.; Ma, K.; Liu, C. 1999. Removal and predation of *Quercus liaotungensis* acorns by animals. Ecological Research. 14: 225–232. https://doi.org/10.1046/j.1440-1703.1999.143297.x.

LEAF AREA INDEX ACROSS EASTERN COTTONWOOD AND HYBRID POPLARS AND ITS RELATIONSHIP WITH PRODUCTIVITY AND WATER USE EFFICIENCY UNDER COPPICING AND NONCOPPICING MANAGEMENT IN THE SOUTHEASTERN UNITED STATES

Jiaxin Wang and Heidi J. Renninger

EXTENDED ABSTRACT

Populus is a promising genus for bioenergy production due to its fast growth. Using efficient and productive *Populus* genotypes is critical for sustainable bioenergy production. However, breeding and testing different genotypes can be costly and time consuming, and the results may vary based on environmental conditions. Therefore, using easily measurable leaf traits to predict *Populus* water use efficiency and productivity could lead to a faster and more accurate selection of top genotypes.

To estimate eastern cottonwood (*Populus deltoides*) and hybrid poplar growth performance, we planted 167 unique clones of *P. deltoides* and five interspecific hybrid taxa of *Populus* at two sites in the Southeastern United States. Leaf area index (LAI) of individual trees was measured across the growing season from April to November. Leaf gas exchange was measured during the middle of the growing season. Tree height and diameter at breast height were measured yearly at the end of the growing seasons. We used analysis of variance, linear regression, and machine learning models to explore the difference in LAI, photosynthetic capacity, intrinsic water use efficiency (iWUE), productivity (yearly added dry biomass), and the relationships between different taxa.

The sum of LAI of each tree (Sum_LAI) measured from April to November varied significantly across different taxa. *Populus deltoides* × *P. deltoides* (D×D) genotypes had significantly higher (10–150 percent higher) Sum_LAI than other taxa. *Populus deltoides* × *P. nigra* (D×N) genotypes had the lowest Sum_LAI (2.0 m^2 m^{-2}). *Populus trichocarpa* × *P. maximowiczii* (T×M) genotypes exhibited the highest iWUE (as high as 58 µmol mol^{-1}), and D×D

Jiaxin Wang, Graduate Student, Mississippi State University, Department of Forestry, Mississippi State, MS 39762

Heidi J. Renninger, Associate Professor, Mississippi State University, Department of Forestry, Mississippi State, MS 39762

exhibited the lowest iWUE (as low as 15 μmol mol^{-1}). D×D had the highest biomass (up to 9.0 kg/tree at age 3), and T×M genotypes had the lowest biomass (as low as 2.0 kg/tree at age 3).

Coppicing management (every 2 years after age 2) significantly reduced Sum_LAI, photosynthetic capacity, and productivity compared to noncoppiced management. Weighted linear regression models suggested that Sum_LAI, stomatal conductance, and photosynthetic rate explained >80 percent of the variance in productivity and >90 percent in iWUE under noncoppiced and coppiced management (different models were fitted for noncoppiced and coppiced management).

These findings can be used for future breeding and quicker phenotyping of seedlings for productivity and physiological traits to select *Populus* taxa and genotypes for efficient bioenergy production in the Southeastern United States.

PLANTED WHITE OAK (*QUERCUS ALBA*) RESPONSE TO SPACING DENSITY AND ORIENTATION IN GAP OPENINGS IN THE BLUE RIDGE MOUNTAINS OF NORTH CAROLINA

Stacy L. Clark, Tara Keyser, Scott E. Schlarbaum, and Arnold Saxton

EXTENDED ABSTRACT

White oak (*Quercus alba*) trees are difficult to sustain in many stands across the Central Hardwood Region because natural and anthropogenic processes are failing to establish advance reproduction (Schweitzer and others 2019). Large-scale shifts in species composition result in mesophication of hardwood forests that can lead to lower biodiversity and reduced fire and drought tolerance (Woodbridge and others 2022). Artificial regeneration (i.e., planting) contributes to localized restoration targets with an immediate input of advance reproduction (Clark and Schlarbaum 2019). The goal of our study was to determine how white oak seedlings respond to the effects of planting density and planting orientation in an irregular shelterwood.

We implemented the study in the Blue Ridge ecoregion on the Pisgah National Forest in North Carolina to examine artificial regeneration responses within an uneven-aged irregular shelterwood. The shelterwood was initially implemented as 0.25-acre or 1.0-acre gaps through commercial harvesting in 2019. The gaps will be expanded in approximately 10-year intervals on an area basis to create multiple age classes. Planted seedlings were bareroot (1-0) from 10 pedigreed sources in eastern Tennessee and were planted in six of the 0.25-acre gaps. Trees were randomly assigned into treatments consisting of planting spacing (2 feet by 2 feet or 12 feet by 12 feet), gap location (center, edge, forest matrix), and gap direction from gap center (north or south). Groups of 10 to 30 trees were planted within each gap along the north and south axis extending from gap center at the three gap locations: center (2 to 13 feet from gap center), edge (6 feet inside and outside of the gap edge), and forest matrix (20 to 30 feet outside the gap edge). The 2-foot-by-2-foot-spacing trees were planted in five rows of six trees each, and the 12-foot-by-12-foot-spacing trees were planted in two rows of five trees for each treatment combination. Across all treatments, tree height ranged from 6 to 34 inches

Stacy L. Clark, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Knoxville, TN 37996

Tara Keyser, Center Director, U.S. Department of Agriculture, Forest Service, Southern Research Station, Asheville, NC 28806

Scott E. Schlarbaum, Professor, The University of Tennessee, School of Natural Resources, Knoxville, TN 37996

Arnold Saxton, Professor Emeritus, The University of Tennessee, Department of Animal Science, Knoxville, TN 37996

with an average of 13.1 inches at the time of planting. Trees were measured for survival and height after the third growing season. Data were analyzed using general linear mixed modelling to test fixed effects of spacing (2 feet versus 12 feet), direction from gap center (north versus south), and location within the gap (center, edge, and matrix). Spacing was the whole plot treatment, and direction and location were subplot treatments; replication (n = 3) was a random effect in the model. A 0.05 α-level was used to denote significant differences.

Third year survival was 77 percent across all gaps, and treatments ($F_{1,4} \leq 1.51$, $P \geq 0.29$) and interactions ($F_{2,16} \leq 1.64$, $P \geq 0.23$) were not significant effects. Seedlings planted in gap centers were 2.5 feet tall after three growing seasons, 0.9 to 1.0 feet taller than trees on the gap edge or in the forest matrix, respectively ($F_{2,16} = 20.1$, $P < 0.0001$) (fig. 1). The effect of spacing ($F_{1,4} = 0.10$, $P = 0.77$) and direction ($F_{1,4} = 0.60$, $P = 0.48$) did not significantly affect height, and interactions were not significant ($F_{2,16} \geq 1.38$, $P \geq 0.27$). Deer browsing on terminal stems was negligible (<3 percent of trees browsed).

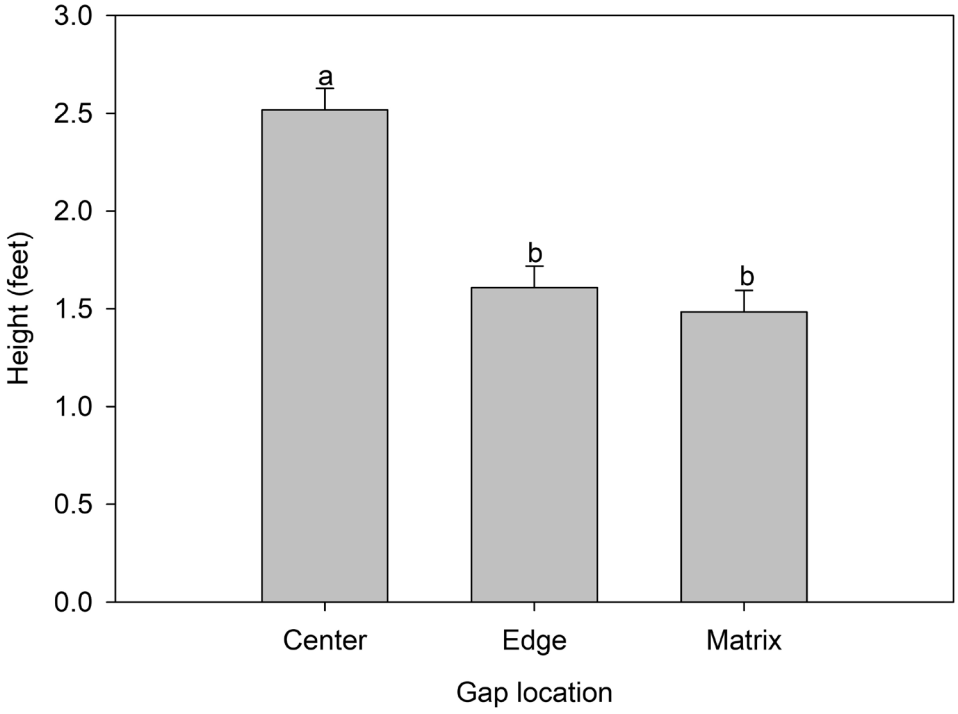

Figure 1—Third-year height (least-squares means) of white oak seedlings planted in 0.25-acre gaps in western North Carolina.

This study was relatively novel for the Central Hardwood Region by incorporating artificial regeneration into an irregular shelterwood. Unlike more traditional regeneration systems (e.g., clearcut, shelterwoods resulting in low basal area) that have received more research and management attention in the Region (c.f., Brose 2010, Schweitzer and Dey 2011), the smaller-sized and irregular-shaped shelterwood was designed to mimic natural disturbances, but its efficacy has not been adequately tested for oak forests. Artificial regeneration of white oak has been historically difficult, owing to poor seedling quality, deer browse,

and lack of competition control (Dey and others 2008), but planting and nursery protocols have been better refined in the last 3 decades to improve planting success (Clark and Schlarbaum 2019, Kormanik and others 1994).

Planted white oak seedlings did not grow appreciably in height, even in the gap centers (17 inches of stem growth over 3 years), especially compared to competition, dominated by yellow-poplar (*Liriodendron tulipifera*) (data not shown). Future gap expansions are predicted to benefit established seedlings in the gap edges and forest matrix, where oak seedlings should maintain their physiological demands in intermediate light environments (Patterson and others 2022). Planting density and gap direction did not affect seedling performance in the first 3 years, but seedling growth dynamics are expected to change over time as the root systems for seedlings become better established and the gaps are expanded, resulting in an increase in light. Despite the limited biological implications related to planting density, the higher density plots required less establishment time because trees were grouped together in relatively small areas, which simplified planting logistics. Higher planting density may also simplify future tending practices (i.e., reduction of the treated area around seedlings).

A similar study in eastern Tennessee showed that a proportion of white oak seedlings planted into gap edges grew into competitive canopy positions 7 years after planting, and these seedlings are now in a favorable position for recruitment into the canopy following gap expansion (Clark and others, In press). In this study, seedling quality was comparatively lower. Seedling quality varies from year to year due to weather, and nursery and management schedules that can disrupt timely sowing, lifting, and seedling storage. Seed quality impacted by insects and seed storage (Rink and Williams 1984) can also degrade seedling growth in the nursery, and delayed sowing time may have reduced overall seedling size in this study. Despite the relatively common challenges, we increased the overall oak regeneration potential of the forest and improved chances for oak recruitment by establishment of new age classes through gap expansion.

ACKNOWLEDGMENTS

This manuscript was improved by reviews from Leila Pinchot (USDA Forest Service) and David Mercker (The University of Tennessee). The authors thank Pisgah National Forest personnel who were involved in the implementation of this study, including Jason Rodrigue and Rachel Dickson. University of Tennessee personnel, Jason Hogan, John Johnson, and Ami Sharp, and U.S. Department of Agriculture, Forest Service personnel, Jacquelyne Adams and William Preston Durham assisted in data collection and planting establishment.

LITERATURE CITED

Brose, P.H. 2010. Long-term effects of single prescribed fires on hardwood regeneration in oak shelterwood stands. Forest Ecology and Management. 260: 1516–1524. https://doi.org/10.1016/j.foreco.2010.07.050.

Clark, S.L.; Schlarbaum, S.E. 2019. Artificial regeneration in the Southern Appalachians. In: Clark, S.L.; Schweitzer, C.J., eds. Oak symposium: sustaining oak forests in the 21st century through science-based management. e-Gen. Tech. Rep. SRS-237. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 95–100. https://doi.org/10.2737/SRS-GTR-237.

Clark, S.L.; Schlarbaum, S.E.; Schweitzer, C.J. [and others]. In press. Planted white oak (*Quercus alba*) and natural regeneration response to even- and uneven-aged silviculture in the Ridge and Valley ecoregion of Tennessee. In: Proceedings of the 22nd central hardwood forest conference. Gen. Tech. Rep. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station.

Dey, D.C.; Jacobs, D.; McNabb, K. [and others]. 2008. Artificial regeneration of major oak (*Quercus*) species in the Eastern United States—a review of the literature. Forest Science. 54(1): 77–106. https://doi.org/10.1093/forestscience/54.1.77.

Kormanik, P.P.; Sung, S.J.S.; Kormanik, T.L. 1994. Towards a single nursery protocol for oak seedlings. In: Lantz, C.W.; Moorhead, D.J., comps. Proceedings of the 22nd southern forest tree improvement conference. Sponsored Publ. No. 44. Atlanta, GA: Southern Forest Tree Improvement Committee: 89–98.

Patterson, C.P.; Hackworth, Z.J.; Lhotka, J.M.; Stringer, J.W. 2022. Light and regeneration patterns following silvicultural gap establishment in *Quercus* dominated stands of the northern Cumberland Plateau, USA. Forest Ecology and Management. 505: 119871. 12 p. https://doi.org/10.1016/j.foreco.2021.119871.

Rink, G.; Williams, R.D. 1984. Storage technique affects white oak acorn viability. Tree Planters' Notes. 35(1): 3–5.

Schweitzer, C.J.; Dey, D.C. 2011. Forest structure, composition, and tree diversity response to a gradient of regeneration harvests in the mid-Cumberland Plateau escarpment region, USA. Forest Ecology and Management. 262: 1729–1741. https://doi.org/10.1016/j.foreco.2011.07.020.

Schweitzer, C.J.; Dey, D.C.; Wang, Y. 2019. White oak (*Quercus alba*) response to thinning and prescribed fire in northcentral Alabama mixed pine-hardwood forests. Forest Science. 65(6): 758–766. https://doi.org/10.1093/forsci/fxz031.

Woodbridge, M.; Keyser, T.; Oswalt, C. 2022. Stand and environmental conditions drive functional shifts associated with mesophication in eastern US forests. Frontiers in Forests and Global Change. 5: 991934. 16 p. https://doi.org/10.3389/ffgc.2022.991934.

MULTIPHASED SHELTERWOOD ON THE DANIEL BOONE NATIONAL FOREST, KENTUCKY, TESTS HERCULEAN EFFORT TO REGENERATE OAK

Callie Schweitzer, Jared Calvert, Stacy Clark, and Jacob Royse

EXTENDED ABSTRACT

Our desire to maintain biodiverse forests is predicated on a desire to maintain oaks (*Quercus* spp.) in dominant canopy positions and high stocking levels. Oak reproduction is advance-growth dependent; that is, there must be adequate numbers of oak seedlings in competitive sizes prior to a canopy-replacing disturbance. Therefore, adaptive regeneration strategies are needed to position oak reproduction for recruitment into future canopy positions.

Regenerating oaks with a shelterwood prescription that manipulates understory light by deadening midstory stems, followed by release of oak stems that have recruited into larger size classes in response to increased light, has been extensively tested across the Eastern United States (Brose and others 2008, Craig and others 2014, Hutchinson and others 2016, Janzen and Hodges 1987, Lockhart and others 2000, Loftis 1990, Miller and others 2014, Parrott and others 2012, Schweitzer and Dey 2011). Because of the mesic nature of contemporary forests, coupled with additional time since last disturbance, modification to this prescription may be warranted. For example, an additional treatment targeting undesirable understory stems is needed to combat the overwhelming competition of red maple (*Acer rubrum*) in some stands (Arthur and others 1997, Lhotka 2012, Parrott and others 2012).

We engaged in stand-level treatments under an adaptative management strategy to position oak reproduction in more competitive positions prior to a regeneration harvest on the Daniel Boone National Forest in Kentucky. The initial hypothesis was that by reducing midstory stems in a mature upland hardwood forest, we would stimulate growth and recruitment of oak advance reproduction while discouraging the response of undesirable species (Cold Hill Silvicultural Assessment 2006). Ten years after the

Callie Schweitzer, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Huntsville, AL 35801

Jared Calvert, Forest Silviculturist, U.S. Department of Agriculture, Forest Service, Daniel Boone National Forest, Winchester, KY 40391

Stacy Clark, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Knoxville, TN 37996

Jacob Royse, Forester, U.S. Department of Agriculture, Forest Service, Daniel Boone National Forest, London, KY 40744

midstory removal, we added an additional treatment targeting the red maple competition that had accumulated in understory and midstory positions prior to overstory removal (Cold Hill HFRA Applied Silvicultural Assessment, Phase II 2018). Finally, in completing the shelterwood process, we predicted that, following overstory canopy removal, the large oak advance reproduction would continue to recruit into upper canopy positions, and the stands would remain well stocked with oak, especially as compared to undisturbed stands.

The oak shelterwood treatment (OSW) was implemented in 2006 by treating midstory stems of undesirable species in six stands (20 to 25 acres each). We followed reproduction by surveying stems in permanent measurement plots by species and 1-foot height classes, from <1 foot tall and ≤1.5 inches in diameter at breast height (d.b.h.). Ten years post-treatment, oaks in the largest size class, ≥4 feet tall and ≤1.5 inches d.b.h., increased to 163 stems per acre (SPA). In this same size class, red maple, a less desirable species, increased to 1,377 SPA (Schweitzer 2019; Schweitzer and others 2008, 2011, 2014, 2020). We did a preharvest treatment in 2019, characterized as "site preparation for natural regeneration" in the National Environmental Policy Act documentation. Red maple stems were targeted using the herbicide imazapyr (Arsenal AC®, BASF Corporation, Research Triangle Park, NC) via a hack-and-squirt or cut-surface application to red maples 0.5–11.9 inches d.b.h. Because of travel restrictions during the COVID-19 pandemic, we were unable to collect data immediately following this treatment. The overstory was removed by Kentucky Hardwood Lumber Company LLC using a mechanical tracked feller-buncher in September 2020, and stands were remeasured in 2022. We compared results in the OSW stands to six control stands in the same study.

In 2005, the OSW overstory basal area was 99 square feet per acre [4 percent hickories (*Carya* spp.), 64 percent oaks, and 13 percent red maple] and density was 127 SPA (3 percent hickories, 51 percent oaks, and 30 percent red maple) for trees >5.4 inches d.b.h. After harvest, the residual basal area was 27 square feet per acre (6 percent hickories, 50 percent oaks, and 29 percent red maple) with 25 SPA (8 percent hickories, 60 percent oaks, and 20 percent red maple) for trees >5.4 inches d.b.h.

Oak reproduction included black oak (*Q. velutina*), chestnut oak (*Q. prinus*), scarlet oak (*Q. coccinea)*, and white oak (*Q. alb*a). Oak reproduction was maintained over time (F = 1.01, *P* = 0.3869) at 2,730 SPA in 2008, 4,183 SPA in 2016, and 3,937 SPA in 2022 (fig. 1). In 2008 (F = 0.57, *P* = 0.4685) and 2016 (F = 3.22, *P* = 0.1030) differences in total oak density and density within any size class were not significant between the control and OSW. In 2022, total oak density was less in the OSW with 3,937 SPA compared to 8,490 SPA in the control (F = 6.46, *P* = 0.0293). This was mostly due to the reproduction of the smallest size class (<1 foot tall), which was 6,457 SPA in the control compared to 2,073 SPA in the OSW (F = 10.19, *P* = 0.0096). From 2005 to 2022 the largest size class (≥4 feet tall and ≤1.5 inches d.b.h.) increased for both treatments, from 20 to 180 SPA for the control and from 43 to 143 SPA for the OSW (fig. 2).

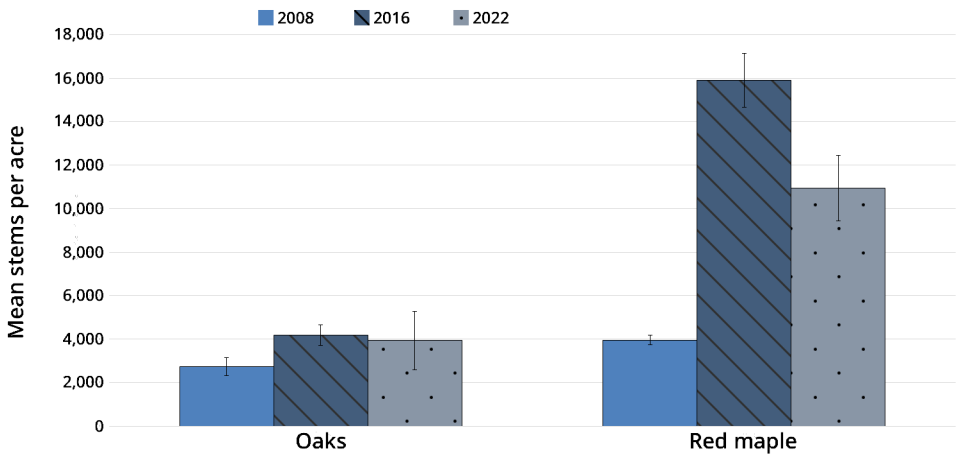

Figure 1—Mean stems per acre for total oak and red maple reproduction (≤1 foot tall and ≤1.5 inches d.b.h.) under an oak shelterwood prescription. 2008 represents pretreatment density, 2016 is after the first midstory herbicide treatment, and 2022 is following a second midstory treatment and final overstory harvest.

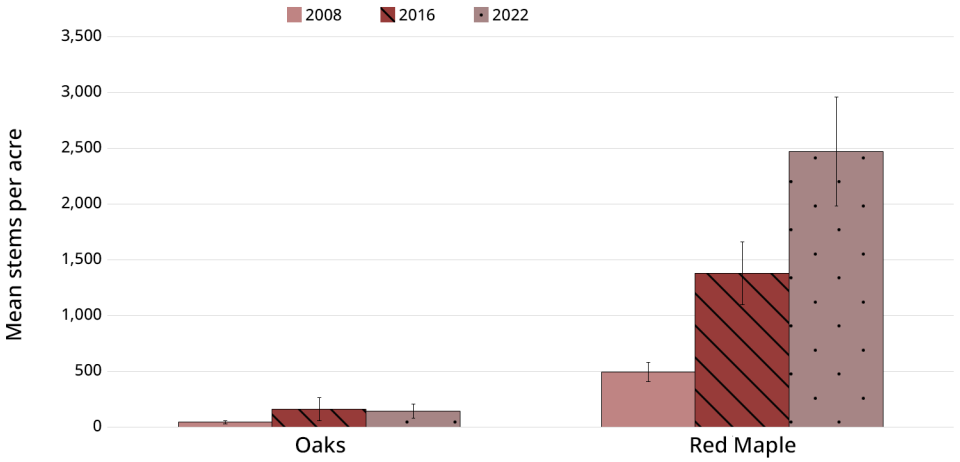

Figure 2—Mean stems per acre for oak and red maple reproduction in the largest size class, ≥4 feet tall and ≤1.5 inches d.b.h., under an oak shelterwood prescription. 2008 represents pretreatment density, 2016 is after the first midstory herbicide treatment, and 2022 is following a second midstory treatment and final overstory harvest.

Total red maple reproduction in the OSW stands significantly changed over time (F = 27.95, *P* < 0.0001), from 3,957 SPA in 2008 to 15,897 SPA in 2016 and 10,950 SPA in 2022 (fig. 1). Red maple in the largest size class in the OSW had significantly more stems in 2022 (2,470) than in 2008 (495) and 2016 (1,377) (F = 9.08, *P* = 0.0026) (fig. 2). In the smallest size class, reproduction of OSW red maple was significantly greater (F = 20.26, *P* <0.0001) in 2016 (11,257 SPA) than in 2008 (2,187 SPA) and 2022 (4,283 SPA). Total red maple stems in the control stands increased (F = 73.04, *P* <0.0001), changing from 4,423 (2008) to 14,327 (2016) to 20,290 (2022) SPA. This was the result of significant increases in the smallest size class only (F = 54.04, *P* <0.0001), and represented 57, 85, and 90 percent of the total red maple SPA for 2008, 2016, and 2022, respectively. In 2008 and 2016 total red maple seedling densities were not different between control and OSW stands. However, in 2022 control stands had a greater density of red maple stems (20,290 SPA) than OSW (10,950 SPA) (F = 25.74, *P* = 0.0005). Control stands also had greater stem densities in all size classes compared to OSW except for the 1- to 2-foot height class, which did not differ.

Intensive management was undertaken to ensure sufficient oak stocking after a regeneration harvest, and oaks maintained their stocking within the OSW stands. The effects of mesophication are complicating the reproduction response, with small and most likely new germinants of red maple proliferating in both treatments. Targeting red maple with multiple treatments reduced its density compared to controls. These intense and targeted treatments have resulted in the increase and maintenance of competitive oak, although future release may be warranted.

LITERATURE CITED

Arthur, M.A.; Muller, R.N.; Costello, S. 1997. Species composition in a central hardwood forest in Kentucky 11 years after clear-cutting. American Midland Naturalist. 137(2): 274–281. https://doi.org/10.2307/2426846.

Brose, P.H.; Gottschalk, K.W.; Horsley, S.B. [and others]. 2008. Prescribing regeneration treatments for mixed-oak forests in the Mid-Atlantic region. Gen. Tech. Rep. NRS-33. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 100 p. https://doi.org/10.2737/NRS-GTR-33.

Cold Hill HFRA Applied Silvicultural Assessment, Phase II. 2018. Decision memo. Laurel County, KY: U.S. Department of Agriculture, Forest Service, Southern Region, Daniel Boone National Forest, London Ranger District. 20 p.

Cold Hill Silvicultural Assessment. 2006. Decision memo. Laurel County, KY: U.S. Department of Agriculture, Forest Service, Southern Region, Daniel Boone National Forest, London Ranger District. 12 p.

Craig, J.M.; Lhotka, J.M.; Stringer, J.W. 2014. Evaluating initial responses of natural and underplanted oak reproduction and a shade-tolerant competitor to midstory removal. Forest Science. 60(6): 1164–1171. https://doi.org/10.5849/forsci.13-602.

Hutchinson, T.F.; Rebbeck, J.; Stout, S.L. 2016. The devil is in the small dense saplings: a midstory herbicide treatment has limited effects on short-term regeneration outcomes in oak shelterwood stands. Forest Ecology and Management. 372: 189–198. https://doi.org/10.1016/j.foreco.2016.04.016.

Janzen, G.C.; Hodges, J.D. 1987. Development of advanced oak regeneration as influenced by removal of midstory and understory vegetation. 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 Experiment Station: 455–461.

Lhotka, J.M. 2012. Effect of gap size on mid-rotation stand structure and species composition in a naturally regenerated mixed broadleaf forest. New Forests. 44(3): 311–325. https://doi.org/10.1007/s11056-012-9319-7.

Lockhart, B.R.; Hodges, J.D.; Gardiner, E.S. 2000. Response of advance cherrybark oak reproduction to midstory removal and shoot clipping. Southern Journal of Applied Forestry. 24(1): 45–50. https://doi.org/10.1093/sjaf/24.1.45.

Loftis, D.L. 1990. Predicting post-harvest performance of advance red oak reproduction in the southern Appalachians. Forest Science. 36(4): 908–916. https://doi.org/10.1093/forestscience/36.4.908.

Miller, G.W.; Kochenderfer, J.N.; Kochenderfer, J.D.; Gottschalk, K.W. 2014. Ten-year response of competing vegetation after oak shelterwood treatments in West Virginia. In: Groninger, J.W.; Holzmueller, E.J.; Nielsen, C.K.; Dey, D.C., eds. Proceedings of the 19th central hardwood forest conference. Gen. Tech. Rep. NRS-P-142. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 156–171.

Parrott, D.L.; Lhotka, J.M; Stringer, J.W.; Dillaway, D.N. 2012. Seven-year effects of midstory removal on natural and underplanted oak reproduction. Northern Journal of Applied Forestry. 29(4): 182–190. https://doi.org/10.5849/njaf.12-001.

Schweitzer, C.J. 2019. Recruiting oak using midstory herbicide shelterwood prescriptions in Cumberland Plateau forests of Alabama, Tennessee, and Kentucky. In: Clark, S.L.; Schweitzer, C.J., eds. Oak symposium: sustaining oak forests in the 21st century through science-based management. e-Gen. Tech. Rep. SRS-237. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 51–61.

Schweitzer, C.J.; Calvert, J.; Clark, S. 2020. Managers and scientists unite to adapt a shelterwood prescription to shift stand dynamics for competitive oak reproduction. In: Pile, L.S.; Deal, R.L.; Dey, D.C. [and others], comps. The 2019 National Silviculture Workshop: a focus on forest management-research partnerships. Gen. Tech. Rep. NRS-P-193. Madison, WI: U.S. Department of Agriculture Forest Service, Northern Research Station: 141–154. https://doi.org/10.2737/NRS-GTR-P-193-paper20.

Schweitzer, C.J.; Clark, S.L.; Gaines, G. [and others]. 2008. Integrating Land and Resource Management Plans and applied large-scale research on two national forests. In: Deal, R.L., tech. ed. Integrated restoration of forested ecosystems to achieve multiresource benefits: proceedings of the 2007 national silviculture workshop. Gen. Tech. Rep. PNW- 733. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station: 127–134.

Schweitzer, C.; Clark, S.L.; Gottschalk, K.W. [and others]. 2014. Proactive restoration: planning, implementation, and early results of silvicultural strategies for increasing resilience against gypsy moth infestation in upland oak forests on the Daniel Boone National Forest, Kentucky. Journal of Forestry. 112: 401–411. https://doi.org/10.5849/jof.13-085.

Schweitzer, C.J.; Dey, D.C. 2011. Forest structure, composition, and tree diversity response to a gradient of regeneration harvests in the mid-Cumberland Plateau escarpment region, USA. Forest Ecology and Management. 262: 1729–1741. https://doi.org/10.1016/j.foreco.2011.07.020.

Schweitzer, C.J.; Gottschalk, K.W.; Stringer, J.W. [and others]. 2011. Using silviculture to sustain upland oak forests under stress on the Daniel Boone National Forest, Kentucky. In: Fei, S.; Lhotka, J.M.; Stringer, J.W. [and others], eds. Proceedings of the 17th central hardwood forest conference. Gen. Tech. Rep. NRS-P-78. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 476–489.

A LOOK BACK ON 26 YEARS OF HARDWOOD DEVELOPMENT AFTER AFFORESTATION ON FLOOD-PRONE AGRICULTURAL LANDS IN THE LOUISIANA DELTA

Joshua P. Adams and William B. Patterson

EXTENDED ABSTRACT

Over the last half-century, thousands of acres have been afforested in the Mississippi Alluvial Valley (MAV) with projections of nearly 30 million acres by 2040 (Wear 2002). This is after a previous 2 centuries where nearly 75 percent of the original hardwoods were removed from the MAV (Nature Conservancy 1992). Establishment of new stands across this large area has resulted in a wide variety of survival and growth characteristics across soils, topography, and silvicultural techniques (Cunningham and Stuhlinger 2018, Self and Willis 2022)

In 1996, an 80-acre site located on the Bouef River Wildlife Management Area between the Ouachita and Bouef Rivers was planted in hardwood tree species. The site from west to east was composed of Alligator clay in the lowest elevation, Perry clay in the middle of the site, and Hebert silt loam on the highest part of the site. Each soil type composed roughly a third of the site and there was a 3- to 4-foot rise from west to east. Approximately 439 trees per acre were planted at 10-foot by 10-foot spacing. Further details on establishment of this study are reported by Patterson and Adams (2003).

In 2022, 88 tenth-acre plots were established across the 3 soil types. While the original study included a bedding treatment, this study randomly distributed plots across each of the three soil types. In each plot, total number of trees that were ≥4 inches in diameter at breast height (d.b.h.) were recorded and classified as overcup oak (*Quercus lyrata*), Nuttall oak (*Q. texana*), willow oak (*Q. phellos*), or other. The other grouping included species such as Chinese tallow tree (*Triadeca sebifera*), bitter pecan (*Carya aquatica*), baldcypress (*Taxodium distichum*), honey locust (*Gleditsia triacanthos*), and others that occurred in low numbers. Total height and d.b.h. were recorded for each tree.

Tree density across the three soil types averaged 106, 84, and 87 trees per acre for the Alligator, Perry, and Hebert soils types, respectively (table 1). Though Alligator soils had a greater number of trees per acre, trees on Alligator and Perry soils were not significantly different in either height or d.b.h. However, trees on the Hebert soil were significantly different (*P*

Joshua P. Adams, Associate Professor, Louisiana Tech University, Ruston, LA 71272
William B. Patterson, Associate Professor, Louisiana Tech University, Ruston, LA 71272

<0.01) than trees on the other two soil types and were much larger overall with a 41 percent larger diameter and 43 percent greater height than the average of the other two soil types. Trees per acre in conjunction with basal area resulted in the Hebert soil type having 69 square feet of basal area per acre compared to 40.5 and 31.5 square feet for the Alligator and Perry soils, respectively (table 1). When plotted on an established stocking guide (Goelz 1995) both the Alligator and Hebert were above the C-10 line but had still not reached the B-line, although the Hebert was the closest. The Perry soil stocking was slightly below the C-10 line (fig. 1).

Distinguishing these three soil types was a change in the suite of species represented. All sites were dominated by Nuttall oak with a 36, 70, and 75 percent representation within the Alligator, Perry, and Hebert soils, respectively. The lowest site and most westerly, Alligator clay, was composed of 36 percent overcup oak but had no willow oak. This shifted immediately to the east, with the slightly higher-elevated Perry soil having greatly reduced overcup oak representation of 12 percent while having a slight 1.2 percent representation of willow oak. The Hebert soil had only 2 percent overcup oak representation with 12.7 percent willow oak. Across the three soils types there were only 1.5–3.6 percent in the other category. Notably, the absence of green ash (*Fraxinus pennsylvanica*) that was specifically planted in this study is glaring (Patterson and

Table 1—Diameter at breast height, height, basal area per acre, and trees per acre on the three soil types at the Bouef River Wildlife Management Area study site

Soil type	d.b.h.	Height	Basal area	Trees
	inches	*feet*	*square feet per acre*	*per acre*
Alligator clay	8.15	39.41	40.6	342
Perry silt loam	8.05	54.49	31.5	244
Hebert clay	11.46	36.70	69.1	236

d.b.h. = diameter at breast height.

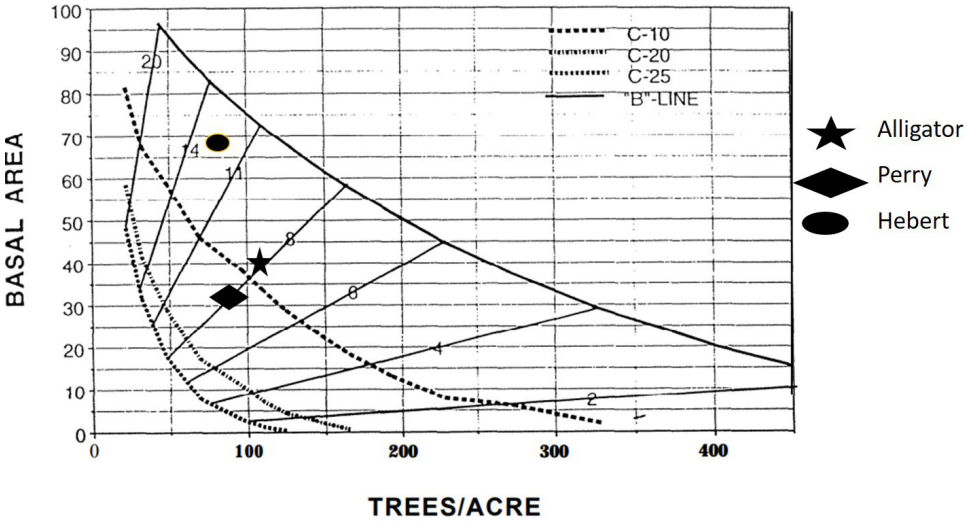

Figure 1—Basal area in square feet per acre and trees per acre estimates across species for each soil type on the Goelz (1995) stocking guide.

Adams 2003), though there are no specific reasons apparent, and ash was assumed to be less competitive on this particular site. Similar to other studies, there was a narrowing in species number representation as the stand aged (Lockhart and others 2006) in which one or a few species exerted dominance. This supports the difficulty observed in other studies for the establishment of stands immediately after afforestation. Furthermore, attempts to devote resources (i.e., seedlings representing many species) to promote diversity is not effective. Rather, attempts to plant a small number of species that will survive in the first few decades may be prioritized. However, if such a route is taken, success will depend on correct species selection.

LITERATURE CITED

Cunningham, K.K.; Stuhlinger, H.C. 2018. A comparison of Nuttall oak establishment methods using improved and unimproved seedlings, seedling treatments, and site preparation intensity in the Lower Mississippi Alluvial Valley. In: Kirschman, J.E., comp. Proceedings of the 19th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-234. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 184–189.

Goelz, J.C.G. 1995. A stocking guide for southern bottomland hardwoods. Southern Journal of Applied Forestry. 19(3): 103–104. https://doi.org/10.1093/sjaf/19.3.103.

Lockhart, B.R.; Ezell, A.W.; Hodges, J.D.; 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(1-3): 202–210. https://doi.org/10.1016/j.foreco.2005.09.029.

Nature Conservancy. 1992. Restoration of the Mississippi River alluvial plain as a functional ecosystem. Baton Rouge, LA: The Nature Conservancy.

Patterson, W.B.; Adams, J.C. 2003. Soil, hydroperiod and bedding effects on restoring bottomland hardwoods on flood-prone agricultural lands in North Louisiana, USA. Forestry. 76(2): 181–188. https://doi.org/10.1093/forestry/76.2.181.

Self, A.B.; Willis, J.L. 2022. Eleven-year effects of mechanical site preparation on oaks planted on former agricultural fields. Forests. 13: 1202. 10 p. https://doi.org/10.3390/f13081202.

Wear, D.N. 2002. Land use. In: Wear, D.N.; Greis, J.G., eds. Southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 153–173. https://doi.org/10.2737/SRS-GTR-53.

REGENERATIVE EFFECTS OF PATCH-CUT HARVESTS AT NATCHEZ TRACE STATE FOREST IN WEST TENNESSEE AFTER MORE THAN 25 YEARS

Wayne K. Clatterbuck and Joshua K. Biggerstaff

ABSTRACT

Patch cutting is a regeneration method with little precedent in the Central Hardwoods Region of the United States. Patch cuts are defined as small scale clearcuts of 1–5 acres and are generally prescribed to lessen the aesthetic impact of harvesting in highly visible areas. This study examines species composition and stand development after patch cutting in the 1990s at Natchez Trace State Forest, located in the Coastal Plain of western Tennessee. Patch cuts were hypothesized to have a greater proportion of shade-intolerant species in the center of the opening and transitioning to more shade-intermediate and shade-tolerant species toward the edge of the harvest. Results indicate that patch cuts were dominated by yellow-poplar (*Liriodendron tulipifera*) across the light gradient of the patch opening. Few shade-intermediate (oaks, *Quercus* spp.) and shade-tolerant species were encountered. Yellow-poplar has greater light plasticity in these transitory light conditions toward the edge of the patch cut, regenerating, and outgrowing other species with lesser light requirements, thus nullifying the hypothesis in this research.

INTRODUCTION

Patch cutting, which is often referred to as "small clearcuts" or "patch clearcutting" refers to a regeneration harvest of 1–5 acres. These patches create small, even-aged areas within a larger, uneven-aged landscape (Meadows and Stanturf 1997). This regeneration regime is used primarily to lessen the aesthetic impact of harvesting. Patch cutting is not a common harvest regime in central hardwood or southern bottomland hardwood areas and its regenerative effects have not been extensively studied, especially after more than 25 years. Similar harvest regimes of interest such as clearcutting and group selection have been evaluated in the southern Appalachian Mountains (McNab and Oprean 2021). Typically, larger area, complete cuts support regeneration of shade-intolerant species while group-selection regeneration favors more intermediate-light and shade-tolerant species. Patch cutting is more common in New England (Yamasaki and others 2014) and most of the literature on the subject is from that region.

The patch cuts at Natchez Trace State Forest (NTSF) were initially publicly referred to as group-selection regeneration. True group-selection openings are much smaller than patch cuts. Nyland (2002) defines group selection as an opening that is one to two times the height of the surrounding mature trees. With group selection, the entire opening is influenced by the adjacent trees on the edge of the opening (LeDoux 1999). Patch cuts are large enough that there is at least some area in the center of the opening that receives direct or full sunlight and is not influenced by adjacent trees (Yamasaki and others 2014).

The goal of this study was to investigate regenerative effects of patch cuts on species composition in relation to the establishment and growth of oak (*Quercus*) species, a keystone genus for wildlife and timber in Tennessee (Clatterbuck 2019). Oaks are classified as light-intermediate species (Fox and Creighton 2016, Johnson and others 2009). Patch cuts may provide moderate light conditions that benefit oak regeneration, growth, and development. Faster-growing,

Wayne K. Clatterbuck, Professor, University of Tennessee, School of Natural Resources, Knoxville, TN 37996
Joshua K. Biggerstaff, Graduate Student, University of Tennessee, School of Natural Resources, Knoxville, TN 37996

shade-intolerant yellow-poplars (*Liriodendron tulipifera*) are common in full sunlight, while more shade-tolerant species such as maples (*Acer* spp.) and American beech (*Fagus grandifolia*) are frequent in partial light areas adjacent to edges of openings.

METHODS

Study Area

Natchez Trace State Forest, State Park, and Wildlife Management Area encompasses 46,000 acres in western Tennessee, approximately 30 miles east of Jackson, TN. The State Forest (36,000 acres) completely encircles the 10,000-acre State Park. The land was acquired in 1949 when purchased by the Federal Government as part of the New Deal. Much of the flatter ridges and upper slopes were farmed, causing broad erosion and abandonment within a few years (McClurkin and others 1985). These areas were planted with loblolly pine (*Pinus taeda*) to control erosion. Other areas had more rolling topography including narrow ridges, short side slopes ranging to 10 percent, and V-shaped valleys that were left undisturbed in forests. Smalley (1991) classified the forest as the Coastal Plain Physiographic Province, East Gulf Coastal Plain Region, Upper Coastal Plan Hills Subregion, and two Landtype Associations: (A) Sandy and Loamy Hills, and (B) Clayey and Sandy Clay Hills. Most of the unsustainable agricultural practices occurred on Landtype Association A. Sample stands were from Landtype Association B which historically were undisturbed and included Lexington, Providence, Ruston, Savannah, and Smithdale soil series, soil textures of sandy loams and silt loams, and classified as alfisols and ultisols, some with fragipans. Site index at 50 years ranged from 80 to 90 feet for yellow-poplar and 65 to 75 feet for oaks (Smalley 1991).

Stands were mature at harvest with closed canopies. Overstory species included red oaks (*Quercus rubra*) and white oaks (*Quercus alba*), yellow-poplar, hickories (*Carya* spp.), sweetgum (*Liquidambar styraciflua*), black cherry (*Prunus serotina*), and maples. Planted, mature loblolly pine stands were adjacent to a few harvest units, but not directly next to the patch cuts.

Data Collection

Four patch-cut harvest units were selected across NTSF. Within each harvest unit, there were many individual patch cuts. Four patch cuts were selected from each harvest unit based on accessibility and determining patch cut boundaries. Patch cuts were often difficult to delineate and determine boundaries of the cut area after more than 25 years of growth. Six 0.05-acre plots were sampled within each selected patch cut. The plots taken were categorized based on their proximity to the edge of the harvest area. Two plots were taken near the edge of the harvest area, two plots near the center, and two plots in the intermediate range between the edge and center. Most patch cuts were 2–3 acres. Since the size and shape of patch cuts varied, plot selection was subjective, but without preconceived bias (Mueller-Dombois and Ellenberg 1974). The three plot locations (center, intermediate, and edge) were sampled to determine if a change in species composition and stand structure occurred along the gradient of light conditions at time of harvest. A chi-square test evaluated whether there was a relationship between plot location and individual species. Within each plot, every tree with a diameter at breast height (4.5 feet, d.b.h.) ≥5 inches was measured and species determined. This process yielded 96 plots: 4 harvest units (stands), 4 patch cuts sampled for each harvest unit, and 6 plots for each patch cut.

RESULTS

A total of 969 trees that were ≥5 inches d.b.h. were tallied on 96 plots. Measured trees on the patch cuts across harvest units represented an average of 201.7 trees per acre (TPA) and 65.6 square feet of basal area for the study area.

Yellow-poplar was the most prominent species present in the patch cuts across all stands and locations, accounting for 44 percent of total stems measured (table 1) with 88 TPA. Sweetgum with 19 percent and 39 TPA and black cherry with 13 percent and 27 TPA were the second- and third-ranked species. Red oaks and white oaks combined composed 3 percent of the total stems with 6 TPA. The remaining trees were maples (6 percent, 12 TPA) and the miscellaneous category (15 percent, 30 TPA).

Table 1—Composition, average diameter, and basal area of patch-cut openings for stems ≥5 inches in diameter

Species	Composition	Average diameter	Basal area
	percent (trees per acre)	*inches*	*square feet per acre (percent)*
Yellow-poplar	44 (88)	7.8	32.7 (50)
Black cherry	13 (27)	6.2	5.9 (9)
Sweetgum	19 (39)	7.0	11.1 (17)
Maples	6 (12)	5.9	2.6 (4)
White oaks	2 (4)	7.0	1.3 (2)
Red oaks	1 (2)	6.7	0.7 (1)
Miscellaneous	15 (30)	7.6	11.1 (17)

Each miscellaneous species composed <4 percent of the total stems.

The diameter distribution follows the reverse-J (negative exponential) pattern typical of stratified, even-aged stands (fig. 1). Yellow-poplar was not only the most abundant species, but also the greatest in average diameter at 7.8 inches (table 1). Sweetgum was the second-ranked species in diameter.

Yellow-poplar was even more prominent with 50 percent of the total basal area and 32.7 square feet of basal area in the patch cuts (table 1). Sweetgum and black cherry were the second- and third ranked species with 11.1 and 5.9 square feet of basal area per acre, respectively. Red and white oaks combined represented 3 percent or 2 square feet of basal area per acre.

Plot Location

For plots near the edges of the patch cuts, yellow-poplar was the most common species accounting for 34 percent of total stems per acre but composed greater percentages in the intermediate (51 percent) and center (46 percent) locations (table 2). The proportion of individual species

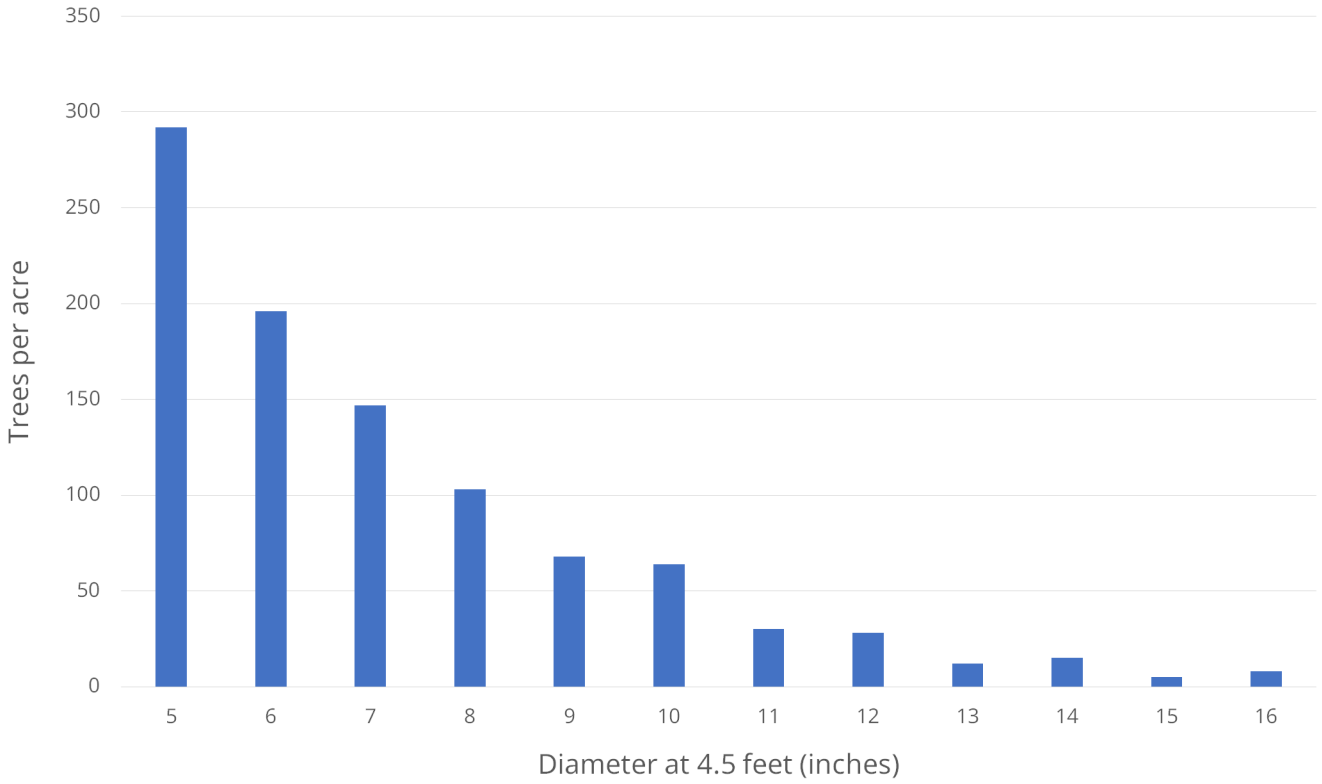

Figure 1—Patch cut diameter distribution across species from 96 plots at Natchez Trace State Forest.

Table 2—Species composition as a percentage of total trees for each of the three patch-cut locations

Species	Edge	Intermediate	Center
	~~ *percent* ~~~~~~~~~~		
Yellow-poplar	34.1	50.8	45.6
Black cherry	14.4	12.0	13.8
Sweetgum	18.1	13.6	24.9
Maples	8.4	7.1	3.0
White oaks	2.3	1.6	1.7
Red oaks	0.3	2.6	1.1
Miscellaneous	22.4	12.3	9.9

The increase in miscellaneous species in the edge plots was from the natural seeding of loblolly pine from adjacent stands in one of the harvest units.

The chi-square test failed to demonstrate any real association between plot location and species at the $P = 0.05$ level.

was similar across plot locations. The increase in miscellaneous species in the edge plots was from the natural seeding of loblolly pine from adjacent stands in one of the harvest units.

DISCUSSION

The patch-cut survey results were surprising. While the hypothesis of this study predicted that shade-tolerant species would constitute a significantly greater proportion of the species composition in the patch-cut area, this was not the case. Yellow-poplar dominated these areas, including the edges. Shade-tolerant species such as maple were present, but they were virtually absent from the overstory. Given the dominance of yellow-poplar on these sites, it was not surprising that sweetgum and black cherry were commonly observed species (Eyre 1980). Sweetgum regenerates with yellow-poplar at sites within a stand that are too moist for yellow-poplar. Black cherry is shade-intolerant and is often associated with yellow-poplar as a pioneer species. However, yellow-poplar superseded sweetgum and black cherry on the study sites. The environmental and site conditions in the regeneration and continued growth of the patch cuts favored yellow-poplar more than sweetgum and black cherry.

Using basal area of the patch-cut areas, yellow-poplar composed 50 percent of the total species composition (table 1), further emphasizing the dominance of yellow-poplar, not only in number, but in occupied growing space. Yellow-poplars are the largest stems and occupy more growing space than any other species. Maples were the shade-tolerant species expected to be the beneficiary of patch cuts along the edges, but they accounted for 6 percent of stems and 4 percent of the basal area.

The edge plots did show an increased proportion of shade-tolerant species when compared to the intermediate and center plots (table 2). This increase was not as much, as expected, based on the decrease of direct sunlight approaching the edge of the opening and the literature review (McNab and Oprean 2021, Yamasaki and others 2014) that suggested shade-tolerant and light-intermediate species would be more common near the edge of the openings. Species composition did not change appreciably from the edge, to the intermediate, to the center of the patch openings.

The near absence of oaks in patch cuts despite mature oaks being prominent before the harvest is perplexing. Oaks, being shade intermediate, lead to the expectation that they would appear more often than they did as the patch-cut opening gradually transitioned to the edge. Many studies have documented that 20 to 30 percent of full sunlight is necessary for developing competitive oak seedlings (Clatterbuck 2019). Sander (1972) and Sander and others (1976) suggest that adequate number and size (≥4 feet tall) of oak advance reproduction is vital to successfully regenerate oak. Preharvest regeneration data for the areas that were patch cut at NTSF are not available. We assume that there was little oak advance reproduction present at the sites prior to harvest. Even if oak advance reproduction were present, there would be some question whether the slower growth of oaks would be able to develop with faster growing yellow-poplar, even in diminished light conditions (Schweitzer and others 2016).

CONCLUSION

Yellow-poplar was expected to dominate in the center of regenerated patch cuts with its fast growth, seed viability for several years, and sprouting ability under full light conditions (Beck and Della-Bianca 1981). However, yellow-poplar was not expected to dominate the

regenerated stands as the amount of sunlight subsided toward the edge of the patch cuts. More shade-intermediate (oaks) and shade-tolerant species (maple and beech) were anticipated. The hypothesis that a greater diversity of species would occur across the changing light conditions from the center of the patch cut to the edge was rejected. Based on these studied sites, yellow-poplar as a shade-intolerant species is much more adaptable to less sunlight than previously reported.

ACKNOWLEDGMENTS

Appreciation is expressed to Adam Ziegenbein and Molly Aldridge, foresters with the Tennessee Department of Agriculture, Division of Forestry at Natchez Trace State Forest for their guidance in providing study locations and records for this research, Canaan Dugger who assisted with data collection, and Sheng-I Yang and Charles Kwit at the School of Natural Resources, University of Tennessee for their research advisement.

LITERATURE CITED

Beck, D.E.; Della-Bianca, L. 1981. Yellow-poplar: characteristics and management. Agric. Handb. No. 583. Washington, DC: U.S. Department of Agriculture Forest Service. 92 p.

Clatterbuck, W.K. 2019. What do we know about oaks? Keystones of oak silviculture. In: Clark, S.L.; Schweitzer, C.J., eds. Oak symposium: sustaining oak forests in the 21st century through science-based management. e-Gen. Tech. Rep. SRS-237. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 3–12. https://doi.org/10.2737/SRS-GTR-237.

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

Fox, T.R.; Creighton, J. 2016. Silvics of oaks: their biology and ecology. In: Keyser, P.D.; Fearer, T.; Harper, C.A., eds. Managing oak forests in the Eastern United States. Boca Raton, FL: CRC Press: 51–61.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2009. The ecology and silviculture of oaks. 2nd ed. Wallingford, UK: Center for Agriculture Biosciences International. 580 p. https://doi.org/10.1079/9781845934743.0000.

LeDoux, C.B. 1999. An integrated approach for determining the size of hardwood group-selection openings. Forest Products Journal. 49(3): 34–37.

McClurkin, D.C.; Duffy, P.D.; Ursic, S.J.; Nelson, N.S. 1985. Water quality effects of clearcutting upper Coastal Plain loblolly pine plantations. Journal of Environmental Quality. 14(3): 329–332. https://doi.org/10.2134/jeq1985.00472425001400030005x.

McNab, W.H.; Oprean, T.M., III. 2021. Composition and structure of reproduction in group selection openings after 20 years in a southern Appalachian mixed-hardwood forest. Forest Science. 67(3): 335–346. https://doi.org/10.1093/forsci/fxab001.

Meadows, J.S.; Stanturf, J.A. 1997. Silvicultural systems for southern bottomland hardwood forests. Forest Ecology and Management. 90(2-3): 127–140. https://doi.org/10.1016/S0378-1127(96)03898-4.

Mueller-Dombois, D.; Ellenberg, H. 1974. Aims and methods of vegetation ecology. New York: John Wiley. 547 p.

Nyland, R.D. 2002. Silviculture: concepts and applications. 2nd ed. Long Grove, IL: Waveland Press. 682 p.

Sander, I.L. 1972. Size of oak advance reproduction: key to growth following harvest cutting. Res. Pap. NC-79. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Forest Experiment Station. 6 p.

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

Schweitzer, C.J.; Janzen, G.; Dey, D. 2016. Regenerating oak stands the “natural” way. In: Keyser, P.D.; Fearer, T.; Harper, C.A., eds. Managing oak forests in the Eastern United States. Boca Raton, FL: CRC Press: 75–84.

Smalley, G.W. 1991. Classification and evaluation of forest sites on 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. https://doi.org/10.2737/SO-GTR-85.

Yamasaki, M.; Costello, C.A.; Leak, W.B. 2014. Effects of clearcutting, patch cutting, and low-density shelterwoods on breeding birds and tree regeneration in New Hampshire northern hardwoods. Res. Pap. NRS-26. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station. 15 p. https://doi.org/10.2737/NRS-RP-26.

Shortleaf Pine

SITE PREPARATION FOR SHORTLEAF PINE CLUSTER PLANTING AND RESTORATION OF SHORTLEAF PINE-HARDWOOD MIXTURES: FIRST YEAR RESULTS

David Clabo and Michael Hinson

ABSTRACT

Shortleaf pine (*Pinus echinata*)-hardwood restoration continues to be a management goal throughout the interior South. Cluster or group planting involves placing groups of tightly spaced trees on favorable microsites or easily accessible areas in clearcut or partially harvested stands. Management operations for cluster planting may be focused on smaller areas reducing costs and easing access for areas with difficult topography. The objective of this study was to investigate site preparation methods to establish shortleaf pine using cluster planting in an upland mixed hardwood stand that had received a two-aged deferment harvest. The study area was located in Dade County, GA, near the summit of Sand Mountain. Four treatments were installed as three replications on 60-foot by 60-foot experimental units. All site preparation treatments included felling of understory hardwoods during late summer 2021. Treatments included (1) fell only (control); (2) fell and burn; (3) fell, herbicide, and burn; and (4) fell and herbicide. Bareroot, 1-0 stock shortleaf pine seedlings were planted during December 2021 at a 5-foot by 5-foot spacing. First-year results indicated better survival for treatments that included burning and ranged from 43.6 to 69.5 percent. Growth was greatest with the most intense treatment (fell-herbicide-burn). Naturally regenerating oak species (*Quercus* spp.) and other desirable hardwoods and natural shortleaf pine did not differ in stem densities across treatments, while red maple (*Acer rubrum*) and gums (*Nyssa sylvatica* and *Liquidambar styraciflua*) were plentiful in the least intensive treatments. Future release treatments will be needed to promote shortleaf pine and improve hardwood species composition.

INTRODUCTION

Restoration is a management goal throughout the 23-state native range of shortleaf pine (*Pinus echinata*). Shortleaf pine often occurs in mixedwoods or pine-hardwood mixtures with hardwoods [preferably oaks (*Quercus* spp.), hickories (*Carya* spp.), and other economically or ecologically important hardwood species] and other pines where shortleaf or hardwoods may compose as much as 80 percent of stand stocking (Anderson and others 2016, Deal 2018, Guldin and Black 2018). Many suitable sites for shortleaf pine in the eastern half of its range either lack a shortleaf pine seed source, or have scattered, older, and less vigorous remnant trees (Moser and others 2007). Conditions in stands like these can cause difficulties with natural regeneration of shortleaf pine, making artificial regeneration an important option.

A form of artificial regeneration known as underplanting can be used to help restore shortleaf pine in stands where hardwoods have become dominant along with prescribed fire, herbicide applications, thinnings or harvests that allow more sunlight into the understory. Underplanting may be an attractive option for restoration efforts as residual overstory shade can slow fast-growing intolerant hardwood species, shortleaf pine is moderately shade tolerant as a seedling, and vertical structure and potential mast-producing trees are left in the stand for wildlife (Kabrick and others 2015, Schnake and others 2021). Cluster planting is a form of underplanting where seedlings are selectively planted within suitable microsites (often <0.25

David Clabo, Assistant Professor, University of Georgia, Warnell School of Forestry and Natural Resources, Tifton, GA 31793
Michael Hinson, Stewardship Forester, Georgia Forestry Commission, Dry Branch, GA 31020

acres), canopy openings, or easily accessible locations to improve stocking with desirable species (Saha and others 2017, Schönenberger 2001). The location and number of clusters across a stand can be adjusted to obtain stocking and restoration goals. Within clusters, tight spacings (often less than 6 feet by 6 feet) are used to accelerate natural pruning and intraspecific competition (and potentially interspecific competition for mixed plantings or if natural hardwood regeneration is allowed to develop) of planted trees, with the expectation of natural thinning as clusters develop. Cluster planting has previously been tested as a restoration technique for shortleaf pine-hardwood mixedwoods in the Cumberland Plateau region with promising early results (Clabo and Clatterbuck 2020b). The purpose of this study was to expand on those results by investigating other techniques to improve survival and development of shortleaf pine and beneficial hardwoods.

The objectives of this study were to assess first-year survival and growth of underplanted shortleaf pine following various site preparation treatments in a two-aged mixed hardwood stand, and to determine if natural regeneration species composition and densities can be improved by various site preparation practices.

MATERIALS AND METHODS

The study area was located on privately owned land in Dade County, GA (34.947941, -85.491797). This area is part of the Cumberland Plateau physiographic province near the summit of Sand Mountain. The landform is characterized by narrow ridges and convex upper slopes with elevations ranging from 1,250 to 1,290 feet and a predominately southeast aspect (Smalley 1979). Soils were characterized as Hector Complex with 5- to 60-percent slopes. These shallow soils are classified as loamy, siliceous, subactive, thermic Lithic Dystrudepts, and rock composes up to 50 percent of the A horizon by volume (USDA NRCS 2023). The area averages 51.9 inches of rainfall annually (UGA CAES 2023).

Stand conditions prior to study establishment consisted of an average of 21 square feet per acre of overstory basal area consisting primarily of white oak (*Quercus alba*), chestnut oak (*Quercus montana*), black oak (*Quercus velutina*), mockernut hickory (*Carya tomentosa*), shortleaf pine, and blackgum (*Nyssa sylvatica*) >6 inches in diameter at breast height (d.b.h.). Approximately 5 years prior to study establishment, the stand received a deferment harvest to establish a two-age system, a shelterwood with reserves structure (Stringer 2006). During the interim 5 years prior to study establishment, approximately 2,000–5,000 hardwood seedlings and saplings per acre developed in the understory ranging from 1 to 15 feet tall. Regenerating species composition consisted of white oak, post oak (*Quercus stellata*), chestnut oak, black oak, scarlet oak (*Quercus coccinea*), blackgum, red maple (*Acer rubrum*), yellow-poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*) mockernut hickory, sand hickory (*Carya padilla*), shortleaf pine, *Vaccinium* spp., sumac (*Rhus* spp.), sassafras (*Sassafras albidum*), winged elm (*Ulums alata*), black birch (*Betula lenta*), dogwood (*Cornus florida*), and black cherry (*Prunus serotina*).

Study installation began during July 2021 when twelve 60-foot by 60-foot cluster planting plots were established along the slope contour. One of four treatments were randomly assigned to clusters but burn treatment clusters were established in close proximity to one another due to required installation of a firebreak around these treatments. Treatments consisted of fell only (control), fell and burn, fell and herbicide, and fell, herbicide, and burn. All woody stems, woody vines and brambles >6 inches tall within clusters and 5 feet outside of clusters were cut as close to ground level as possible during September. This treatment was done to simulate stand conditions immediately after the overstory deferment harvest. For the herbicide treatments, a cut stump herbicide application was used within 15 minutes after woody stem, vine, or bramble cutting. A 4-pound acid equivalent Triclopyr ester herbicide (Alligare® Triclopyr 4, Opelika, AL) was used at a 25-percent rate in basal oil and applied to all sides of cut stumps. The burn treatment was completed November 9, 2021 using a backing fire ignited downslope. Burn coverage was 100 percent within these clusters. Bareroot, second-cycle shortleaf pine seedlings were hand planted December 27, 2021, by a professional planting crew. Trees were planted at a 5-foot by 5-foot spacing totaling 156 trees per cluster.

Permanent sampling plots were established with six rows of six trees located in the center of each cluster. The first and last tree of each row were tagged. Baseline shortleaf pine survival and height were recorded prior to the start of the growing season. In addition, seedling damage that could be attributed to browse or freeze damage was recorded. At the center of the shortleaf pine sampling plots, a circular 0.01-acre natural regeneration plot was established and permanently marked. Photosynthetically active radiation (PAR) was measured to ensure that understory light conditions were similar across clusters, and to improve shortleaf pine and natural regeneration development comparisons among treatments. During August of the first growing season, PAR measurements were taken on a uniformly overcast day within 2 hours of solar noon (Parent and Messier 1996). In-stand point PAR measurements were taken with a LP-80 LAI ceptometer (Meter Group®, Pullman, WA) with an external calibration sensor located at d.b.h. height at the four corners of each cluster facing the nearest ordinal direction. Open PAR measurements were taken approximately 500 feet from the study area in a large food plot opening. Open sky measurements were taken with a PAR photon flux sensor (Meter Group®, Pullman, WA) operated by a ZL6 data recorder (Meter Group®, Pullman, WA) calibrated to automatically record PAR measurements every 5 minutes. In-stand measurements were then paired with the corresponding time an open sky measurement at each timestamp to give an estimate of percent full sunlight.

Shortleaf pine and natural regeneration were assessed during the 2022–2023 winter. For shortleaf pine, survival, groundline diameter, total height, and presence of obvious damage were recorded. Within the natural regeneration sampling plots, information for all woody species >6 inches tall was recorded, including species and height class to the nearest foot. One-way analysis of variance (ANOVA) with a completely randomized design and treatment or location (direction) within a cluster as the fixed factor was used to analyze percent full sunlight across treatments. Statistically similar sunlight levels would indicate that comparisons of survival and growth across treatments were appropriate. The same analysis with only treatment as a fixed factor was used to assess planted shortleaf pine survival and height (h) before and after the first growing season, change in height growth, groundline diameter (gld) post-first growing season, and volume index (vi) as in Spurr (1952):

$$vi = gld^2 \times h \quad (1)$$

All clusters did not have the same regenerating species so species groups were assigned for analysis (Clabo and Clatterbuck 2020b). Groups consisted of oaks (denoted as OAK—chestnut, white, post, scarlet, and black oak), gums (GUM—blackgum and sweetgum), red maple (RM), undesirable (UND) hardwood tree and shrub species (winged elm, *Vaccinium* spp., sumac spp., sassafras, black birch, dogwood, and black cherry), while the group with other desirable (DES) hardwood species and pine consisted of yellow-poplar, hickory species, and shortleaf pine. An ocular percent cover assessment (5-percent intervals) of grape vine (*Vitis* spp.) and blackberry (*Rubus* spp.) was also completed for each plot. Natural regeneration data were analyzed using a two-factor ANOVA in a completely randomized design with treatment and species group as fixed factors. All analyses were completed using Proc Mixed in SAS 9.4 (SAS Institute 2012). Least squares means were separated using Fisher's protected least significant difference and an alpha of 0.10. The 10-percent significance level was chosen to reduce Type II error rate due to low replication, and to increase the power to detect a treatment difference that could be used as a practical site preparation alternative for sites with similar vegetation (Remington and Schork 1985). The shortleaf pine volume index and natural regeneration stem density and height by species group analyses were log transformed and untransformed means and standard errors are reported.

RESULTS

Percentage of full sunlight results revealed no significant differences among treatments (P = 0.401) or by location within clusters (P = 0.601), with percentage of full sunlight ranging from 54.8 to 68.5 across treatments and 56.6 to 62.0 by location within a cluster. No survival differences prior to the first growing season were detected (P = 0.598), and survival ranged from 95.6 to 98.7

percent, though 12.6 percent of live seedlings had evidence of freeze or browse damage. Survival differences did occur after the first growing season (P = 0.074). Survival was least in the fell-only (control) treatment and greatest in the fell-herbicide-burn treatment (table 1). After the first growing season, 33.6 percent of live seedlings had evidence of browse damage or insect defoliation (fig. 1). Treatment differences for average height at planting did occur (P <0.0001) (range 7.9 inches for fell-herbicide-burn to 9.4 inches for the fell-only treatment), but these differences were not evident by the end of the first growing season because height did not differ among treatments (P = 0.909) and ranged from 11.6 to 11.8 inches. Differences in first growing season height growth did occur (P = 0.067) (table 1), with greatest growth occurring in the fell-herbicide-burn treatment. Differences in first growing season groundline diameter growth occurred (P = 0.008) (table 1). Growth was greatest in the fell-herbicide-burn treatment and least in the fell-only (control) treatment. Shortleaf pine average volume index differences were detected (P = 0.056) as well with the fell-herbicide-burn treatment having greater average volume index than the fell-only (control) and fell-herbicide treatments which were statistically similar (table 1).

Table 1—Mean survival, height growth, groundline diameter, and volume index for planted shortleaf pine one growing season after planting for a shortleaf pine cluster planting site preparation study

	Survival		Height		Groundline diameter		Volume index	
Treatment	Mean	SE	Mean	SE	Mean	SE	Mean	SE
	percent		*inches*		*inches*			
Fell, burn, and herbicide	69.5a	6.2	4.0a	0.36	0.19a	0.005	781.3a	66
Fell and burn	64.9a	6.6	3.4ab	0.36	0.18a	0.007	756.9ab	67
Fell only	43.6b	6.1	2.4b	0.39	0.15b	0.005	532.0b	73
Fell and herbicide	58.4ab	6.0	3.1ab	0.34	0.16b	0.004	532.9b	63

SE = standard error of the mean.
Treatment means in a column followed by the same letter do not differ statistically at P = 0.10.

Figure 1—Examples of shortleaf pine (left) freeze and (right) browse damage approximately 2 months after planting. (Photos courtesy of David Clabo)

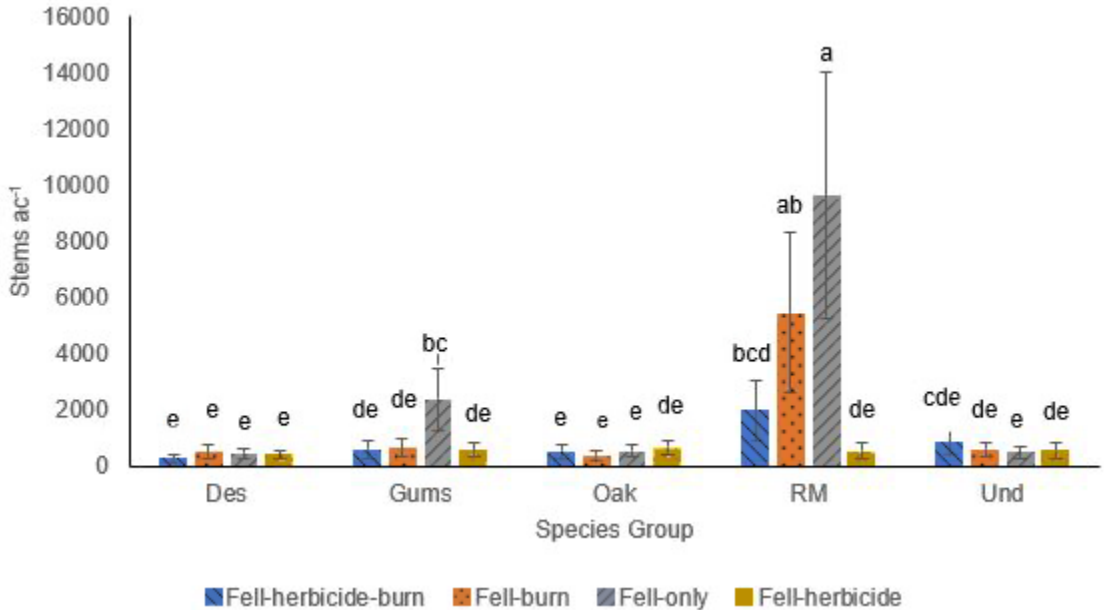

Figure 2—Natural regeneration stem density averages and standard error bars. DES is yellow-poplar, hickory, and shortleaf pine; GUM is blackgum and sweetgum; OAK is chestnut, white, post, scarlet, and black oak; RM is red maple; and UND is winged elm, *Vaccinium* spp., sumac, sassafras, black birch, dogwood, and black cherry. Bars with similar letters are not statistically different at $P = 0.10$.

Table 2—Mean height of natural regeneration one growing season after study establishment for a shortleaf pine cluster planting site preparation study

	Stem height	
Treatment	Mean	SE
	~~ *inches* ~~~~~~~	
Fell, burn, and herbicide	15.8b	6.5
Fell and burn	42.1a	5.6
Fell only	21.6b	6.4
Fell and herbicide	12.1b	5.6

Treatment means in a column followed by the same letter do not differ significantly at $P = 0.10$.
SE = standard error of the mean.

The natural regeneration stem density treatment by species group interaction was significant ($P = 0.012$) (fig. 2). Red maple was prevalent in all treatments except the fell-herbicide-burn treatment. Gum species were common in the fell-only treatment, while the OAK and DES groups did not differ in stem density across treatments (fig. 2). Nine of 12 clusters had greater than 10 percent grape or blackberry coverage at the end of the first growing season. Only treatment was significant ($P = 0.013$) for the natural regeneration height analysis. Stems in the fell-burn treatment were significantly taller than the three other treatments, which did not differ (table 2).

DISCUSSION

The light analyses revealed that treatments and clusters did not differ in percentage of full sunlight, which simplified comparisons of shortleaf pine and natural regeneration variables across treatments. Growing season light levels in this study were similar 5 years after harvest to a similar 25 percent retention level (38 square feet per acre) shelterwood study (66.7 percent full light) conducted across three sites in neighboring Jackson County, AL (Schweitzer and Dey 2011).

Shortleaf pine survival was still acceptable for cluster planting restoration purposes across all site preparation treatments following one growing season, but was poorer across treatments than a similar underplanting study (76 percent survival after one growing season) conducted in Missouri (Kabrick and others 2011). Due to the high planting densities within clusters, greater early mortality may be acceptable with cluster planting below minimum stocking thresholds. Site preparation may offer better early survival than release treatments. Site preparation tended to result in better survival after one growing season for all treatments except the fell-only treatment compared to similar shortleaf pine cluster release treatments after two growing

seasons conducted by Clabo and Clatterbuck (2020b) on similar sites in Tennessee. More time will indicate if that trend continues. Treatments that included site preparation burning usually had better survival and larger seedlings than fell-only or fell-herbicide treatments. Site preparation burning has been recognized as important for improved long-term shortleaf pine growth in other studies (Gwaze and others 2007). Average growth with the fell-herbicide treatment tended to be less than with the fell-herbicide-burn treatment, similar to findings by Clabo and Clatterbuck (2020a). Burning appears to give added vegetation control during the first growing season, especially compared to herbicide treatments that only target woody and semiwoody vegetation stumps.

Oak stem densities did not differ across treatments after one growing season. This result should be expected as other studies have not shown oak stem density increases due to sprouting or new seedling establishment following cutting and prescribed fire especially when aggressive mesic species such as red maple, blackgum, and sassafras are present (Schweitzer and others 2016). Herbicide treatments also would have reduced oak stem densities in those treatments where they were applied. Sprout height and density of red maple stems increased significantly with the fell-only and fell-burn treatments. Red maple sprouts can grow quickly compared to competitor species in relatively high light and drought-prone sites similar to the sites used in this study (Abrams 1998). Other studies have reported increases in stem density and height of mesic species such as red maple and blackgum following a single prescribed burn due to prolific sprouting (Arthur and others 1998, Elliott and Vose 2005). The rapid sprout growth of species such as red maple and sassafras following cutting and topkill by prescribed fire along with more open canopy conditions associated with the two-age deferment harvest likely explains the significantly greater height growth of natural regeneration (predominately red maple) in the fell-burn treatment (Alexander and others 2008, Arthur and others 1998, Fei and Steiner 2009, Schweitzer and others 2016). The mesic species that have established will likely require release treatments to improve ingrowth to the canopy of shortleaf pine, oaks, and other desirable hardwoods (Guldin 2019, Schweitzer and Dey 2011).

CONCLUSIONS

Shortleaf pine underplanting in clusters is a potential management tool to restore scattered pockets of shortleaf pine in hardwood-dominated systems. More intensive site preparation that includes herbicide application and prescribed fire can improve early pine survival and growth. Due to shortleaf pine's slow growth after planting, added release treatments to suppress fast-growing competitor hardwood species and vines may be needed to foster shortleaf pine's competitive status on similar sites until shortleaf pine begins more rapid height growth 3–4 years after planting.

LITERATURE CITED

Abrams, M.D. 1998. The red maple paradox. BioScience. 48(5): 355–364. https://doi.org/10.2307/1313374.

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. https://doi.org/10.1016/j.foreco.2008.06.004.

Anderson, M.; Hayes, L.; Keyser, P.D. [and others]. 2016. Shortleaf pine restoration plan: restoring an American forest legacy. Shortleaf Pine Initiative. 57 p. https://shortleafpine.org/shortleaf-pine-initiative/shortleaf-pine-restoration-plan/shortleaf-pine-restoration-plan. [Date accessed: November 1, 2023].

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. https://doi.org/10.2307/2997220.

Clabo, D.C.; Clatterbuck, W.K. 2020a. Establishment and early development of even-age shortleaf pine-hardwood mixtures using artificially regenerated shortleaf pine and various site preparation and release treatments. Forest Science. 66(3): 351–360. https://doi.org/10.1093/forsci/fxz082.

Clabo, D.; Clatterbuck, W. 2020b. Restoration of shortleaf pine (*Pinus echinata*)-hardwood mixtures in low quality mixed upland hardwood stands using cluster planting and natural regeneration. Forests. 11: 457. 18 p. https://doi.org/10.3390/f11040457.

Deal, R., ed. 2018. Dictionary of forestry. Bethesda, MD: Society of American Foresters. 208 p.

Elliott, K.J.; Vose, J.M. 2005. Effects of understory prescribed burning on shortleaf pine (*Pinus echinata* Mill.)/mixed-hardwood forests. Journal of the Torrey Botanical Society. 132(2): 236–251. https://doi.org/10.3159/1095-5674(2005)132[236:EOUPBO]2.0.CO;2.

Fei, S.; Steiner, K.C. 2009. Rapid capture of growing space by red maple. Canadian Journal of Forest Research. 39: 1444–1452. https://doi.org/10.1139/X09-065.

Guldin, J.M. 2019. Restoration of native fire-adapted southern pine-dominated forest ecosystems: diversifying the tools in the silvicultural toolbox. Forest Science. 65(4): 508–518. https://doi.org/10.1093/forsci/fxz005.

Guldin, J.M.; Black, M.W. 2018. Restoration of shortleaf pine in the Southern United States—strategies and tactics. In: Kirschman, J.E., comp. Proceedings of the 19th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-234. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 281–287.

Gwaze, D.; Melick, R.; McClure, L. [and others]. 2007. Effects of site preparation subsoiling 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.

Kabrick, J.M.; Dey, D.C.; Shifley, S.R.; Villwock, J.L. 2011. Early survival and growth of planted shortleaf pine seedlings as a function of initial size and overstory stocking. In: Fei, S.; Lhotka, J.M.; Stringer, J.W. [and others], eds. Proceedings of the 17th central hardwood forest conference. Gen. Tech. Rep. NRS-P-78. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 277–287.

Kabrick, J.M.; Knapp, B.O.; Dey, D.C.; Larsen, D.R. 2015. Effect of initial seedling size, understory competition, and overstory density on the survival and growth of *Pinus echinata* seedlings underplanted in hardwood forests for restoration. New Forests. 46: 897–918. https://doi.org/10.1007/s11056-015-9487-3.

Moser, W.K.; Hansen, M.; McWilliams, W.H.; Sheffield, R.M. 2007. Shortleaf pine composition and structure in the United States. In: Kabrick, J.M.; Dey, D.C.; Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 19–27.

Parent, S.; Messier, C. 1996. A simple and efficient method to estimate microsite light availability under a forest canopy. Canadian Journal of Forest Research. 26: 151–154. https://doi.org/10.1139/x26-017.

Remington, R.D.; Schork, M.A. 1985. Hypothesis testing. In: Statistics with applications to the biological and health sciences. Englewood Cliffs, NJ: Prentice Hall: 165–204.

Saha, S.; Kuehne, C.; Bauhus, J. 2017. Lessons learned from oak cluster planting trials in central Europe. Canadian Journal of Forest Research. 47: 139–148. https://doi.org/10.1139/cjfr-2016-0265.

SAS Institute. 2012. SAS/STAT user's guide. Version 9.4. Cary, NC: SAS Institute. 824 p.

Schnake, D.K.; Roberts, S.D.; Willis, J.L. [and others]. 2021. Overstory retention and stock type impact survival and growth of underplanted shortleaf pine beneath a hardwood canopy. Forest Science. 67(2): 219–230. https://doi.org/10.1093/forsci/fxaa046.

Schönenberger, W. 2001. Cluster afforestation for creating diverse mountain forest structures—a review. Forest Ecology and Management. 145: 121–128. https://doi.org/10.1016/S0378-1127(00)00579-X.

Schweitzer, C.J.; Dey, D.C. 2011. Forest structure, composition, and tree diversity response to a gradient of regeneration harvests in the mid-Cumberland Plateau escarpment region, USA. Forest Ecology and Management. 262: 1729–1741. https://doi.org/10.1016/j.foreco.2011.07.020.

Schweitzer, C.J.; Dey, D.C.; Wang, Y. 2016. Hardwood-pine mixedwoods stand dynamics following thinning and prescribed burning. Fire Ecology. 12(2): 85–104. https://doi.org/10.4996/fireecology.1202085.

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

Spurr, S.H. 1952. Forest inventory. New York: Ronald Press Company. 476 p.

Stringer, J. 2006. Two-age system and deferment harvests. Extension Publ. SP679. Knoxville, TN: University of Tennessee, Agricultural Extension Service. 12 p.

University of Georgia, College of Agricultural and Environmental Sciences [UGA CAES]. 2023. UGA Weather Network. Athens, GA: University of Georgia. http://www.georgiaweather.net/mindex.php?variable=AV&site=LAFAYET. [Date accessed: February 2, 2023].

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2023. Web soil survey. https://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm. [Date accessed: February 2, 2023].

SUSTAINABLE PATHWAYS FOR SHORTLEAF PINE RESTORATION IN UNCERTAIN CLIMATES

Casey Iwamoto, Courtney Siegert, Joshua Granger, Krishna Poudel, Adam Polinko, and Zachary Freedman

EXTENDED ABSTRACT

Strip mining has degraded soil throughout the Southeastern United States. To enhance the probability of success in forest restoration, combinations of organic and microbial soil amendments can be utilized to improve soil health and water availability (Blanco-Canqui 2017). However, climate change may challenge the effectiveness of restoration. In this study, shortleaf pine (*Pinus echinata*) was used because its current range is shrinking due to disease, climate change, and the preferential planting of other pine species (Sutter 2019). Additionally, shortleaf pine prefers acidic soil and has been observed on postmined sites in the past (Lawson 1990).

We conducted a comprehensive greenhouse experiment using shortleaf pine seedlings treated with three different moisture regimes to understand potential impacts of precipitation uncertainty due to climate change. In a greenhouse, shortleaf pine seedlings were planted in replicates across moisture treatments receiving the following amendments: organic, microbial, organic and microbial mixture, unamended control, or control with no trees. Soil was sourced and homogenized from a postmined site in Alabama. Tree growth parameters (survival, groundline diameter, height, root-to-shoot ratio), soil-water fluxes (dissolved organic carbon, absorbance at 254 μm, specific ultraviolet absorbance at 254 nm), and soil health indicators (pH, carbon, and nitrogen content) were measured throughout the study.

After 4 months, survival was 62 percent across treatments. Across water regimes, the unamended control, organic and microbial mixture, and organic soil amendments saw the largest increases in average height across water regimes. Across soil treatments, mean height increased from 20.01 ±

Casey Iwamoto, Graduate Student, Mississippi State University, Department of Forestry, Starkville, MS 39762

Courtney Siegert, Associate Professor, Mississippi State University, Department of Forestry, Starkville, MS 39762

Joshua Granger, Assistant Professor, Mississippi State University, Department of Forestry, Starkville, MS 39762

Krishna Poudel, Assistant Professor, Mississippi State University, Department of Forestry, Starkville, MS 39762

Adam Polinko, Assistant Professor, Mississippi State University, Department of Forestry, Starkville, MS 39762

Zachary Freedman, Assistant Professor, University of Wisconsin-Madison, Department of Soil Science, Madison, WI, 53706

1.57 (mean ± standard error) cm to 27.33 ± 1.52 cm in the dry water regime, from 22.37 ± 0.67 cm to 28.37 ± 3.61 cm in the average water regime, and from 22.14 ± 0.43 cm to 30.17 ± 1.37 cm for the wet water regime.

The highest specific ultraviolet absorbance at 254 nm occurred in the organic treatment in the wet regime at 9.239 ± 0.533 L $mg^{-1}m^{-1}$ and the lowest occurred in the microbial amendment treatment in the wet regime at 2.424 ± 0.125 L $mg^{-1}m^{-1}$ (fig. 1). Further tree growth, soil-water flux, and soil indicator results are pending. Results from this greenhouse experiment will inform current restoration techniques using organic and microbial soil amendments.

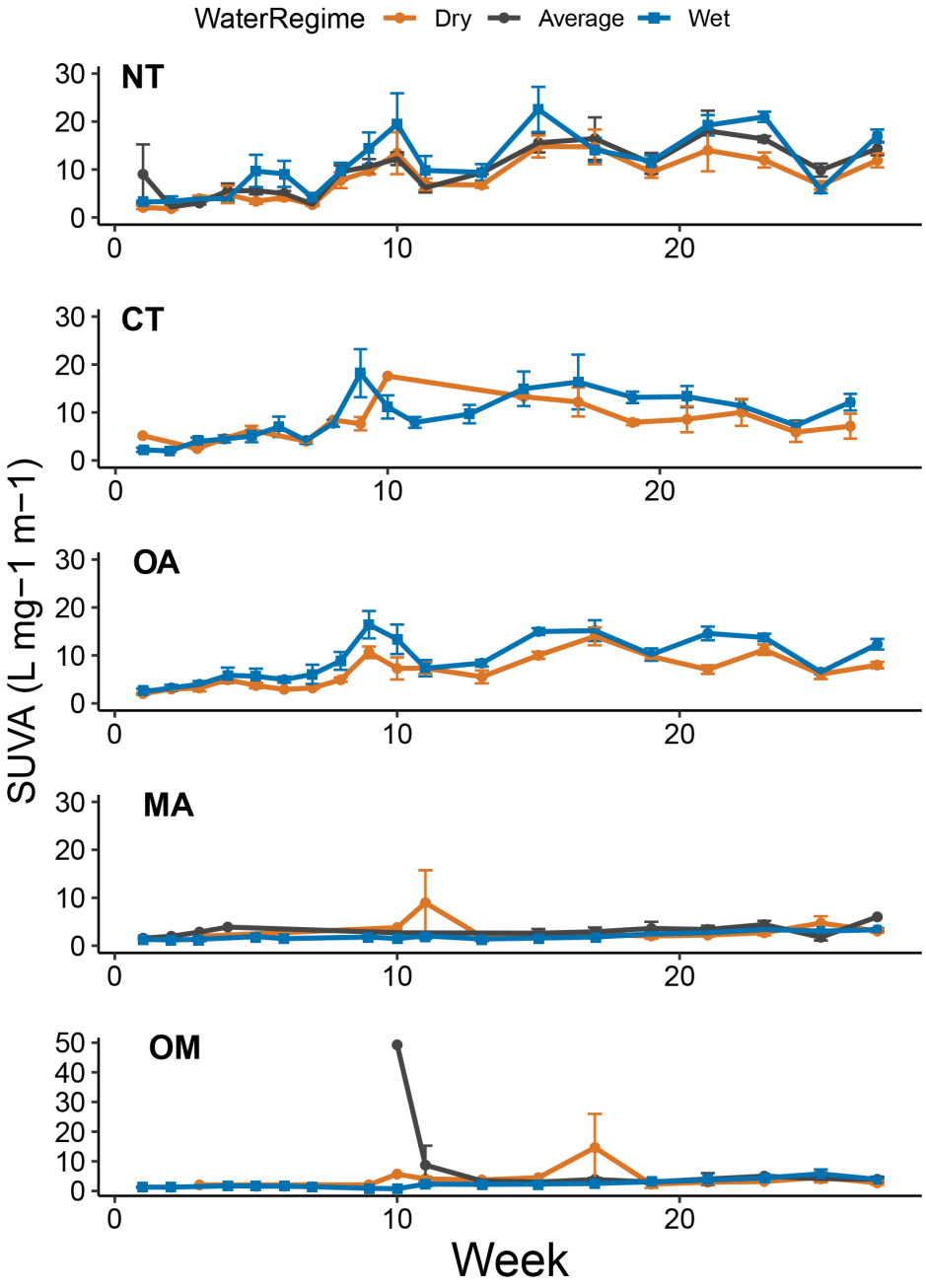

Figure 1—Specific UV absorption at 254 nm (SUVA) across the soil amendments by week. NT = control with no trees; CT = unamended control; OA = organic amendment; MA = microbial amendment; OM = organic and microbial amendment. Note that OM has a larger y-axis range compared to other soil treatments.

LITERATURE CITED

Blanco-Canqui, H. 2017. Biochar and soil physical properties. Soil Science Society of America Journal. 81: 687–711. https://doi.org/10.2136/sssaj2017.01.0017.

Lawson, E.R. 1990. *Pinus echinata* Mill. Shortleaf pine. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. Vol. 1. Conifers. Agric. Handbook 654. Washington, DC: U.S. Department of Agriculture Forest Service: 316–326.

Sutter, R.D. 2019. Ecological and social history of shortleaf pine. SREF-SLP-020. Athens, GA: Southern Regional Extension Forestry. 7 p.

GROWTH RESPONSE TO COMPETITION AND PRESCRIBED FIRE IN PLANTED SHORTLEAF PINE

Hope M. Fillingim, Benjamin O. Knapp, Daniel C. Dey, Michael C. Stambaugh, John M. Kabrick, and Grant P. Elliott

ABSTRACT

Shortleaf pine (*Pinus echinata*) is an economically and ecologically important species that was once prevalent across the Southeastern United States, and there is growing interest in its restoration throughout its former range. One challenge with restoring shortleaf pine is competition from hardwoods. This study used tree-ring analysis to study growth response of shortleaf pines planted following a 2002 tornado in two stands with different known histories of management activities, specifically prescribed burning and mechanical release, across a range of competition levels. Early release was effective in increasing radial growth of shortleaf pine in this study, and competition, especially from hardwoods, caused decreased annual growth. There was only minimal evidence for direct reductions in shortleaf growth following burning, and fire seemed to effectively control hardwood competition in the midstory. These findings provide insight into the roles of prescribed fire and hardwood competition on growth of planted shortleaf pine throughout its early life.

INTRODUCTION

Shortleaf pine (*Pinus echinata*) covered approximately 300,000 km^2 across what is now the Southeastern United States prior to European colonization (Anderson and others 2016) but has been reduced to approximately 25,000 km^2 currently, or less than 10 percent of the historical estimate (Oswalt 2012). Restoring shortleaf pine and shortleaf pine-oak (*Quercus* spp.) mixedwood forests is challenging due to issues with recruiting shortleaf pine from the forest understory to competitive positions in the midstory, and ultimately to the overstory. Although planting shortleaf pine seedlings can provide a solution to regeneration shortages caused by potential problems with seed production or germination, newly planted shortleaf pine seedlings do not compete well with hardwoods (Kabrick and others 2015), often because the hardwood stems have larger advance reproduction or have recently resprouted from an established root system.

Removing understory hardwoods can prove to be an effective release treatment for shortleaf pine (Brinkman and Smith 1968). In a study in Arkansas, a shortleaf pine stand with the hardwood understory removed had greater growth over 5 years than stands with the understory left intact (Bower and Ferguson 1968). In a similar study at the Sinkin Experimental Forest in Missouri, hardwood control increased shortleaf pine growth by 2 m^2/ha over 10 years (Rogers and Brinkman 1965).

Prescribed burning provides another option to suppress hardwood competition in understory and midstory layers. Properly timed prescribed fire may benefit shortleaf pine growth rates and recruitment into the midstory by top-killing less fire-tolerant hardwood species. For example, a study in the Ouachita National Forest examined changes in the fire regime over time using

Hope M. Fillingim, Graduate Student, School of Natural Resources, University of Missouri, Columbia, MO 65211
Benjamin O. Knapp, Associate Professor, School of Natural Resources, University of Missouri, Columbia, MO 65211
Dan C. Dey, Assistant Director of Research, U.S. Department of Agriculture, Forest Service, Northern Research Station, Columbia, MO 65211
Michael C. Stambaugh, Associate Professor, School of Natural Resources, University of Missouri, Columbia, MO 65211
John M. Kabrick, Research Forester, U.S. Department of Agriculture, Forest Service, Northern Research Station, Columbia, MO 65211
Grant P. Elliott, Professor, Department of Geography, University of Missouri, Columbia, MO 65211

dendrochronological and fire scar evidence, and found changes in species composition of recruiting trees between the "European American settlement period," with fire return interval of 3–7 years, and the "Fire Protection period," with fire return intervals of 42 years, with the composition shifting from primarily shortleaf pine (85 percent) with more frequent fires, to primarily oaks (68 percent) and other hardwoods (31 percent) with the longer fire intervals (Flatley and others 2023). Similarly, in a 53-year study of frequently burned loblolly (*P. taeda*) and shortleaf pine woodlands in northern Florida, 1- to 2-year burn intervals maintained pine dominance, while hardwoods gained dominance in stands with a 3-year fire return interval (Robertson and others 2021). Based on these findings, the authors suggested selecting a fire-return interval aligned with the specific site conditions, land-use histories, and stand objectives (Robertson and others 2021). Once shortleaf pine stems are large enough that vulnerability to top-kill is low, repeated prescribed burning can be used as a maintenance treatment to limit the abundance of hardwood stems in the midstory and keep stand conditions in an open structure. In uneven-aged loblolly-shortleaf pine stands in southeastern Arkansas, burn intervals of 3, 6, and 9 years were compared with nonburned areas. While there was no difference in the basal area of the saplings [2.5–8.9 cm diameter at breast height (d.b.h.)] across treatments, there was significantly less sapling diversity in the 3-year burn interval than in the other treatments, which was explained by the elimination of fire-sensitive species (Cain and others 1998).

However, there remain challenges to using prescribed fire to control hardwoods and boost shortleaf pine growth. In pine-oak forests in the southern Appalachians, a single March burn reduced the basal area of fire-sensitive midstory hardwoods, but the density (stems per area) of these hardwoods increased due to prolific sprouting, suggesting that repeated burns were warranted (Elliott and Vose 2005). Furthermore, though reducing hardwood competition with prescribed burning can increase shortleaf pine growth, a recent study in southeast Oklahoma found that radial growth of shortleaf pine trees was reduced by 21–33 percent in the growing season following prescribed fire (Adhikari and others 2021), suggesting that fire may also directly negatively affect tree growth.

Improving our understanding of the effects of both prescribed burning and hardwood competition on shortleaf pine growth and overstory recruitment is important for management of this species. The objective of this study was to use tree-ring analysis to examine the radial growth of shortleaf pines in response to known silvicultural treatments and local competition levels. Our research questions were: 1) How does repeated prescribed burning affect stand development (structure and composition) following release of planted shortleaf pine? 2) How does shortleaf pine radial growth respond to release? 3) Does prescribed burning affect the radial growth of shortleaf pine? 4) Does the level and composition of local competition affect the radial growth of shortleaf pine?

METHODS

Study Site

The study was located in the Poplar Bluff Ranger District of the Mark Twain National Forest in southeastern Missouri. This study focused on two adjacent management units, separated by a gravel road, that were affected by a tornado in 2002 and salvaged in 2003. Scattered residual overstory trees consisted primarily of shortleaf pine and white oak (*Quercus alba*). A site preparation burn was conducted in both units in 2004, and bareroot shortleaf pine seedlings were planted in spring of 2005 at 3-m by 3-m spacing. A release treatment was completed in both units between June and December 2011. The release prescription was to cut any hardwood stems within 1.83 m of a shortleaf pine except those that were less than half the height of the shortleaf pine. In one of the two units, prescribed burns were conducted in March 2011, April 2013, March 2015, April 2018, and March 2021. The burned and nonburned units were approximately 5 ha and 7 ha, respectively.

Soil series for both units are mapped as predominately (93 percent) Captina silt loam (fine-silty, siliceous, active, mesic Typic Fragiudults), which are very deep, moderately well-drained

soils (USDA NRCS 2017). The units are nearly level, with slopes of 1–8 percent. The average annual temperature from 2005 to 2021 was 14 °C and the average annual precipitation for the same period was 1,286 mm (NOAA 2022).

Experimental Design

In each of the two units, henceforth called burned and nonburned despite both having a site preparation burn, 30 shortleaf pine trees were selected as focal trees to quantify growth patterns in relation to the competitive environment. Around these trees the overstory and midstory basal area were measured, and understory stems that reached at least breast height were tallied.

Tree Selection and Overstory Sampling

In the summer of 2022, four or five parallel transects were established in each management unit, with sampling points set every 15 m for a total of 45 sampling points per unit. The closest overstory (d.b.h. ≥11.4 cm) shortleaf pine to each point was selected as a potential focal tree and d.b.h. was measured. All overstory trees within a 7.3-m radius [3x the average crown radius, as recommended by Lorimer (1983)] of the focal pine were also identified and d.b.h. recorded.

The level of local competition surrounding each potential focal tree was quantified using the following competition index (CI): $(\Sigma D_j)/D_i$, where D_j is the diameter of the competitor and D_i is the diameter of the focal tree (Lorimer 1983).

The competition index measured across the initial 45 potential focal trees for each unit ranged from 1.5 to 23.5, with fewer than 10 trees having CIs >15. From this range, 6 competition classes were established, with the goal of randomly selecting 5 focal trees from each stand per class for a total of 30 focal trees per stand covering a range of competition levels. The CI classes were <5; 5–7.5; ≥7.5–10; ≥10–12.5; ≥12.5–15; and >15. If a CI class did not have sufficient trees to select from, additional trees from the closest class were randomly selected. In the burned unit, the two highest competition classes had insufficient representation, so all available trees within the classes were used and an additional tree from each of the three smaller competition classes were used. In the nonburned unit, only three trees were included in the 12.5–15 CI class, so all three were used and one extra tree from the class immediately higher and lower were used.

Midstory and Understory Sampling

Within a 7.3-m radius of the 30 focal trees per unit, the midstory trees (d.b.h. 2.5–11.4 cm) were identified and d.b.h. was measured. The understory trees (≥1.37 m tall and <2.5 cm d.b.h.) were also identified and tallied by species within the same radius.

Tree Coring

Two cores were taken from each focal tree, as close to the base of the trees as possible. All tree cores were glued to wooden core mounts and sanded with increasingly fine (up to 1,200 grit) sandpaper. Images of each core were scanned at 2,400 dpi and imported into the CooRecorder software (Cybis Electronik & Data AB, Sweden) for demarking ring boundaries and measuring annual ring widths. The best core per tree was used for analysis. Criteria for selecting the best core included a core's proximity to or inclusion of the pith, the clarity of ring boundaries, and the absence of damage that interfered with ring-width measurements. The distance to pith was estimated in the CooRecorder software for each core used in analysis, and ring widths were converted to basal area increment (BAI) for analyses.

There are several methods for detecting release events using tree-ring data. Several of these are reviewed in Trotsiuk and others (2018). One early method is called radial growth averaging, which was first described by Lorimer (1985) and uses the procedure of averaging the radial growth over the 15 years preceding and following the potential release event and then calculating the percent growth change by subtracting the pre-event average from the postevent average and dividing that by the pre-event average. Lorimer (1985) defined an event as a major release if the percentage growth change was ≥100 percent and

a defined a moderate release as a percentage growth change between 50 and 99 percent. The method was later modified by Nowacki and Abrams (1997) by using 10-year averages and expanding the definition of a moderate release to include growth percentage increases of ≥25 percent. Another method for detecting release is the absolute increase method developed by Fraver and White (2005). This method also uses 10-year averages but includes a species-specific threshold value rather than a percent. They recommend thresholds for several species, but for other species they suggested using the formula of 1.25 times the standard deviation of all absolute increases to calculate the threshold.

For this study, we modified the above methods to use an average of only 5 years prior to and following release because of the young age of these trees. To attempt to detect a release using the absolute increase method, we calculated the threshold value of 1.8 mm using the standard deviation method, because shortleaf pine was not a species included in Fraver and White (2005).

Data Analysis

Because this study compares two adjacent management units and does not have true replication at the stand level, the first research question regarding stand composition and repeated burning was addressed with descriptive statistics only. Composition for each unit was reported by grouping white oak species [white oak (*Quercus alba*) and post oak (*Q. stellata*)] and red oak species [scarlet oak (*Q. coccinea*), black (*Q. velutina*), blackjack (*Q. marilandica*), and northern red oak (*Q. rubra*)]. Hickories (*Carya* spp.) were also grouped, but not identified to the species level. Sumac species were predominately fragrant sumac (*Rhus aromatica*) with a smaller component of winged sumac (*R. copallinum*).

To address questions 2 and 3, regarding growth response to release and prescribed burning, a repeated measures analysis of variance (ANOVA) was used with ring width or BAI as the response variable, year as the within-subject variable, and treatment (burned versus nonburned) as the between-subject variable. Because repeated measures ANOVA requires listwise deletion of missing values (meaning any sample missing ring-width measurements for any year is deleted entirely from the dataset), analysis included only years with full replication, which began in 2011. The focal tree is the experimental unit for these analyses. To determine growth response to competition, a generalized linear model with a lognormal distribution and a logit link was used with the average ring width or BAI of 2019–2021 as the response variable and plot-level basal area, stocking, or competition index as fixed effects. Because the results of the repeated measures ANOVA showed separation starting in 2019, the years 2019–2021 were used to analyze competition.

RESULTS

Stand Level

In the nonburned area, the average overstory basal area was 15 m^2/ha and the average midstory basal area was 6 m^2/ha for a total of 21 m^2/ha. The mean number of understory stems (≥1.37 m tall) was 2,060 stems/ha. In the burned area the average overstory basal area was 12 m^2/ha and the average midstory basal area was 1.4 m^2/ha for a total of 13.4 m^2/ha. The average number of understory stems was 3,291 stems/ha. Species composition was similar in the overstory but differed in the midstory and understory (tables 1 and 2). The average d.b.h. of cored trees in the burned unit was 9.8 cm and the average d.b.h. of cored trees in the nonburned unit was 9.1 cm. All cored trees were considered part of the 2005 planting.

Table 1—Overstory and midstory species composition by percentage of basal area in the burned and nonburned management units

	Overstory		Midstory	
Species	Burned	Nonburned	Burned	Nonburned
Shortleaf pine	73	74	59	–
White oak species	12	12	12	19
Hickory species	11	4	6	17
Red oak species	4	8	10	21
Winged elm	–	–	–	13
Other	0	2	13	30

Table 2—Understory species composition by percentage of stems in the burned and nonburned management units

Species	Burned	Nonburned
Sumac species	48	–
Black cherry	21	4
Red oak species	14	8
Black gum	5	–
Hickory species	4	28
Winged elm	–	40
White oak species	–	8
Other	8	12

Annual Ring Width

Analysis of raw ring-width data revealed a significant effect of year (P <0.001) and a significant interaction between year and treatment (P <0.001) on ring width. Years in which treatments were significantly different included 2013 (P <0.001) with the nonburned greater than burned, and 2019 (P = 0.002), 2020 (P <0.001), and 2021 (P <0.001), with the burned greater than nonburned (fig. 1). Year differences within units were complex. In the burned unit, generally years after 2014 had greater ring widths than years before, with the exception of 2018 being similar to early years. The nonburned areas had fewer significant differences, with 2016 being significantly higher than some early years.

Analysis of BAI also revealed significant effects of year (P <0.001) and a significant interaction between year and treatment (P <0.001). Years in

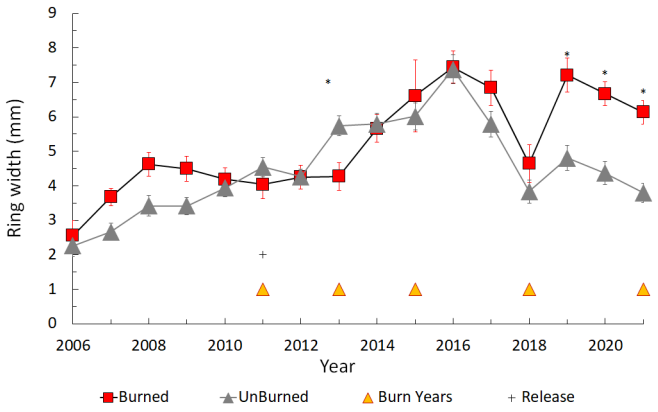

Figure 1—Average ring width for shortleaf pine tree cores taken from the burned and nonburned management units by year. *indicates years with significant treatment differences. Error bars are standard errors.

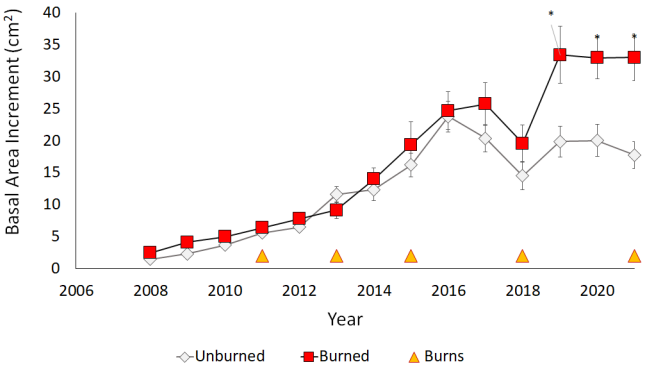

Figure 2—Average basal area increment for shortleaf pine tree cores taken from the burned and nonburned management units by year. *indicates years with significant treatment differences. Error bars are standard errors. A release occurred in 2011.

which BAI was greater in the burned unit than in the nonburned unit were 2019 (P = 0.012), 2020 (P = 0.003), and 2021 (P <0.001) (fig. 2). In both the burned and nonburned units, BAIs in later years were generally higher than in earlier years.

Effectiveness of Release

Using averages for the years 2006–2010 and 2012–2016 (omitting the release year 2011 in the averaging), the percentage of growth change for the nonburned unit was 69 percent, the burned unit was 38 percent, and combined was 52 percent, so there is evidence that the 2011 release was effective according to the 50 percent and 25 percent thresholds established by Lorimer (1985) and Nowaki and Abrams (1997). The differences between the 5-year post release average and the prerelease average were 2.38 mm in the nonburned unit, 1.65 mm in the burned unit, and 2.02 mm when data were combined, indicating a detectable release from the treatment in 2011 in the nonburned unit and when looking at the units together, according to the absolute increase method (Fraver and White 2005).

Both units showed a decline in growth from 2016 to 2018, then in 2019 the burned area had significantly greater ring widths for the remainder of the measurements. It is possible that the 2018 burn acted as a release from midstory competition in the burned unit, but with only 3 years following the 2018 burn, this was not calculated.

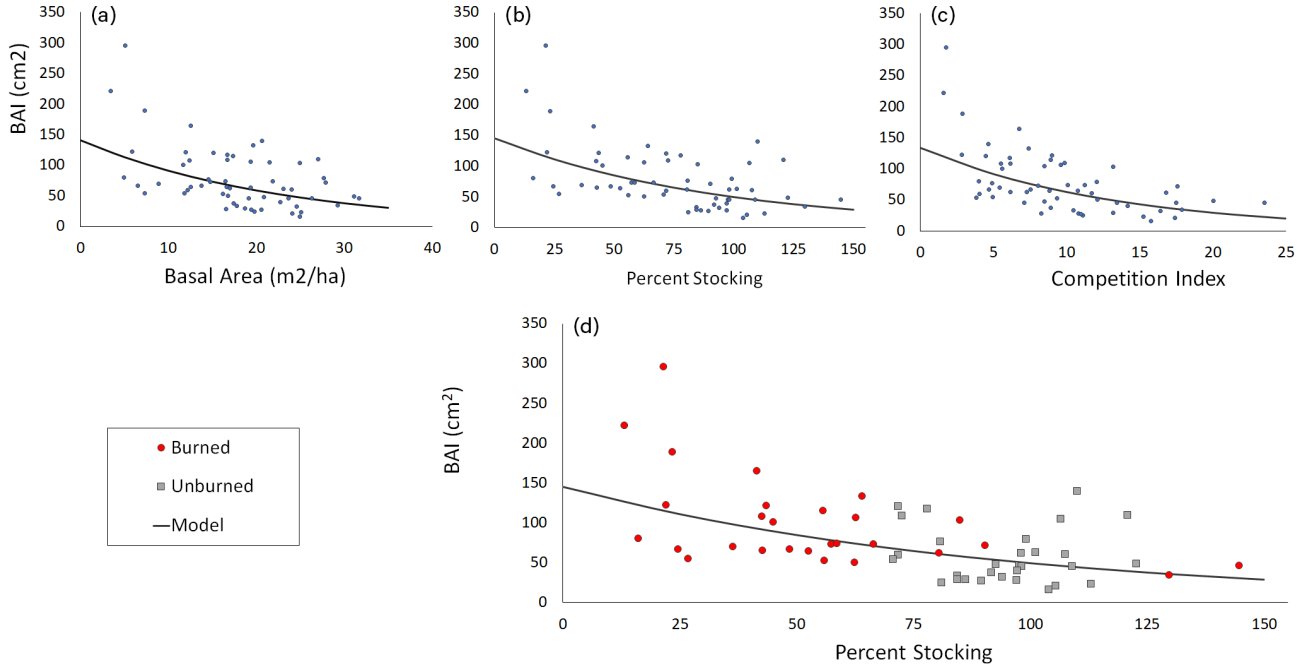

Figure 3—Cumulative basal area increment (BAI) from 2019 to 2021 across both burned and nonburned units by (a) basal area, (b) stocking, and (c) competition index. (d) BAI by stocking is shown again magnified with data points separated by unit.

Effects of Competition

The cumulative BAI from 2019 to 2021 was negatively correlated with basal area (P <0.001), stocking (P <0.001), and competition index (P <0.001) (fig. 3). The raw ring-width data averaged across the years 2019 to 2021 was also negatively correlated with basal area (P <0.001), stocking (P <0.001), and competition index (P <0.001).

Cumulative BAI for 2019–2021 was significantly correlated with basal area (P = 0.002) and the interaction between basal area and percentage of hardwood basal area (P <0.001) (fig. 4). This pattern was apparent for ring-width data averaged from 2019–2021 as well, with significant effects of basal area (P = 0.048) and the interaction between basal area and percent hardwoods (P = 0.018). There was not a significant interaction between stocking and percentage of hardwoods for either BAI (P = 0.210) or ring width (P = 0.068).

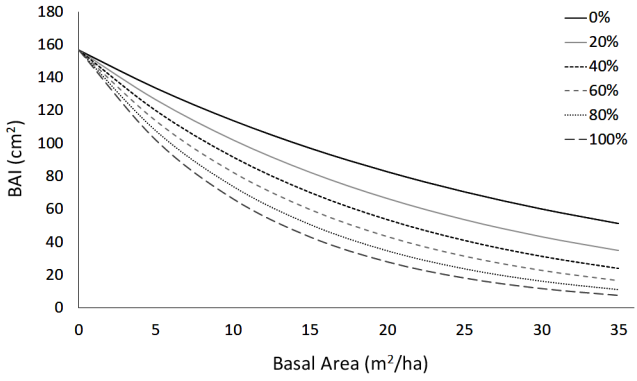

Figure 4—Cumulative basal area increment (BAI) from 2019 to 2021 across both burned and nonburned units by basal area and percentage of basal area composed of hardwoods.

DISCUSSION

This study had several findings about the relationships between prescribed burning and hardwood competition, and how both factors influence the growth of shortleaf pine. We found that the prescribed burning regime in this study altered midstory and understory composition and that the release treatment done in this study was moderately effective according to two different methods. We also found that competition reduced the radial growth of shortleaf pine while prescribed burning, in most cases, did not.

Our analysis of midstory and understory composition pointed to differences in the burned and nonburned units, with the burned unit having more shortleaf pine and fewer hardwoods in the midstory. Similar compositional changes have been seen in other studies (Cain and others 1998, Elliott and Vose 2005, Flatley and others 2023, Robertson and others 2021). In the Mark Twain tornado area, our results suggest that a 2- to 3-year fire return interval is suited for

maintaining shortleaf dominance, but continued monitoring is needed to determine if this trend continues as the stand ages.

The release treatment took place in 2012, which was a hot, droughty year in Missouri. Carter County had the highest maximum growing season temperature for the period of 2000–2021 in 2012 (29.8 °C) as well as the lowest precipitation (401 mm) (NOAA 2022). This drought likely reduced shortleaf growth, so a potentially greater response to the 2012 release may have been possible were the release not immediately followed by drought. There is evidence that hot, dry conditions slow the growth of shortleaf pine. Grissino-Mayer and Butler (1993) found significant negative relationships between growth and temperature in April, June, July, August, and September. They also found significant positive relationships between growth and precipitation during May, June, and July. These same trends were observed when using seasonal averages of temperature and precipitation, and the strongest predictor of growth was growing season precipitation. In Oklahoma, on the western edge of shortleaf pine's range, Adhikari and others (2021) found that annual growth was positively correlated with growing season precipitation and negatively correlated with maximum summer temperature. They also found evidence that higher minimum temperatures the previous October correlate with higher growth and that trees growing in open savanna conditions were less sensitive to annual variation in precipitation than trees growing in closed forests. Using long-term data from the Missouri Ozark Forest Ecosystem Project, Stambaugh and Guyette (2004) found that shortleaf pine growth was sensitive to a 21-year drought cycle, and that growth was correlated with the minimum winter temperature prior to the growing season.

The effects of competition, especially from hardwoods, on shortleaf pine growth seen in this study are consistent with literature on pine and hardwood competition. In southeastern Arkansas, shortleaf pine and loblolly pine growth was examined in naturally regenerated stands with different types of competition control. From ages 5 to 13, pine diameter growth increased on plots with woody competition control compared with plots with only herbaceous competition control because of hardwood competition in the herbaceous control areas (Cain 1997). In the Pisgah National Forest, McNab (2022) found that shortleaf and pitch pines (*P. rigida*) planted in a clearcut hardwood stand were significantly taller when growing in groups of at least two pines than when surrounded by hardwood species. They concluded that the pine groups provided a buffer from interspecific competition, which was more detrimental than the intraspecific competition of pine groups (McNab 2022). The increased effects of hardwoods to growth reduction in this study were not seen when analyzing growth by stocking, as the stocking equations themselves account for the greater growing space required by hardwoods (Gingrich 1967, Rogers 1983). When predicting radial growth of shortleaf pine in response to competition, managers should use stocking measurements or account for the differential effects of hardwoods if using basal area measurements.

In this study, 2013 was the only year in which prescribed burning resulted in significantly lower growth in the burned unit, and only when analyzing raw ring-width data. This is somewhat consistent with the study by Adhikari and others (2021) in the western Ouachita Mountains, in which tree rings were analyzed in cores collected from shortleaf pine trees in oak-pine forest burned periodically since 1985. In areas burned on 2- or 3-year intervals, the radial growth in the year following fire was reduced by 21 to 33 percent. A reduction in growth following prescribed burning was not seen in the 4-year return interval stands, which the authors attributed to differences in fuel characteristics (Adhikari and others 2021). Both this study and Adhikari and others (2021) seem to indicate that fire behavior affects possible growth reductions in shortleaf. The 2013 burn was anecdotally described as being hot, and took place in April, a bit later than the typical dormant season burns in this unit. Further research on prescribed burning and radial growth reduction of shortleaf pine, including data on fire timing and behavior, would likely be useful to determine how and when prescribed burning could lead to growth reductions.

Overall, this study shows that repeated burning may change midstory composition by reducing fire-sensitive species and promoting shortleaf pine. One of the burn years in this study resulted in lower ring widths in the burned unit, while the others did not, indicating that factors such as burn conditions and fire behavior may mediate the effects of burning on shortleaf pine growth, but that burning does not necessarily reduce BAI. Local competition density reduced shortleaf pine radial growth in this study, and this was especially true for hardwood competition. Additionally, the hardwood midstory may have contributed to lower growth in the nonburned area in the later years of this study.

LITERATURE CITED

Anderson, M.; Hayes, L.; Keyser, P.D. [and others]. 2016. Shortleaf pine restoration plan: restoring an American forest legacy. Shortleaf Pine Initiative. 57 p. https://shortleafpine.org/shortleaf-pine-initiative/shortleaf-pine-restoration-plan/shortleaf-pine-restoration-plan [Date accessed: February 1, 2022].

Adhikari, A.; Masters, R.E.; Adams, H. [and others]. 2021. Effects of climate variability and management on shortleaf pine radial growth across a forest-savanna continuum in a 34-year experiment. Forest Ecology and Management. 491: 119125. 12 p. https://doi.org/10.1016/j.foreco.2021.119125.

Bower, D.R.; Ferguson, E.R. 1968. Understory removal improves shortleaf pine growth. Journal of Forestry. 66(5): 421-422. https://doi.org/10.1093/jof/66.5.421.

Brinkman, K.A.; Smith, R.C. 1968. Managing shortleaf pine in Missouri. Station Bull. 875. Columbia, MO: Agricultural Experiment Station, University of Missouri-Columbia. 35 p.

Cain, M.D. 1997. Woody and herbaceous competition effects on the growth of naturally regenerated loblolly and shortleaf pines through 11 years. New Forests. 14: 107–125. https://doi.org/10.1023/A:1006512721318.

Cain, M.D.; Wigley, T.B.; Reed, D.J. 1998. Prescribed fire effects on structure in uneven-aged stands of loblolly and shortleaf pines. Wildlife Society Bulletin. 26: 209–218.

Elliott, K.J.; Vose, J.M. 2005. Effects of understory prescribed burning on shortleaf pine (*Pinus echinata* Mill.)/mixed-hardwood forests. Journal of the Torrey Botanical Society. 132(2): 236–251. https://doi.org/10.3159/1095-5674(2005)132[236:EOUPBO]2.0.CO;2.

Flatley, W.T.; Bragg, L.M.; Bragg, D.C. 2023. Dynamic fire regimes and forest conditions across three centuries in a shortleaf pine-oak forest in the Ouachita Mountains, Arkansas, USA. Annals of the American Association of Geographers. 113: 1365–1382. https://doi.org/10.1080/24694452.2023.2189937.

Fraver, S.; White, A.S. 2005. Identifying growth releases in dendrochronological studies of forest disturbance. Canadian Journal of Forest Research. 35(7): 1648–1656. https://doi.org/10.1139/x05-092.

Gingrich, S.F. 1967. Measuring and evaluating stocking and stand density in upland hardwood forests in the Central States. Forest Science. 13: 38–53. https://doi.org/10.1093/forestscience/13.1.38.

Grissino-Mayer, H.D.; Butler, D.R. 1993. Effects of climate on growth of shortleaf pine (*Pinus echinata* Mill.) in northern Georgia: a dendroclimatic study. Southeastern Geographer. 33(1): 65–81. https://doi.org/10.1353/sgo.1993.0003.

Kabrick, J.M.; Knapp, B.O.; Dey, D.C; Larsen, D.R. 2015. Effect of initial seedling size, understory competition, and overstory density on the survival and growth of *Pinus echinata* seedlings underplanted in hardwood forests for restoration. New Forests. 46(5): 897–918. https://doi.org/10.1007/s11056-015-9487-3.

Lorimer, C.G. 1983. Tests of age-independent competition indices for individual trees in natural hardwood stands. Forest Ecology and Management. 6(4): 343–360. https://doi.org/10.1016/0378-1127(83)90042-7.

Lorimer, C.G. 1985. Methodological considerations in the analysis of forest disturbance history. Canadian Journal of Forest Research. 15(1): 200–213. https://doi.org/10.1139/x85-038.

McNab, W.H. 2022. Height of pitch and shortleaf pine is increased by intraspecific competition 13 years after planting in the Southern Appalachians. In: Willis, J.; Self, A.B.; Siegert, C.M., eds. Proceedings of the 21st biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-268. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 181–188. https://doi.org/10.2737/SRS-GTR-268.

National Oceanic and Atmospheric Administration [NOAA]. 2022. Climate at a glance: County time series. National Centers for Environmental Information, National Oceanic and Atmospheric Administration. https://www.ncei.noaa.gov/access/monitoring/climate-at-a-glance/county/time-series. [Date accessed: October 24, 2023].

Nowacki, G.J.; Abrams, M.D. 1997. Radial-growth averaging criteria for reconstructing disturbance histories from presettlement-origin oaks. Ecological Monographs. 67(2): 225–249. https://doi.org/10.2307/2963514.

Oswalt, C.M. 2012. Spatial and temporal trends of the shortleaf pine resource in the eastern United States. In: Kush, J.; Barlow, R.J.; Gilbert, J.C., eds. Proceedings of the shortleaf pine conference: east meets west. Special Rep. 11. Auburn, AL: Alabama Agricultural Experiment Station: 33–37.

Robertson, K.M.; Hermann, S.M.; Staller, E.L. 2021. Frequent prescribed fire sustains old field loblolly pine-shortleaf pine woodland communities: results of a 53-year study. Journal of Forestry. 119(6): 549–556. https://doi.org/10.1093/jofore/fvab035.

Rogers, N.F.; Brinkman, K.A. 1965. Shortleaf pine in Missouri: Understory hardwoods retard growth. Res. Paper CS-15. Columbus, OH: U.S. Department of Agriculture Forest Service, Central States Forest Experimental Station, Columbus, Ohio. 9 p.

Rogers, R. 1983. Guides for thinning shortleaf pine. In: Jones, E.P., Jr., ed. Proceedings of the second biennial southern silvicultural research conference. Gen. Tech. Rep. SE-24. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 217–225.

Stambaugh, M.C.; Guyette, R.P. 2004. Long-term growth and climate response of shortleaf pine at the Missouri Ozark Forest Ecosystem Project. In: Yaussy, D.A.; Hix, D.M.; Long, R.P.; Goebel, P.C., eds. Proceedings of the 14th central hardwood forest conference. Gen. Tech. Rep. NE-316. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station: 448–458.

Trotsiuk, V.; Pederson, N.; Druckenbrod, D.L. [and others]. 2018. Testing the efficacy of tree-ring methods for detecting past disturbances. Forest Ecology and Management. 425: 59–67.

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2017. Web soil survey. https://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm. [Date accessed: November 1, 2017].

Restoration and Reclamation

SOME SILVICULTURAL IMPLICATIONS OF UPDATING THE SOUTHERN REGION'S OLD-GROWTH GUIDANCE

Don C. Bragg

ABSTRACT

The U.S. Department of Agriculture, Forest Service's Southern Region, in conjunction with the Southern Research Station (SRS), has chartered a formal team of managers, planners, and scientists to update the Southern Region's 1997 guidance document on old-growth management and restoration. There are several silvicultural implications associated with old growth for the Region's national forests. For example, regional-scale restoration initiatives have been established for several important but declining species, including longleaf pine (*Pinus palustris*), shortleaf pine (*P. echinata*), and white oak (*Quercus alba*). Open pine- and oak-dominated old-growth forests were once very common across the region, and their restoration focuses on thinning overstories, greatly reducing midstories, and the application of frequent prescribed fire. These treatments are usually designed to reduce arboreal richness, encourage certain shade-intolerant overstory species, and reduce stand densities to enhance the cover and diversity of grass- and forb-dominated understories. However, these treatments also diminish certain stand conditions (e.g., high stocking and biomass, arboreally rich stands) often mistakenly assigned to all old-growth forests. Other questions related to silvicultural practices in old stands, from determining the degree of maturity for candidate stands to allowable treatments and the long-term sustainability of old growth, will be addressed by this update. Ultimately, the team's goal is to produce updated old-growth guidance by fiscal year 2024, thereby providing Southern Region managers and planners better, more scientifically robust information.

INTRODUCTION

As the 20th century progressed, the silvicultural practices of Federal agencies faced increased scrutiny. Socioeconomic issues, cultural differences, and environmental concerns led to increasingly contentious planning and management activities by these agencies. This was particularly true for the U.S. Department of Agriculture, Forest Service lands in the Western United States. In these national forests, many of which were still widely logging old-growth forests, concerns were raised about endangered species associated with older forests, such as the northern spotted owl (*Strix occidentalis caurina*) and marbled murrelet (*Brachyramphus marmoratus*). By the 1980s, disagreements had become outright controversies and even conflict, and the Federal Government sought a new approach to the management of their old forest resources that were not already wilderness (or otherwise protected).

In October of 1989, Forest Service Chief F. Dale Robertson issued a formal position statement to provide a "generic" definition and description of old growth and directed national forests to develop region-specific old-growth definitions for their major forest cover types (Robertson 1990). This statement was largely in response to the immediate controversy at that time (the logging of mesic old-growth in the Pacific Northwest) and so perhaps unsurprisingly the generic description of old-growth attributes best matched forests from that area. For instance, large trees were (and are still by many) considered a fundamental component of old growth; the same is also often said for high amounts of dead wood, high arboreal diversity, multiple tree size and age structures, and gap-phase stand dynamics (fig. 1).

Don C. Bragg (deceased), Project Leader and Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Monticello, AR 71656

Figure 1—Much of how old growth is considered today comes from peoples' experiences with the enormous western conifers, such as these coastal redwoods (*Sequoia sempervirens*) in northern California. (USDA Forest Service photo by Don C. Bragg, August 2014)

Such considerations also affected the choices of possible silvicultural treatments.

Robertson's directive prompted the Southern Region, the Eastern Region and the Southern, Southeastern, North Central, and Northeastern Forest Experiment Stations to produce a guidance document (Gaines and others 1997) and supporting science materials (for example, Batista and Platt 1997, Murphy and Nowacki 1997) related to the management and restoration of old growth. Although the Gaines and others (1997) guidance represented the best available science and sought to achieve a balance between requirements to protect a vastly diminished resource and the need to manage and restore southern forests, it represented a snapshot in time and contained a number of fundamental limitations and flaws that quickly proved challenging. For example, the available "characteristic" examples of old growth from the Southern Region often mentioned in the supporting science documents were based on the few examples of contemporary old growth that remained, most of which had been protected from harvesting and fire decades earlier. This means that in the case of the xeric pine and pine-oak forests (Murphy and Nowacki 1997), stands that were once nearly pure and often open shortleaf (*Pinus echinata*), pitch (*P. rigida*), or Table Mountain (*P. pungens*) pines historically were now increasingly dominated by fire-intolerant hardwoods and conifers, with much higher stocking. Furthermore, the supporting science document for "wet" pine-dominated forests [which included longleaf (*P. palustris*), slash (*P. elliottii*), and pond (*P. serotina*) pines] was bereft of modern examples because of how few had been described (Harms 1996).

The conventional wisdom of old growth being well-stocked, diverse stands of big (and, hence, presumably old) trees also did not match particularly well with some of the best surviving and most intact remaining examples of old growth. For instance, the open and stunted post (*Quercus stellata*) and blackjack (*Q. marilandica*) oak-dominated forests of the Cross Timbers in parts of Oklahoma, Texas, and Arkansas often had trees more than 200 years old that were less than 50 feet tall and less than a foot in diameter (fig. 2). These hard-to-access, gnarled, and hollow specimens of oak were of very limited value for conventional wood products. Hence, these stands were often left as not worth the effort to cut, even if only to replace with more productive species, because land managers recognized the poor quality of these sites (Bragg and others 2012).

Over the years, the Forest Service also updated some of its more encompassing regulations, including the 2012 Planning Rule (USDA Forest Service 2012), that affected old-growth management on national forest lands while leaving the specifics on how old growth would be addressed to individual regions and even national forests. The remainder of this paper describes efforts over the past couple years to update the Southern Region old-growth guidance and the science that supports it, including that related to silvicultural practices.

Figure 2—An example of a 150- to 200-year-old xeric oak stand in the Cross Timbers of extreme western Arkansas with stunted and gnarly, uncommercial specimens. (USDA Forest Service photo by Don C. Bragg, January 2009)

DEVELOPING AN UPDATED GUIDANCE DOCUMENT

Organization and Directions for Teams

Since 1997, individual projects and even forest plans in the Southern Region have received increased scrutiny over their handling of old growth. Depending on local issues and constituencies, Forest Service units in the Southern Region developed forest plans and environmental impact statements with a range of detail related to addressing old growth, from very superficial (USDA Forest Service 2004b) to lengthy and intricate (USDA Forest Service 2004a), yet all were based upon the framework provided by Gaines and others (1997). After nearly a quarter-century of challenges and ambiguities plus scientific advances related to old forests, the Southern Region leadership decided to re-evaluate its old-growth guidance. With multiple Southern Region national forests starting the process of developing new forest plans (especially following the protracted plan development process experienced by the Nantahala and Pisgah National Forests, NC), addressing old growth and how it may be managed and restored would prove timely.

In 2020, a Southern Region leadership training project on the 1997 old-growth guidance produced multiple recommendations, including the formation of a team from both the Southern Region and the Southern Research Station to explore options. By April 2021, this informal team was assembled and virtually met weekly over the course of the next year. Discussions included what had been done in the quarter-century prior, the current (1997) guidance, and the interests of Southern Region leadership, as well as hearing from some Southern Region national forests that had experienced recent—even ongoing—challenges in managing old growth. By the spring of 2022, and following additional directions from Southern Region leadership, the informal team developed a set of alternatives for leadership to consider. These alternatives ranged from doing nothing (keeping 1997 old-growth guidance in place) to modest changes to that guidance to the development of completely new old-growth guidance. The decision was made to develop thoroughly updated old-growth guidance, and by May 2022 a team was formally chartered (largely composed of the same informal Southern Region and Southern Research Station team members) and directed to spend the next 2 years completing the following tasks:

- Update the Southern Region's 1997 guidance document on old-growth management and restoration
- Modernize the supporting science for this updated old-growth guidance document
- Create a database of relevant scientific, technical, and other supporting literature and resources
- Based on the aforementioned database, publish an annotated bibliography and a review/synthesis on what is known about old growth in the Southeastern United States
- Produce supporting materials for the Southern Region's old-growth planning, management, and restoration needs

Most of these tasks are related, at least partly, to forest management and related silvicultural practices.

Updating Old-Growth Science for the South

The science subteam focused most of their efforts during the first year on building a reference database that included scientific papers, technical reports, newspaper articles, historical photographs, maps, and other useful items relevant to old growth in the Southeastern United States. As a part of this literature and resource review, the science subteam made several decisions about what kinds of content to include in the database. For instance, the subteam elected to include descriptions of old-growth stands from States outside of the 13 that compose the Southern Region. Missouri—technically part of the Eastern Region—has good examples of shortleaf pine, oak-hickory (*Carya* spp.), and bottomland hardwood old growth that are very similar to those found in

the Southern Region. Other resources make only a passing mention of old growth in southern forests that contributes little to nothing of material value, and the subteam elected not to include these in the database. Additional sources will take more nuanced interpretation to be applicable to this effort, such as enhancing the conceptual or theoretical understanding of old growth, and hence may appear in a review/synthesis paper supporting the updated old-growth guidance but will not be part of the resource database or annotated bibliography.

Updating Other Old-Growth Guidance Components

While the updated science underpinning the Southern Region's old-growth guidance will support all aspects of this effort, other elements of this work could likewise impact the practice of silviculture in the Region. After all, a quarter-century of advances in knowledge, new management practices, and frequent policy changes will require many changes. As an example, Gaines and others (1997) applied the same forest type classification that most other contemporary efforts used (Tyrrell and others 1998) when developing their guidance—the Society of American Forester's forest cover types (Eyre 1980). However, starting in the early 2000s, Federal agencies started complying with a mandate to make their forest classification systems to comply with national standards established by the Federal Geographic Data Committee (FGDC) and published in 2008 as the U.S. National Vegetation Classification (USNVC) (Nelson and others 2015; see also https://usnvc.org/). Hence, the forest type classifications of the national forests of the Southern Region must now be crosswalked from those used in Gaines and others (1997) to those developed in compliance with the USNVC.

Additional updates (still in progress at the time of writing this article) will also be made on guidance related to aspects such as patch size, inventory and monitoring, field identification of candidate stands, management options, and planning documentation.

RESULTS AND DISCUSSION

Progress to Date

Over the past 2 years, a framework for an updated guidance document was developed, first by the informal team and then by the chartered team. During this process, potential consequences to silvicultural options for managers and planners have been considered. For example, shifts in Forest Service land management priorities from primarily timber production to predominantly ecosystem restoration over the past 25 years have dramatically altered silvicultural systems. However, active management of Forest Service lands outside of reserves such as wilderness areas is still expected. Though old growth in the Southern Region is found in both protected and unprotected public lands, the lessons learned could also be adapted to other lands not considered to be old forest and therefore treatments adapted to encourage old-growth attributes conducive to desired goods and services. Hence, old-growth science has evolved from developing a better understanding of historical forest conditions to include how certain stand structures and compositions are compatible with desired old-growth conditions and what silvicultural tools may be best suited to achieve managers' goals. In other words, it may not be desirable to simply set aside stands as examples of old growth if those reserves are not functioning as such, especially if treatments such as overstory reduction and the re-establishment of frequent prescribed fire are more effective silvicultural options (fig. 3).

To date, more than 750 references on old growth have been incorporated into the literature database (with more anticipated); some of these references date back to before 1900, but the vast majority are from the mid- to late 20th and early 21st centuries. Most of these sources have been qualitatively descriptive, providing examples with some insights into aspects such as stand structure, composition, function, and dynamics. A number are also relatively novel for such reference gathering exercises—the addition of photographs to provide examples and supplemental information should prove

Figure 3—Silvicultural treatments to the over-, mid-, and understory of national forest stands will likely be required in many mature or old forests to improve their contributions to ecosystem goods and services. In this example, (left) an old-growth longleaf pine (*Pinus palustris*) stand in southern Georgia from 1903 shows how open these fire-prone ecosystems were; in addition to limiting stand densities, frequent fires (probably annual in this case) also eliminated most other tree species and consumed most of the dead wood that formed. Restoration of (right) the mature second-growth longleaf stand from eastern Texas will require removal of most undesired overstory hardwoods and pines of other species, fire (and perhaps herbicides) to control the woody midstory vegetation, and the re-establishment of a diverse ground flora. (USDA Forest Service photos (left) by E. Block; (right) modern Texas longleaf pine stand by Don C. Bragg)

highly informative (fig. 4). After all, old-growth identification efforts that look for stands with high levels of arboreal richness, large trees, dense stocking, and high levels of dead wood will miss old woodlands and savannas without these characteristics. A few sources are directly related to silvicultural practices designed to restore or accentuate old-growth attributes, such as those related to managing stands for open conditions (for example, thinning overstocked stands, girdling undesired or nonnative trees to remove them, or the return of frequent prescribed fire to manipulate vegetation patterns).

This science review and update will also inform silviculture and other management options for old growth (known, likely, and planned), as work continues on the science, techniques, and technologies related to this stand condition. As an example, efforts to restore the ecosystem services of mature, open forests in the Southern Region have developed considerably since the 1990s to support the needs of declining species such as the red-cockaded woodpecker (*Leuconotopicus borealis*). Today, extensive areas receive restoration treatments that include over- and mid-story thinnings, increased use of prescribed fire, lengthened rotations, invasive species control, and other treatments designed to accentuate elements (such as a rich understory of native herbaceous species) deemed critical to healthy ecosystems. This restoration work has been supported and sustained by significant amounts of new and expanded research, some of which is specific to old-growth forests in the Southeastern United States (Bragg 2012, Gilliam and Platt 1999, Spetich and others 2022, Therrell and Stahle 1998, Varner and Kush 2004).

Eventually, the reference database on Southern Region old growth being developed by the Southern Research Station will be turned over to the Region for maintenance and content will be added as discovered. Supporting publications will also be produced from the database, including an annotated bibliography on regional old-growth

Figure 4—Further examples of important considerations regarding old growth in the Southeastern United States—not all "atypical" old forests in this region are the result of xeric sites or frequent fires. In these cases, (top) a historical photograph of an old pond pine (*Pinus serotina*) growing in an open stand in a pocosin on the Croatan National Forest near the coastline of North Carolina, and (bottom) a modern-day old-growth tupelo gum (*Nyssa aquatica*) and baldcypress (*Taxodium distichum*) swamp in eastern Arkansas. (USDA Forest Service photos (top) by Daniel O. Todd, July 1952; (bottom) modern swamp by Don C. Bragg)

resources and a peer-reviewed science synthesis/review research paper. Fundamental to these products, as well as the updated guidance document itself, will be how silvicultural options may be affected in the Southern Region.

Work Elsewhere…and How it May Affect This Guidance Update and Regional Silviculture

In addition to a growing body of work in the remaining old-growth forests of the Southern United States, new technologies have spurred efforts to assess forest age/structural conditions at a regional level, some of which readily translate themselves to finer-scale (even stand-level) mapping. One of the first such "big data" products to estimate age structure for North America was developed by Pan and others (2011), who combined forest inventory data, satellite data, and several types of disturbance datasets to predict forest age distributions for Canada and the United States. Pan and others (2011: figs. 1a and 4) estimated that about 80 percent of the forests in the Southeastern United States were <60 years old, only a few stands were >80 years old, and very few (<1 percent) >110 years old. Their estimate corresponds reasonably well with another estimate for old forests in the South from the 2020 Resource Planning Act (RPA) Assessment (Oswalt and others 2019). Using the Forest Service Forest Inventory and Analysis (FIA) data, Oswalt and others (2019: table 12) estimated that about 3.16 million acres (1.5 percent) of the forests in the Southeastern United States were >100 years old, with most of these older forests being hardwoods. According to their assessment (Oswalt and others 2019), only 0.02 percent of the forests of the South were older than 150 years, a common threshold for old-growth consideration.

The fairly conservative estimates of Pan and others (2011) and Oswalt and others (2019) were dwarfed by those presented in two recent national-level assessments of old growth using big data. DellaSala and others (2022) and Barnett and others (2023) used different approaches to estimate forest maturity in the United States. While a full account of both approaches is beyond the scope of this paper, both DellaSala and others (2022) and Barnett and others (2023) claimed to have found significantly more old growth in the Southeastern United States, especially across the Atlantic and Gulf coastal plains.

Over the past year, other events have occurred that may affect the Southern Region old-growth guidance update and silvicultural practices, especially those related to ecosystem restoration. In April 2022, President Joseph Biden issued Executive Order 14072 entitled "Strengthening the Nation's Forests, Communities, and Local Economies." Section 2b of this Executive Order directed the Secretaries of Agriculture and Interior to have the Forest Service and Bureau of Land Management compile an estimate of how much mature and old-growth forest is found on these federal lands. To accomplish this,

the agencies used FIA, existing information, and available regional old-growth definitions to interpret older forests across their lands nationwide. The resulting report (USDA Forest Service 2023) provided estimates for old-growth area by FIA forest type group on these two federal ownerships.

As presented in USDA Forest Service (2023), it is not possible to decompose their forest type-based old-growth acreage estimates down too finely, as the FIA forest type groups listed for the Southern Region are also found in the Eastern Region. However, as can be seen in figure 5, older forests in the Southern Region tend to be concentrated in the more mountainous areas, such as the Appalachians in the east and the Ouachita and Ozarks in the west. Such elevated and remote forests were less likely to have been logged in the past for various reasons, including accessibility. These rocky, xeric, sometimes nutrient poor mountainous sites also often produced trees of smaller dimensions and poorer form that were ignored by loggers and farmers. Given that most of the national forests in this region were assembled from abused, abandoned, cleared, and cutover lands using the legislative authorities granted after the 1911 Weeks Act—and especially in the late 1920s and 1930s after additional laws strengthened the acquisition capacity of the Forest Service (Bramwell 2012)—the low levels of old growth are not surprising.

Finally, "mature" forests—defined in USDA Forest Service (2023) as the stage of forest development immediately prior to old growth—are more widely distributed (fig. 5) and found in greater quantity on most national forests in the Southern Region. Silvicultural practices including ecosystem restoration will likely remain a priority on these mature federal forests, so long as the language conflating mature and old-growth forests found in Executive Order 14072 and suggested by others (DellaSala and others 2022) does not lead to administrative withdrawals from harvesting. These mature forests offer the best opportunities to use silviculture to accelerate the development of old-growth characteristics and restore needed ecosystem goods and services, but only if the desire to sequester carbon is tempered by the recognition of how open forest ecosystems support other now-rare attributes.

CONCLUSIONS

This effort to develop new Southern Region old-growth guidance still has a long way to go, but steady progress has been made. Silviculturists in Southern Region national forests will need to balance many competing interests and a constantly evolving scientific and social environment to navigate the challenges presented by older forests. For instance, the updated Southern Region guidance on old growth will also need to address the significant ecological differences between mature and old-growth forests to assist forest managers and planners in determining what types of silvicultural systems and treatment options are available. In addition to management options, the updated guidance will provide information on important attributes related to old trees and forests for field staff to look for when entering stands that have little to no recent harvesting. This would include how extensive of an area may need to be considered if old trees are found before a silvicultural prescription is developed. Hopefully, a robust update to the existing Southern Region old-growth guidance will serve these silvicultural purposes, as well as address many other challenges and concerns of forest managers and planners.

ACKNOWLEDGMENTS

I would like to thank the following for their contributions to this project: Margaret Woodbridge, J. Davis Goode, Marty Spetich, Christopher Schalk, Duke Rankin, Joanne Baggs, Cavan Fitzsimmons, Jason Rodrigue, David Taylor, S. Lee Madson, Chelsea Leitz, Allen Nicholas, and the other members of the Region 8 Old-Growth Team.

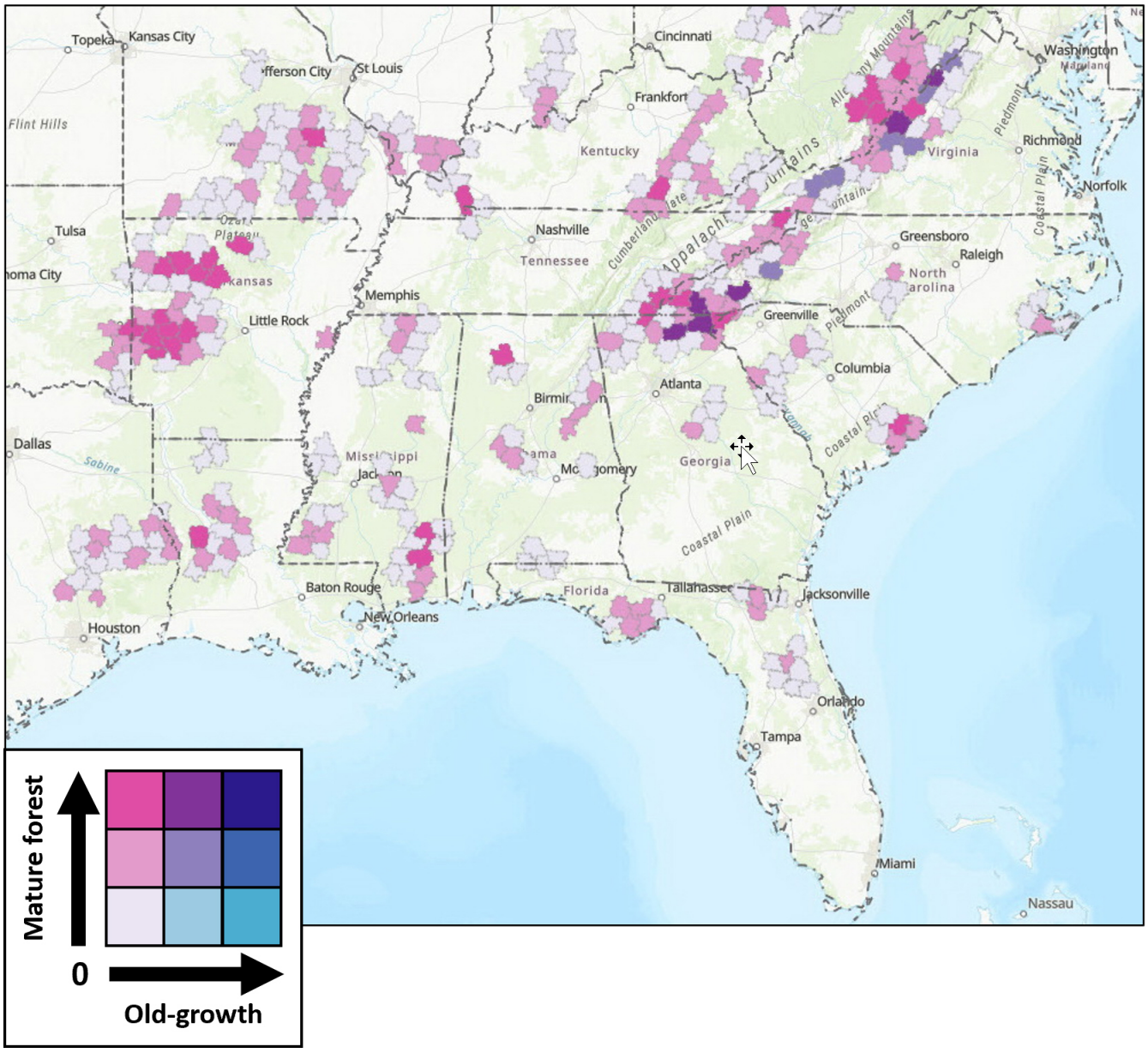

Figure 5—An adaptation of the map providing estimated levels of mature and old-growth forests for the Southern Region, taken from the Forest Service's Climate Risk Viewer (beta 0.1.1) story map (https://storymaps.arcgis.com/collections/87744e6b06c74e82916b9b11da218d28). The polygons used to develop this map are based on approximately 250,000-acre "firesheds" for which FIA data plots on Forest Service lands were assessed for their relative maturity using USDA Forest Service (2023) protocols.

REFERENCES

Barnett, K.; Aplet, G.H.; Belote, R.T. 2023. Classifying, inventorying, and mapping mature and old-growth forests in the United States. Frontiers in Forests and Global Change. 5: 1070372. 25 p. https://doi.org/10.3389/ffgc.2022.1070372.

Batista, W.B.; Platt, W.J. 1997. An old-growth definition for southern mixed hardwood forests. Gen. Tech. Rep. SRS-9. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 11 p. https://doi.org/10.2737/SRS-GTR-9.

Bragg, D.C. 2012. Developing contemporary and historical live tree biomass estimates for old pine-hardwood stands of the Midsouth, USA. Forest Ecology and Management. 281: 32–40. https://doi.org/10.1016/j.foreco.2012.06.015.

Bragg, D.C.; Stahle, D.W.; Cerny, K.C. 2012. Structural attributes of two old-growth Cross Timbers stands in western Arkansas. American Midland Naturalist. 167(1): 40–55. https://doi.org/10.1674/0003-0031-167.1.40.

Bramwell, L. 2012. 1911 Weeks Act: the legislation that nationalized the U.S. Forest Service. Journal of Energy and Natural Resources Law. 30(3): 325–336. https://doi.org/10.1080/02646811.2012.11435298.

DellaSala, D.A.; Mackey, B.; Norman, P. [and others]. 2022. Mature and old-growth forests contribute to large-scale conservation targets in the conterminous United States. Frontiers in Forests and Global Change. 5: 979528. 20 p. https://doi.org/10.3389/ffgc.2022.979528.

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

Gaines, G.; Arndt, P.; Croy, S. [and others]. 1997. Guidance for conserving and restoring old-growth forest communities on national forests in the Southern Region: report of the Region 8 Old-Growth Team. Forestry Report R8-FR 62. Atlanta, GA: U.S. Department of Agriculture, Forest Service, Southern Region. 122 p.

Gilliam, F.S.; Platt, W.J. 1999. Effects of long-term fire exclusion on tree species composition and stand structure in an old-growth *Pinus palustris* (longleaf pine) forest. Plant Ecology. 140: 15–26. https://doi.org/10.1023/A:1009776020438.

Harms, W.R. 1996. An old-growth definition for wet pine forests, woodlands, and savannas. Gen. Tech. Rep. SRS-2. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 7 p. https://doi.org/10.2737/SRS-GTR-2.

Murphy, P.A.; Nowacki, G.J. 1997. An old-growth definition for xeric pine and pine-oak woodlands. Gen. Tech. Rep. SRS-7. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 7 p. https://doi.org/10.2737/SRS-GTR-7.

Nelson, M.L.; Brewer, C.K.; Solem, S.J., tech. eds. and coords. 2015. Existing vegetation classification, mapping, and inventory technical guide, version 2.0. Gen. Tech. Rep. WO-90. Washington, DC: U.S. Department of Agriculture, Forest Service, Ecosystem Management Coordination Staff. 210 p.

Oswalt, S.N.; Smith, W.B.; Miles, P.D.; Pugh, S.A., coords. 2019. Forest resources of the United States, 2017: a technical document supporting the Forest Service 2020 RPA Assessment. Gen. Tech. Rep. WO-97. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 223 p. https://doi.org/10.2737/WO-GTR-97.

Pan, Y.; Chen, J.M.; Birdsey, R. [and others]. 2011. Age structure and disturbance legacy of North American forests. Biogeosciences. 8: 715–732. https://doi.org/10.5194/bg-8-715-2011.

Robertson, F.D. 1990. Position statement of National Forest old-growth values. Senate Hearing 101-553. In: Joint hearing before the Committee on Agriculture, Nutrition, and Forestry and the Committee on Energy and Natural Resources, United States Senate. 101st Congress, First Session on Issues Related to the National Forest Planning Process. Washington, DC: Government Printing Office: 115–118.

Spetich, M.A.; Jenkins, M.A.; Shifley, S.R. [and others]. 2022. Characteristics of dry-mesic old-growth oak forests in the Eastern United States. Earth. 3: 975–1009. https://doi.org/10.3390/earth3030057https://doi.org/10.3390/earth3030057.

Therrell, M.D.; Stahle, D.W. 1998. A predictive model to locate ancient forests in the Cross Timbers of Osage County, Oklahoma. Journal of Biogeography. 25: 847–854. https://doi.org/10.1046/j.1365-2699.1998.00224.x.

Tyrrell, L.E.; Nowacki, G.J.; Crow, T.R. [and others]. 1998. Information about old growth for selected forest type groups in the Eastern United States. Gen. Tech. Rep. NC-197. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 507 p. https://doi.org/10.2737/NC-GTR-197.

U.S. Department of Agriculture, Forest Service [USDA Forest Service]. 2004a. Appendices for the land and resource management plan revision for the Chattahoochee-Oconee National Forests. Manage. Bull. R8-MB 113 E. Atlanta, GA: U.S. Department of Agriculture, Forest Service, Southern Region. 230 p.

U.S. Department of Agriculture, Forest Service [USDA Forest Service]. 2004b. Land and resource management plan, Land Between the Lakes National Recreation Area. Manage. Bull. R8-MB-119A. Atlanta, GA: U.S. Department of Agriculture, Forest Service, Southern Region. 113 p.

U.S. Department of Agriculture, Forest Service [USDA Forest Service]. 2012. National Forest System land management planning. 36 CFR 219. Federal Register. 77(68): 21162–21276.

U.S. Department of Agriculture, Forest Service [USDA Forest Service]. 2023. Mature and old-growth forests: definition, identification, and initial inventory on lands managed by the Forest Service and Bureau of Land Management. FS-1215a. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 63 p.

Varner, J.M., III; Kush, J.S. 2004. Remnant old-growth longleaf pine (*Pinus palustris* Mill.) savannas and forests of the southeastern USA: status and threats. Natural Areas Journal. 24(2): 141–149.

RECONSTRUCTING STAND DYNAMICS OF A SHORTLEAF PINE-OAK WOODLAND IN EAST TEXAS: A CASE STUDY

Benjamin O. Knapp and Steven B. Jack

ABSTRACT

Upland forests of east Texas were historically composed of varying mixtures of pine and hardwood species, with local structure and composition driven by site characteristics and fire regimes. The restoration of natural communities, including upland woodlands with mixtures of shortleaf pine (*Pinus echinata*) and oaks (*Quercus* spp.), provides conservation benefit, but we lack information on optimal silvicultural practices to meet restoration objectives. We examined contemporary and past stand structure and composition and stand dynamics of a shortleaf pine-post oak woodland at Boggy Slough Conservation Area (BSCA) near Lufkin, TX, using dendrochronological stand reconstruction approaches. The stand was dominated by loblolly pine (*P. taeda*) and post oak (*Q. stellata*), as well as shortleaf pine in lower abundance. Post oaks were relatively small and not part of the upper canopy. The age distributions strongly differed among these species, with loblolly pine in the younger age classes (mostly 40- to 50-year-old age classes), post oak in the older age classes (mostly 100- to 110-year-old age classes), and shortleaf pine in between (mostly 80-year-old age class). These results suggest the shortleaf pine-oak woodland is in the process of transitioning to loblolly pine, with management intervention needed to regenerate and recruit shortleaf pine and post oak.

INTRODUCTION

Open forest ecosystems are characterized by open structure, in both horizontal and vertical dimensions, and an abundance of herbaceous vegetation in the understory (Hanberry and others 2018). Tree stocking can vary across a structural gradient, with savanna ecosystems having the lowest stocking and woodlands having greater stocking (Bragg and others 2020, Dey and others 2017). Consistent for all open forest ecosystems, however, is a lack of midstory stems, resulting in greater availability of light reaching the understory. Across most of the central and Eastern United States, frequent surface fires are important for maintaining these natural communities by reducing the development of a midstory and favoring fire-adapted species (Hanberry and others 2018). Evidence suggests that open forest ecosystems were historically more prevalent across the region, with the period of widespread fire exclusion during the 20th century contributing to their loss (Hanberry and others 2020, Nowacki and Abrams 2008). Given their unique conservation benefits, recent interest in restoring and maintaining open forest ecosystems has increased.

In addition to the conservation benefits provided from their open habitat structure, mixtures of softwood-hardwood tree species (termed 'mixedwoods') provide other benefits, such as diversification of forest products, heterogeneity in wildlife habitat, resilience to future uncertainty with climate change or forest health risks, and the potential for greater carbon productivity relative to similar structures of single species (Kabrick and others 2017, Kenefic and others 2021). However, managing for a mixture of tree species is challenging due to the different silvics of the species (Kenefic and others 2021). Mixtures of shortleaf pine (*Pinus echinata*) and oaks (*Quercus* spp.) occur commonly as open forest ecosystems in portions of the central and Southern United States. While often a restoration target, management of shortleaf pine-oak

Benjamin O. Knapp, Associate Professor, School of Natural Resources, University of Missouri, Columbia, MO 65211
Steven B. Jack, Executive Director, Boggy Slough Conservation Area, T.L.L. Temple Foundation, Lufkin, TX 75901

woodlands is challenged by differences in the fire adaptions of the constituent species (Arthur and others 2012, Stambaugh and others 2007) that result in difficulties regenerating the desired species together.

In the Pineywoods region of east Texas, upland sites include mixed composition of pines [shortleaf pine, loblolly pine (*P. taeda*), longleaf pine (*P. palustris*)] and oaks [southern red oak (*Q. falcata*), post oak (*Q. stellata*), blackjack oak (*Q. marilandica*), white oak (*Q. alba*)]. Historically, mixtures of shortleaf pine and post oak were common, with longleaf pine dominant on sandier soils and loblolly pine dominant in low-lying areas and drainages (Bray 1904). Expansion of the timber industry resulted in widespread conversion of upland sites to loblolly pine plantations, and fire exclusion policies of the 20th century facilitated the increase in dominance of loblolly pine. With current interest in restoring shortleaf pine-oak woodlands, managers require information on regeneration patterns and stand dynamics that occur within these forest types.

The Boggy Slough Conservation Area (BSCA), owned by the T.L.L. Temple Foundation and located in Houston and Trinity Counties, TX, is approximately 19,000 acres and includes a mixture of bottomland hardwood and upland forest areas. With a history of ownership connected to the Southern Pine Lumber Company and the family of company founder T.L.L. Temple, BSCA is currently managed under a conservation easement for wildlife, timber, and conservation objectives. While portions of the BSCA had been converted to planted loblolly pine for timber production, approximately 10,600 acres are classified as naturally regenerated upland pine or pine-hardwood stands. In recent years, management objectives have included restoration of the open forest structure and maintaining canopy mixtures of shortleaf pine and oaks. Existing stands with this canopy mixture provide examples of compositional targets for management, but unknown stand histories obscure our understanding of the stand dynamics that resulted in these stands. In this study, we reconstruct stand dynamics by aging canopy trees within a shortleaf pine-oak woodland at BSCA. Our objectives were to describe 1) current structure and composition; 2) the age structure of the dominant pines and oaks; and 3) relationships between tree size and age for the major species (shortleaf pine, loblolly pine, and post oak). Results from this study will inform current management practices to restore and sustain shortleaf pine-oak woodlands.

METHODS

Study Site

At BSCA, we selected an approximately 100-acre, naturally regenerated upland pine-hardwood stand (31°18’20” N, 94°58’00” W), with no record of recent harvest activity. Soils in the study stand were mapped to the Kellison-Keltys-Lovelady map unit, with the Kellison soil series contributing 90 percent of the map unit (USDA NRCS 2022). Kellison loam (fine, smectitic, thermic Vertic Hapludalf) is well drained, with slopes of 5 to 15 percent. Site index (base age 50) of Kellison loam is 65 for shortleaf pine and 75 for loblolly pine (Coile and Schumacher 1953). The stand was mapped as a loamy claypan upland, supporting a shortleaf pine-post oak natural community.

Data Collection

We randomly located ten 0.20-acre, fixed-area circular plots throughout the selected stand. Within each plot, we marked each tree with diameter at breast height (d.b.h., 4.5 feet above the ground) ≥5 inches with a numbered aluminum tag and recorded species and d.b.h. Using an increment borer, we extracted one core at breast height from a subsample of trees on each plot. The subsample of cored trees was designed to systematically sample from the smallest and the largest trees from both oak and pine species groups, with interest in representation of mature canopy trees as well as potentially younger, recruiting trees. To achieve this, we used the tree list from each plot to determine the 3 smallest (in d.b.h.) oak trees and pine trees and the 3 largest (in d.b.h.) oak trees and pine trees, resulting in a total of 12 possible trees cored on each plot. If a plot did not have at least six oaks or pines, the sample size was reduced to the number of trees present. Cores were stored in straws and returned to the laboratory for processing. Each core was

mounted to a wooden rail and sanded with progressively finer grit to allow determination of cellular anatomy.

Data Analysis

Each tree core was scanned with an Epson Expression 11000XL (Epson America, Inc., Los Alamitos, CA) scanner to create a digital image for analyses. Images were imported into CooRecorder/CDendro (Cybis Elektronik & Data AB, Saltsjobaden, Sweden) to determine boundaries of annual rings. If the pith was not reached with the core, the distance to the pith was estimated using default parameters in CooRecorder. The cores were crossdated using CDendro by creating a reference chronology for the stand, iteratively adding samples to the reference chronology as they were measured.

Descriptive statistics of forest structure (stocking, basal area, trees per acre, quadratic mean diameter) were calculated for each species of interest, including post oak, shortleaf pine, loblolly pine, other species, and total. We used Gingrich-style stocking as a measure of growing space occupancy, with stocking for hardwood species calculated based on upland hardwood equations of Gingrich (1967), and stocking for pine species calculated based on shortleaf pine equations of Rogers (1983). Frequency distributions were created for tree size, using 2-inch d.b.h. classes, using all trees sampled in the fixed-area plots and for tree age, using 10-year age classes, for the cored trees only. Because trees were cored at breast height, age represents an age of release (effective age) rather than true chronological age. For each cored tree, scatterplots were created for tree size (d.b.h.) by age for species of interest (post oak, shortleaf pine, and loblolly pine). We fit each scatterplot with a quadratic regression line to characterize the strength and form of the relationships.

RESULTS

The stand was dominated by loblolly pine and post oak, each contributing approximately equally to total stand stocking, followed by shortleaf pine. Total stand stocking was 68 percent, and basal area was 93 square feet per acre. Shortleaf pine had the largest quadratic mean diameter, with the other species group having the lowest (table 1). The small-diameter classes were dominated by loblolly pine and post oak, with very few post oaks occurring in diameter classes >12 inches. While loblolly pine was abundant in small-size classes, it was also the dominant species in large-diameter classes (fig. 1). Shortleaf pine was represented in similar abundance across all size classes.

The age distribution showed separation by species (fig. 2). The loblolly pines were in the younger age classes, with the greatest representation in the 40- and 50-year-old age classes. There was little overlap between loblolly pine and post oak, with the greatest representation of post oak in the 100- and 110-year-old age classes. Shortleaf pine occurred in the central portion of the age distribution, with the greatest representation in the 80-year-old age class. In contrast to loblolly pine, there were no shortleaf pines or post oaks younger than the 50-year-old age class.

Table 1—Stand structure characteristics for a pine-hardwood stand at Boggy Slough Conservation Area by species groups

Species group	Stocking	Basal area	Trees per acre	Quadratic mean diameter
	percent	*square feet per acre*	*n*	*inches*
Post oak	27.5	31.0	62.5	9.5
Other oaks	1.7	2.0	3.0	11.1
Shortleaf pine	7.5	12.3	14.5	12.5
Loblolly pine	26.8	42.6	69.0	10.6
Other species	4.9	5.3	13.5	8.5
Total	68.3	93.2	162.5	10.3

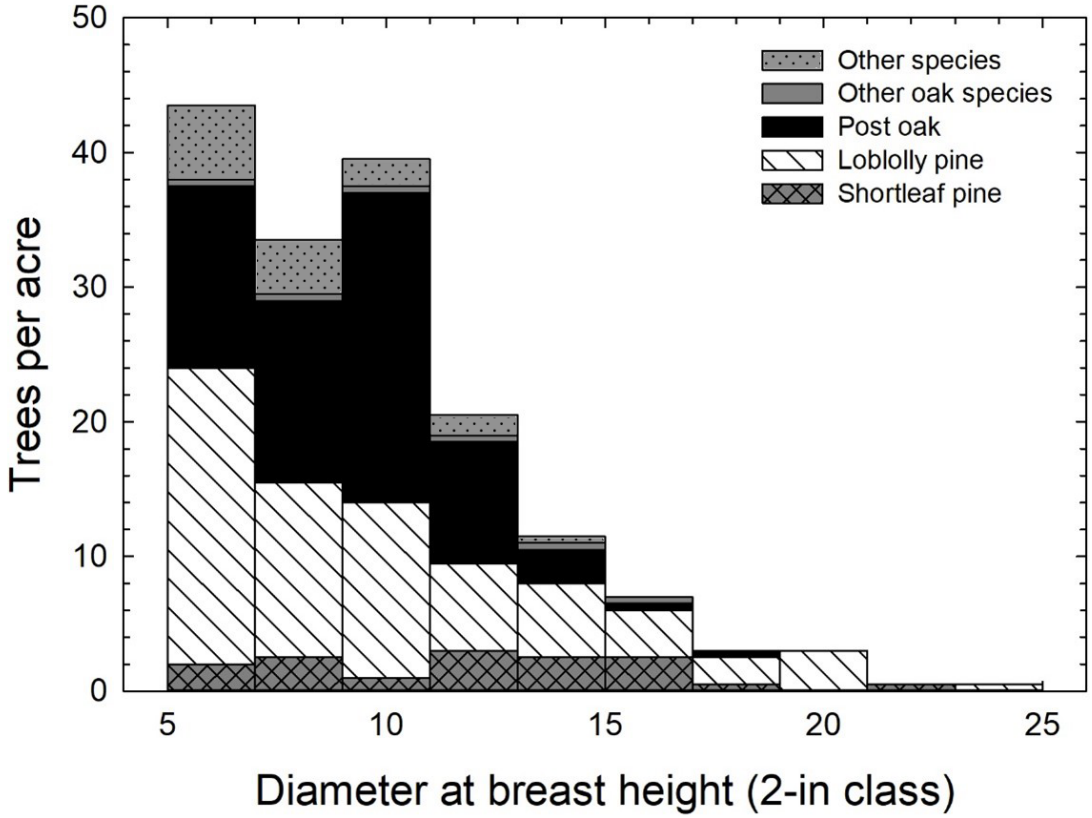

Figure 1—Diameter distribution (2-inch d.b.h. classes) by selected species groups of interest for the pine-hardwood stand at Boggy Slough Conservation Area.

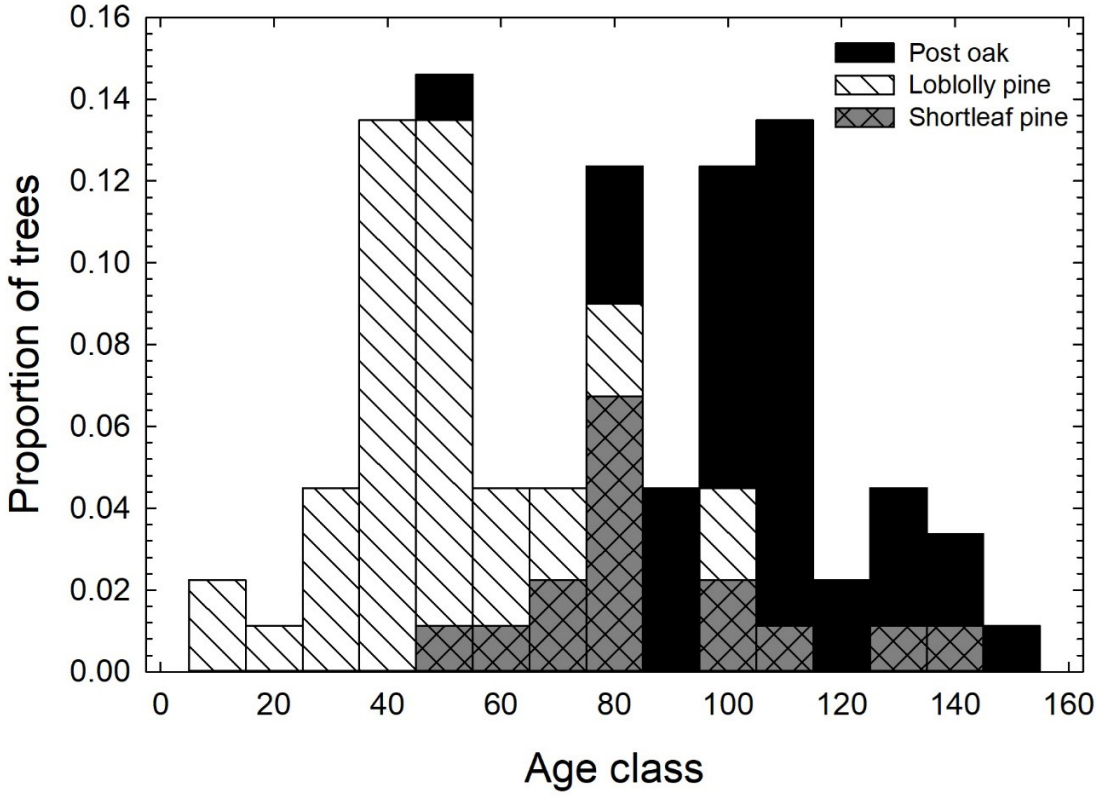

Figure 2—Age distribution (10-year age classes) by species for all cored trees from the pine-hardwood stand at Boggy Slough Conservation Area.

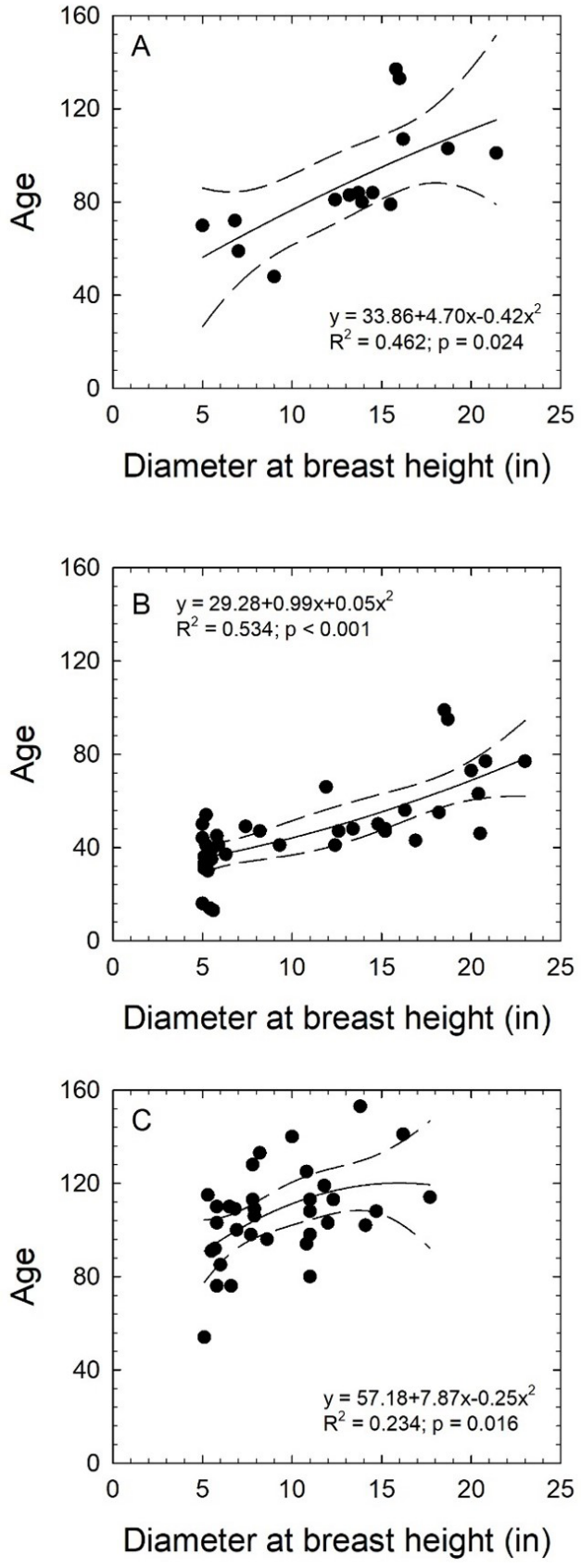

Figure 3—Relationship between diameter at breast height and tree age for (A) shortleaf pine, (B) loblolly pine, and (C) post oak.

The relationships between tree size and tree age also varied by species (fig. 3). Among the three species, tree size explained the greatest amount of variation in age for loblolly pine, with modeled estimates of age increasing more sharply for the largest trees. Across the range of tree sizes, shortleaf pines were older than loblolly pines at a given size, also showing a pattern of increased tree age associated with larger trees. Post oaks were the oldest across all tree sizes, with even the smallest trees (5 inches d.b.h.) being over 80 years old. Among the three species, tree size explained the least amount of variation in tree age for post oaks.

DISCUSSION

The stocking of the stand was generally consistent with a closed-canopy woodland. Recent studies and management guides have identified stocking levels associated with different types of open forest ecosystems, with <30 percent stocking associated with savannas, 30 to 55 percent stocking associated with open canopy woodlands, and 55 to 75 percent stocking associated with closed canopy woodlands (Dey and others 2017, Hanberry and others 2014). In general, at stocking levels in closed canopy woodlands, the trees fully occupy the growing space and maintain a closed canopy, but light levels beneath the canopy are greater than what would occur in a dense forest. Closed canopy woodlands can support abundant herbaceous vegetation in the understory, often dominated by forbs rather than graminoids due to the greater shading relative to open canopy woodlands or savannas (Knapp and others 2015, Peterson and others 2007).

The diameter distribution of the canopy trees in our study approximated a negative exponential, or reverse-J, form, showing greater abundance of small-diameter trees than large-diameter trees. In uneven-aged silvicultural systems, this structure is desirable, provided that the smaller trees are younger and represent the recruitment of new trees to the overstory to sustain the forest type through time (Guldin and Baker 1998). In our study, we observed a different pattern; the largest trees in the stand comprised older loblolly and shortleaf pines, and the smallest trees were dominated by older post oak and younger loblolly

pines. The lack of young post oak or shortleaf pine shows that recruitment of these species has not occurred in recent decades, suggesting the stand may be in the process of transitioning away from the shortleaf pine-post oak woodland community.

Taken together, our results suggest a few key areas in which stand dynamics of the site may be related to management history. First, post oak and shortleaf pine are absent from young age classes, suggesting that conditions for regeneration or recruitment to the overstory size class have been unfavorable for these species. In contrast, the abundance of loblolly pine in the younger and smaller size classes suggests that conditions for regeneration and recruitment have been favorable for it. It is likely that periods of fire exclusion over the past several decades have favored loblolly pine regeneration and reduced success of shortleaf pine and post oak. Second, the differences in age distributions among the species suggest past disturbance events may be associated with regeneration pulses of each species. For example, the effective age of most of the post oaks was 100 to 110 years old, aligning with early ownership by Southern Pine Lumber Company, a period in which initial timber harvesting was widespread and focused on pine species in the uplands. Finally, the differences in size among the species, particularly given the size-age relationships, suggests that edaphic factors may contribute to the growth rates and stand structure. Despite their age, post oaks were relatively small throughout the stand. However, the history of timber harvest is not known for the area, so it is unclear if the post oaks were previously suppressed trees that were left after a past timber harvest and remained small. Future analyses of radial growth will be used to identify potential release events and quantify growth rates.

CONCLUSIONS

We characterized the structure and age demographics of a shortleaf pine-post oak woodland in east Texas. During the past 50 years, loblolly pines have established to become the dominant species, contributing most of the existing basal area and occurring across all size classes. While stand stocking is within the range associated with closed canopy woodlands, our results suggest that the stand is transitioning away from shortleaf pine-post oak composition and towards greater loblolly pine dominance. Maintaining a shortleaf pine-post oak woodland into the future would require management actions to reduce loblolly pine ingrowth and increase the establishment and recruitment of shortleaf pine and post oak.

ACKNOWLEDGMENTS

Funding for this research was provided by the T.L.L. Temple Foundation through the Buddy Temple Visiting Scholar program. We thank Robert Sanders, Jacob Grochowski, and Ian Vaughan for field assistance and Drew Anderson, Abby Huffman, Luke Noel, and Ava Wood for their efforts with sample preparation.

LITERATURE CITED

Arthur, M.A.; Alexander, H.D.; Dey, D.C. [and others]. 2012. Refining the oak-fire hypothesis for management of oak-dominated forests of the eastern United States. Journal of Forestry. 110: 257–266. https://doi.org/10.5849/jof.11-080.

Bragg, D.C.; Hanberry, B.B.; Hutchinson, T.F. [and others]. 2020. Silvicultural options for open forest management in eastern North America. Forest Ecology and Management. 474: 118383. 14 p. https://doi.org/10.1016/j.foreco.2020.118383.

Bray, W.L. 1904. Forest resources of Texas. Bull. 47. Washington, DC: U.S. Department of Agriculture, Bureau of Forestry. 71 p.

Coile, T.S.; Schumacher, F.X. 1953. Site index of young stands of loblolly and shortleaf pines in the Piedmont Plateau Region. Journal of Forestry. 51: 432–435. https://doi.org/10.1093/jof/51.6.432.

Dey, D.C.; Kabrick, J.M.; Schweitzer, C.J. 2017. Silviculture to restore oak savannas and woodlands. Journal of Forestry. 115: 202–211. https://doi.org/10.5849/jof.15-152.

Gingrich, S.F. 1967. Measuring and evaluating stocking and stand density in upland hardwood forests in the central states. Forest Science. 13: 38–53. https://doi.org/10.1093/forestscience/13.1.38.

Guldin, J.M.; Baker, J.B. 1998. Uneven-aged silviculture, southern style. Journal of Forestry. 96: 22–26. https://doi.org/10.1093/jof/96.7.22.

Hanberry, B.B.; Bragg, D.C.; Alexander, H.D. 2020. Open forest ecosystems: an excluded state. Forest Ecology and Management. 472: 118256. 11 p. https://doi.org/10.1016/j.foreco.2020.118256.

Hanberry, B.B.; Bragg, D.C.; Hutchinson, T.F. 2018. A reconceptualization of open oak and pine ecosystems of eastern North America using a forest structure spectrum. Ecosphere. 9: e02431. 22 p. https://doi.org/10.1002/ecs2.2431.

Hanberry, B.B.; Jones-Farrand, D.T.; Kabrick, J.M. 2014. Historical open forest ecosystems in the Missouri Ozarks: reconstruction and restoration targets. Ecological Restoration. 32: 407–416. https://doi.org/10.3368/er.32.4.407.

Kabrick, J.M.; Clark, K.L.; D'Amato, A.W. [and others]. 2017. Managing hardwood-softwood mixtures for future forests in eastern North America: assessing suitability to projected climate change. Journal of Forestry. 115: 190–201. https://doi.org/10.5849/jof.2016-024.

Kenefic, L.S.; Kabrick, J.M.; Knapp, B.O. [and others]. 2021. Mixedwood silviculture in North America: the science and art of managing complex, multi-species temperate forests. Canadian Journal of Forest Research. 51: 921–934. https://doi.org/10.1139/cjfr-2020-0410.

Knapp, B.O.; Stephan, K.; Hubbart, J.A. 2015. Structure and composition of an oak-hickory forest after over 60 years of repeated prescribed burning in Missouri, U.S.A. Forest Ecology and Management. 344: 95–109. https://doi.org/10.1016/j.foreco.2015.02.009.

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and 'mesophication' of forests in the eastern United States. BioScience. 58: 123–138. https://doi.org/10.1641/B580207.

Peterson, D.W.; Reich, P.B.; Wrage, K.J. 2007. Plant functional group responses to fire frequency and tree canopy cover gradients in oak savannas and woodlands. Journal of Vegetation Science. 18: 3–12. https://doi.org/10.1658/1100-9233(2007)18[3:PFGRTF]2.0.CO;2.

Rogers, R. 1983. Guides for thinning shortleaf pine. In: Jones, E.P., Jr., ed. Proceedings of the second biennial southern silvicultural research conference. Gen. Tech. Rep. SE-24. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 217–225. https://www.fs.usda.gov/research/treesearch/1767.

Stambaugh, M.C.; Guyette, R.P.; Dey, D.C. 2007. What fire frequency is appropriate for shortleaf pine regeneration and survival? In: Kabrick, J.M.; Dey, D.C.; Gwaze, D., eds. Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Gen. Tech. Rep. NRS-P-15. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 121–128. https://www.fs.usda.gov/research/treesearch/12849

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2022. Web soil survey. https://websoilsurvey.nrcs.usda.gov/app/. [Date accessed: October 14, 2022].

EVALUATING THE TRAJECTORY OF VEGETATION SUCCESSION ON CLEARCUT AND SURFACE-MINED SITES

Ian Kennedy, Melody Mount, Wayne Clatterbuck, David Buckley, and Jennifer Franklin

ABSTRACT

Rates and pathways of secondary succession were investigated and compared between clearcut and surface-mined sites in east Tennessee using chronosequence techniques. Plots were established at 5-year age intervals ranging up to 30 years after the disturbances on sites with similar productivities. Clearcut sites were hypothesized to undergo vegetative succession more rapidly than surface-mined sites due to the difference in intensity and impact of disturbance on the substrate. A control site for each type of disturbance was also evaluated as a metric for the expected pathway of succession, community structure, and species composition (forest type). These control sites were selected for their lack of human disturbance and proximity to disturbed areas. Preliminary results suggest that clearcut sites are progressing away from their previous oak-hickory (*Quercus-Carya*) type at a more rapid successional rate than surface-mined sites as hypothesized. If an oak-hickory species composition is to be maintained after these stand initiation disturbances, silvicultural practices to encourage these species are recommended.

INTRODUCTION

Surface mining is an intense and severe form of disturbance with profound effects on vegetation, soil, and hydrology (Avera and others 2015). The result is often a bare site with a heavily compacted, rocky substrate, and little-to-no topsoil or seed bank. Strict regulations, implemented under the Surface Mine Control and Reclamation Act (SMCRA), include the requirement that sites must be revegetated following reclamation. However, in the effort to avoid noncompliance, many companies plant fast-growing vegetative cover which may not necessarily be native and can inhibit reforestation and recovery (Avera and others 2015). This does not mean that succession does not and cannot occur, rather that it is often greatly delayed. Most sites reclaimed since 2010 have incorporated the Forestry Reclamation Approach, which calls for reduced compaction and the planting of native tree seedlings (Burger and others 2017). As such, the revegetation of surface-mined sites presents a unique opportunity for study as they are a rare example of artificially induced primary succession and can provide insights useful to both surface mine restoration, and the process of succession itself (Wiegleb and Felinks 2001).

The mountainous regions of northeastern Tennessee provide numerous locations for monitoring and evaluating vegetative succession on surface-mined and clearcut sites. Additionally, the compositional trajectory of these heavily disturbed sites, postdisturbance, could provide insights not only into the rate of forest recovery on surface-mined and clearcut sites but also how they compare to the oak-hickory (*Quercus-Carya*) forest type common to the area.

The intent of this study was to compare vegetative growth and species composition between surface-mined, clearcut, and control sites. Due to the intensity and severity of such disturbances,

Ian Kennedy, Graduate Student, School of Natural Resources, University of Tennessee, Knoxville, TN 37996
Melody Mount, Graduate Student, School of Natural Resources, University of Tennessee, Knoxville, TN 37996
Wayne Clatterbuck, Professor, School of Natural Resources, University of Tennessee, Knoxville, TN 37996
David Buckley, Professor, School of Natural Resources, University of Tennessee, Knoxville, TN 37996
Jennifer Franklin, Professor, School of Natural Resources, University of Tennessee, Knoxville, TN 37996

surface-mined sites were hypothesized to lag behind clearcut sites in regeneration. In addition, because clearcut sites were expected to eventually develop toward the oak-hickory stands native to the region, the anticipated growth lag and poor site conditions, such as highly compacted soil, suggest that surface-mined sites were unlikely to show signs of oak or hickory growth within the scope of the study.

MATERIALS AND METHODS

In order to examine the change in clearcut and surface-mined sites over time, five age classes were established based on time since last disturbance: 0–5, 6–10, 11–15, 16–29, and ≥30 years. These were labeled as age classes 1 through 5, respectively, for analysis. The fourth age class covers a greater time period than prior age classes under the assumption that by that time, the canopy should have been closed. Temporal changes within the site would have been on a smaller scale due to the stability provided by a closed canopy microclimate (De Frenne and others 2021). Each age class was allotted a minimum of three sites for replication. Two sites were also selected as control sites (labeled as age class 6) which had not been disturbed in >50 years. This created a total of 32 sites, with one extra surface-mined site in the fourth age class. Sites were selected using historical images available via Google Earth Pro (version 7.3, Google LLC, Mountain View, CA) and were based on a series of criteria inherent in their disturbance type, age class, and other factors such as aspect, elevation, distance from other sites, and accessibility.

All sites selected had a northern or neutral aspect, to ensure similar access to light and other climate and weather effects, and similar elevation. Within age classes, sites that were geographically disparate from one another were also considered ideal to capture a more cohesive image of development across a wider geographic range. All sites were located on the Cumberland Plateau region of northeastern Tennessee. Sites were often clustered in certain areas that had landowner permission to access, and care was taken to avoid duplicating a given disturbance type and site age within each area. Elevation and geographic distance were considered secondary priorities and economic drivers of both timber and mining industries limited the availability of sites in some age classes.

Each site consisted of a single, circular, 0.10-acre plot, set randomly on the site location. Every tree with a diameter at breast height (d.b.h.) >1 inch was measured and identified by species.

Statistical Analysis

Average basal area per acre (BA) was calculated across sites and each measured specimen within each age class using the following equation:

$$BA = d.b.h.^2 \times 0.005454 \quad (1)$$

To achieve average BA for an age class, all BA for each site within a given age class were summed. Average BA did not meet the assumptions of normality and was modified with a square root transformation prior to analysis. Data were analyzed in a factorial design with disturbance type, age class, and their interaction as main effects. All analyses were conducted using SAS 9.4 (SAS 2013). Mean BA of disturbance types within age classes were determined with Duncan posthoc tests. Average d.b.h. and trees per acre were similarly analyzed.

The number of species was tallied and averaged for each age class and disturbance type, including controls. However, species count did not meet normality assumptions; thus it was necessary to conduct two nonparametric Kruskal-Wallis tests, which were carried out manually. One test included both disturbance types with controls, and one only tested age classes with no consideration for disturbance type. Mean ranks of disturbance types or age classes were separated with a Dunn's posthoc test.

Additionally, for the final age class of surface-mined, clearcut, and control sites, the contribution of each species to total BA was calculated.

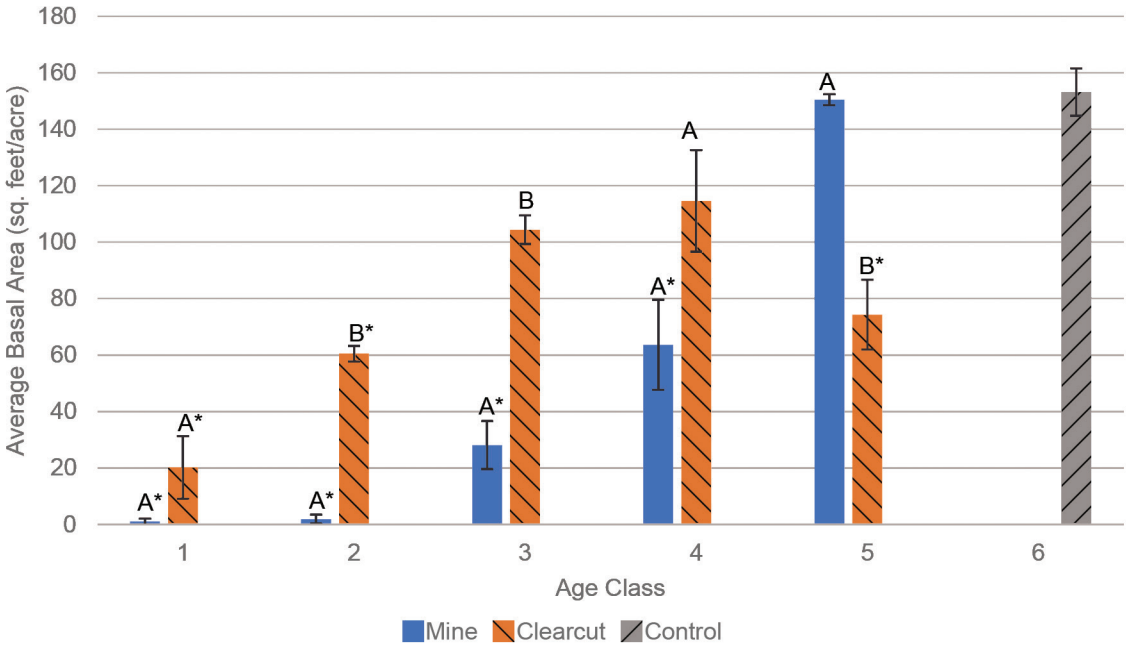

Figure 1—Average basal area by age class and disturbance type. Bars indicate one standard error from the average. Different letters indicate statistically significant differences between disturbance types at $P \leq 0.05$. Asterisk indicates significant difference from control at $P \leq 0.05$.

RESULTS

Average BA (fig. 1) showed a significant interaction between main effects of age class and disturbance type ($P < 0.0001$). When split by disturbance type, age class proved significant for surface-mined sites ($P < 0.0001$) and clearcut sites ($P = 0.0003$). On surface-mined sites, posthoc testing showed that age classes 1 and 2 were not significantly different from each other, but both differed significantly from age class 3. These, in turn, were significantly different from age class 4, which was significantly different from age classes 5 and the control. Age class 5 and the control did not differ significantly from each other. For clearcut sites alone, posthoc testing indicated that age class 1 and the control were significantly different from one another, but neither were significantly different from the other age classes on their own.

Additional posthoc testing was performed comparing average BA on clearcut and surface-mined sites by age class. Within each age class, both disturbance types were also compared to the control sites. In the first age class, clearcut and surface-mined sites were not significantly different from one another, but both were significantly different from control sites. Both site types were significantly different from one another in the second age class, and both clearcut and surface-mined sites were different from control sites. Surface-mined sites differed significantly from clearcut sites in the third age class, in addition to being significantly different from control sites. However, clearcut sites in the third age class no longer showed a significant difference from control sites. In the fourth age class, clearcut sites did not differ significantly from surface-mined sites, or from control sites, though surface-mined sites still differed significantly from control sites. In the fifth age class, control sites and surface-mined sites were not significantly different from each other, but clearcut sites were significantly different from both surface-mined and control sites.

Median species count (table 1) differed among disturbance types ($H = 6.23$ exceeded $\chi^2 = 5.99$, $P < 0.05$). However, when tested across all age classes regardless of disturbance type, no significant differences between age classes 1 and 3 showed that the number of species did not increase significantly with site age ($H = 9.06$ did not exceed $\chi^2 = 11.07$, $P < 0.05$). Average d.b.h. and trees per acre on surface-mined sites differed significantly from control sites, but they were not significantly different from clearcut sites.

Table 1—Average diameter at breast height (d.b.h.), trees per acre, species count, and their standard errors (SE) by disturbance type and age class

Disturbance type	Age class	d.b.h.		Trees per acre		Species	
		Average	SE	Average	SE	Average	SE
	years	~ ~ *inches* ~ ~		~ ~ ~ ~ ~ ~ ~ ~ ~ ~ ~ *n* ~ ~ ~ ~ ~ ~ ~ ~ ~ ~ ~			
Surface mined	**1 (0–5)**	0.9	0.9	23	23	0	0
	2 (6–10)	1.6	1.0	40	21	2	1
	3 (11–15)	2.9	0.3	417	148	9	1
	4 (16–29)	4.5	0.4	453	136	8	1
	5 (30+)	7.7	1.9	377	110	7	3
Clearcut	**1 (0–5)**	4.1	1.5	470	430	5	2
	2 (6–10)	3.7	0.8	630	223	9	2
	3 (11–15)	3.5	0.6	1,247	475	10	1
	4 (16–29)	4.9	0.7	627	208	8	0
	5 (30+)	4.7	0.1	363	47	8	2
Control	**6 (N/A)**	8.9	3.8	255	115	8	1

Species composition results were varied. On clearcut sites, yellow-poplar (*Liriodendron tulipifera*) composed the largest percentage of stems in every age class except the fifth, with the first three age classes having percentages well over 20 percent (60, 52, and 36 percent, respectively), and the fourth marginally lower at 19 percent. In the final age class, red maple (*Acer rubrum*) composed the majority of trees at 39 percent. Oak regeneration on clearcut sites was at a maximum 14 percent in the fourth age class, and overall made up a combined 5 percent of stems across all age classes. Oaks were not present at a significant count in the final clearcut age class. On control sites, 73 percent of the trees present were maple and American beech (*Fagus grandifolia*) and 6 percent of trees were oaks. The majority of trees on surface-mined sites overall were black locust (*Robinia pseudoacacia*) (23 percent), which was the most populous species except in the final age class, where yellow-poplar was dominant (29 percent).

BA and species composition combined also yielded interesting results in the older age classes. On undisturbed sites (fig. 2c), oak species made up a combined 40 percent of BA, with another 34 percent composed of maple species. The remaining percentage consisted of various other native species, including yellow-poplar (10 percent). This was not the case in the final age classes for surface-mined and clearcut sites. Surface-mined sites (fig. 2a) had no oak regeneration and were dominated by yellow-poplar (64 percent), maple species (17 percent), and eastern white pine (*Pinus strobus*) (15 percent). Clearcut sites (fig. 2b) were primarily yellow-poplar (29 percent) and red maple (25 percent) but did contain oak regeneration (22 percent).

DISCUSSION

In the first three age classes, succession proceeded as hypothesized, with clearcut sites recovering faster than surface-mined sites (fig. 1). However, in the fourth age class, while not yet comparable to the control sites, surface-mined sites were not significantly different from the clearcut sites. In the final age class, surface-mined sites showed no significant variation in BA from the control sites, unlike in clearcut sites. This directly contrasts with our expected outcomes, as it was anticipated that BA on surface-mined sites would continue to lag behind clearcut sites even into the fifth and final age class.

There are some additional factors that may have contributed to the differences in BA. In the third and fourth age classes, mined sites experienced significant variation in BA from each other, as

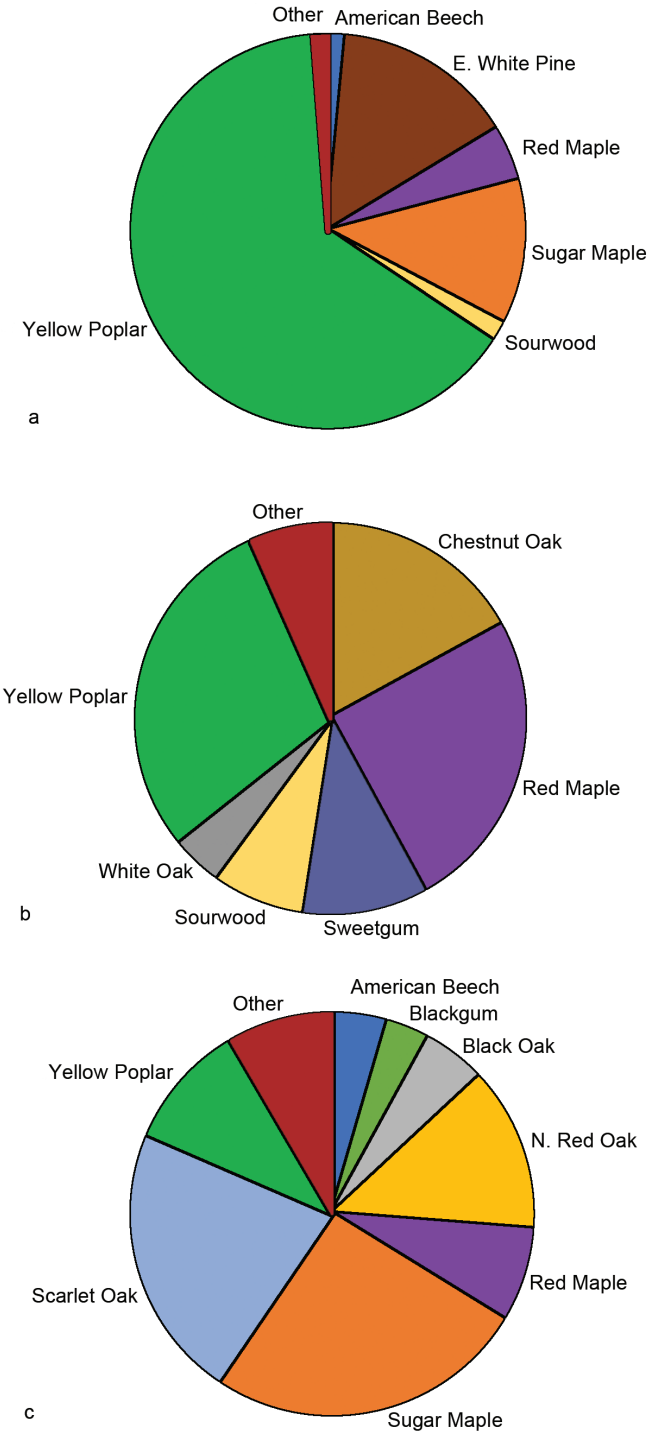

Figure 2—Species composition by basal area on (a) ≥30-year-old surface-mined sites, (b) ≥30-year-old clearcut sites, and (c) control sites. Species are American beech (*Fagus grandifolia*), black oak (*Quercus velutina*), blackgum (*Nyssa sylvatica*), chestnut oak (*Quercus montana*), eastern white pine (*Pinus strobus*), northern red oak (*Quercus falcata*), red maple (*Acer rubrum*), scarlet oak (*Quercus coccinea*), sourwood (*Oxydendrum arboretum*), sugar maple (*Acer saccharum*), sweetgum (*Liquidambar styraciflua*), white oak (*Quercus alba*), and yellow-poplar (*Liriodendron tulipifera*).

well as the first two age classes and the final age class. This suggests that, while younger age classes are, as predicted, limited by their site conditions, sites in the third and fourth age classes may have some unknown advantage or differing conditions from the younger sites. Two sites within the third age class were treated using the Forestry Reclamation Approach (Burger and others 2017). As such, due to the age of the sites in the fourth age class, a preponderance of fast-growing species such as pine and black locust could help explain the variations observed in BA.

Additionally, in the final age class, surface-mined sites showed no statistically significant variation from the control sites, and the same was true for clearcut sites. While those sites were at least 30 years removed from their last disturbance, there was no upward limit to their age. Thus, several of these older mined sites were likely last disturbed prior to the enactment of SMCRA in 1977. Many of these surface-mined sites prior to this point were left uncompacted, and occasionally even had their topsoil replaced with the previously removed spoil. As such, the success and trajectory of these sites may not be comparable to the eventual trajectory of the younger mined sites, specifically those that have been reclaimed under SMCRA.

While the number of species did not increase with age class, species composition provided an interesting perspective, namely the lack of oak regeneration in pure percentage of stems on both clearcut and mine sites. The species composition by BA provides additional insights (figs. 2a–c). While oak species are present on older clearcut sites and control sites, they did not dominate on clearcut sites, and were heavily rivaled on control sites by maples and to a minor degree, yellow-poplar, which dominated the clearcut sites. The advanced age of the older clearcut sites may be the reason. Control sites had not been disturbed in >50 years, and these clearcut sites were only required to be 30 years removed from the last disturbance. However, even on the control sites, the expected populations of hickory were entirely absent, and absent on clearcut sites as well.

Older mined sites lacked oaks entirely and were dominated by yellow-poplar, eastern white pine, and maples. The presence of eastern white pine was explained by differing policies of revegetation on older mined sites, as pines were commonly planted as a treatment for reclamation during that time. The dominance of yellow-poplar and maple could be due to early successional site conditions, which also could explain their large presence on older clearcut sites.

It is worth noting that there is potential evidence of highly limited regeneration of hickory species, and questionable regeneration of oaks, even in the apparent presence of larger reproductively capable adults on both clearcut and surface-mined sites. Neither site type seemed to be experiencing vegetative succession towards the expected oak-hickory forest type. The lack of hickory species on control sites was also worth noting. Both genera are intermediate in shade tolerance, occupy an intermediate successional stage, require a seed source to establish prior to disturbance, and grow slowly as seedlings, often succumbing to faster-growing vegetation. Both oak and hickory would not be prominent during these stages of succession unless their progeny were present prior to the disturbance, especially considering the substrate was removed on surface-mined sites.

A larger inventory of both surface-mined and clearcut sites may provide a more comprehensive analysis, though finding enough sites that meet the criteria to perform such an analysis would likely be challenging. Further study is recommended into the restoration and reforestation of surface-mined sites with the understanding that the growth of vegetation alone is not always a reliable indicator of the progress of restoration (Avera and others 2015). Additionally, the limited oak regeneration and lack of hickory regeneration is of concern, and future silvicultural studies and implementation of silvicultural practices based around these species is recommended.

LITERATURE CITED

Avera, B.N.; Strahm, B.D.; Burger, J.A.; Zipper, C.E. 2015. Development of ecosystem structure and function on reforested surface-mined lands in the Central Appalachian Coal Basin of the United States. New Forests. 46: 683–702. https://doi.org/10.1007/s11056-015-9502-8.

Burger, J.; Graves, D.; Angel, P. [and others]. 2017. The Forestry Reclamation Approach. In: Adams, M.B., ed. The Forestry Reclamation Approach: guide to successful reforestation of mined lands. Gen. Tech. Rep. NRS-169. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northern Research Station: 2-1–2-8. https://doi.org/10.2737/NRS-GTR-169.

De Frenne, P.; Lenoir, J.; Luoto, M. [and others]. 2021. Forest microclimates and climate change: importance, drivers and future research agenda. Global Change Biology. 27(11): 2279–2297. https://doi.org/10.1111/gcb.15569.

SAS Institute, Inc. 2013. Statistical analysis software. User's guide statistics version 9.4. Cary, NC: SAS Institute Inc.

Wiegleb, G.; Felinks, B. 2001. Primary succession in post-mining landscapes of Lower Lusatia—chance or necessity. Ecological Engineering. 17(2-3): 199–217. https://doi.org/10.1016/S0925-8574(00)00159-2.

Loblolly Pine

EXPLORING THE LINKAGE BETWEEN PHYSIOLOGY AND SHADE-RELATED MORTALITY IN LOBLOLLY PINE BRANCHES

Adelodun R. Majekobaje and Thomas J. Dean

ABSTRACT

The importance of crown size in forest management is well recognized, with crown length influencing tree growth and survival. Crown length can serve as a proxy for available photosynthetic area and potential tree growth which makes it an important component of process-based forest models. However, predicting the height to crown base and, hence, crown length, accurately remains challenging due to limited understanding of mechanisms of branch mortality. We conducted an experiment to elucidate the conditions that lead to the death of lower branches under shade conditions in loblolly pine (*Pinus taeda*). To do this, we explored the relationship between parameters of photosynthetic light response curves and branch mortality using point biserial correlation. Light saturated net assimilation rate (A_{max}), dark respiration rate (R_d), light compensation point (LCP), and light saturation point (LSP) showed the strongest relationship with branch mortality (r_{bs} >0.64). We concluded that shade-induced mortality in branches is associated with several physiological changes which include a decline in $A_{max,}$ $R_{d,}$ LCP, and LSP. Results from this study will help us predict crown size more accurately and improved prediction of crown size will ultimately lead to improved process-based models predicting stem growth and mortality.

INTRODUCTION

The importance of crown size has been widely recognized in forest management. It is a commonly used variable in various forest models, including growth, stem taper and volume, biomass, and mortality models (Gonzalez-Benecke and others 2014, Jucker and others 2016). The quantity of foliage on a tree is a key factor in determining its growth and survival, with longer crowns typically carrying more foliage than shorter ones (Antos and others 2010, Macfarlane and others 2007). Mortality in trees has been a topic of extensive research over the years; however, most of these studies have primarily focused on drought-induced mortality of whole trees (Anderegg and others 2015, McDowell and others 2008, Sperry and Love 2015, Venturas and others 2018) while branch mortality has received little attention. Light availability plays a crucial role in tree growth and crown development (Niinemets 2010) and light is a critical resource for the growth and survival of individual branches (Lone and Khan 2007). Photosynthesis, which is fundamental to branch growth and survival, is modulated by the amount of light available in the environment the branch is growing (Givnish 1988). For example, increased light intensity on foliage usually results in a corresponding increase in photosynthetic carbon assimilation per unit of leaf tissue but at very high light levels, photosynthetic rates may decrease as the excess light damages the leaf tissues (Ögren 1993).

The survival of a branch depends on its ability to extend horizontally and expose the terminal bud and leaves to light. However, in older (lower) branches, the cost of horizontal extension is high, and the productivity of the branches is low. Furthermore, under low light conditions there is insufficient production of energy (reduced photosynthetic activity), often resulting in reduced branch growth (Shao and others 2014)

Adelodun R. Majekobaje, Research Assistant, School of Renewable Natural Resources, Louisiana State University Agricultural Center, Baton Rouge, LA 70803

Thomas J. Dean, Professor, School of Renewable Natural Resources, Louisiana State University Agricultural Center, Baton Rouge, LA 70803

and limited carbon assimilation (Samuelson and Stokes 2012). Studies on *Pinus radiata* (Warrington and others 1989), *P. taeda* (Teskey and Shrestha 1985), and *Betula pendula* and *Populus tremula* (Tsel'niker 1979) have all shown that assimilation rates decrease with decreasing light intensities. Inadequate light can also affect the chlorophyll content in leaves, impairing the ability of a branch to efficiently convert light energy into chemical energy (Tang and others 2019).

Trees in closed-canopy forests often experience branch mortality and crown recession because of light limitation, which significantly affects their growth and development (Tinkham and others 2016). The length of the crown is a function of the height to the base of live crown (HCB) and total height of the tree. HCB is calculated as the distance from the base of the live crown to the tip of the highest live branch on the tree and can be a proxy of available photosynthetic area and potential tree growth. The base of the live crown is the point on the stem where the live branches and foliage begin and is typically measured as the distance from the ground to the first live branches on the tree. While easy to conceptualize, the measurement of HCB is a time-consuming process. As a result, several studies have concentrated on the development of predictive models that employ more readily available stand variables to estimate the HCB of trees (Macfarlane and others 2007). Most of these models are based on static measurements using total height, diameter at breast height, stand basal area, crown competition factor, and dominant height or site index as predictor variables (Sharma and others 2017); however, changes in HCB are dynamic. For example, the lowermost branch in the canopy typically dies from shading by the upper branches, thereby increasing HCB as the crown recedes.

Our ability to accurately predict HCB is further hampered by a limited understanding of the mechanisms underlying branch mortality in the lower crown. The main objective of this study was to measure and assess the effects of varying levels of light intensity on the photosynthetic properties of loblolly pine (*Pinus taeda*) branches. Specifically, in this study, we present an exploratory analysis of the relationships between mortality and parameters derived from light response curves of branch foliage developed under different shading conditions.

MATERIALS AND METHODS

Study Area

This study was carried out at Lee Memorial Forest in Washington Parish, southeast Louisiana (32.8°N, 90.0°W). The primary soil type in this forest was the Ruston series (well-drained, fine loamy, siliceous, thermic typic Paleudult). The forest was characterized by an average daily low and high temperatures of 12.0 °C and 25.6 °C, respectively, and mean annual rainfall of 1,600 mm (Dicus and Dean 2008).

Study Design

There was a south-to-north fertility gradient on this site. The southern part had a better site quality which decreased to the northern side. To account for this variation in soil fertility and separate its effects from the treatments being studied, two plots were established, one on the southern and the other on the northern side of the site.

Loblolly pine trees were planted in 2015 in field plots measuring 27 m by 27 m and with initial spacing of 3 m by 3 m. Nine trees per plot of similar height, at the edge of the plots (sun-exposed) and of desirable characteristics such as good form, vigor, and free of disease or insect damage, were selected for this study. Within each plot, one branch on each of three trees received one treatment of either no shade (0 percent), 30 percent shade, or 60 percent shade. Branch whorls were numbered on each tree from the top and an individual branch on the fifth whorl was selected as the target branch.

Shading Technique

Prior to bud burst in the spring, shade structures were installed on the 18 target branches on January 13, 2022 (fig. 1). Shade structures were constructed from concrete reinforcing wire, which was used to support the shading materials.

Figure 1—Shade structure with 30 percent Agfabric sunblock shade cloth on a target branch. (Photo courtesy of Adelodun Majekobaje)

Shading materials used were 30 and 60 percent Agfabric sunblock shade cloth. One shade covered one branch and was adjusted as the tree grew.

Gas Exchange Measurement

Monthly gas exchange measurements were carried out from August to November 2022. These were conducted on sunny days between 0930 and 1330 Central Daylight Time using the default light response program in a portable photosynthesis system (LI-6400xt, Li-Cor, Inc., Lincoln, NE). Attached to the system was a 6-cm^2 cuvette and a blue-red LED light source. For each measurement, two fully expanded fascicles from the first flush were selected. After each measurement, the diameter of the needles and the number of needles per fascicle were measured for data adjustment because the needle fascicles did not fill up the cuvette. The light response curve auto-program within the LI-6400xt system uses photosynthetic photon flux density (PPFD) levels of 1500, 1200, 1000, 800, 600, 400, 200, 100, 80, 60, 40, 20, and 0 μmol m^{-2} s^{-1}. Measurements were also carried out at PPFD level of 2000 μmol m^{-2} s^{-1} which was manually set to simulate full sunlight. In addition to the light-response parameters, stomatal conductance at light saturation point was also used in this study.

Prior to data analysis, gas exchange measurements were standardized with the leaf area per fascicle following the equation from Blazier and others (2018):

$$A = (N + \pi)\, 2LR \quad (1)$$

where

A = Area of needle fascicle (cm^2)

N = Number of needles per fascicle

L = Needle length (which is 3 because a 3-cm by 2-cm light source was used)

R = Radius of fascicle (cm)

Data Analysis

The study was a randomized complete block design with two blocks, three shade levels, and three replications. Eighteen branches were repeatedly measured in August and September while 17 and 13 were measured in October and November, respectively, due to mortality. In total, 66 measurements were carried out during the course of this study.

Statistical analysis on the data was performed in R (R Core Team 2023). A nonrectangular hyperbolic equation was fitted to individual light response curves using a least-square regression following the method of Marshall and Biscoe (1980) and light-response parameters were estimated (fig. 2). Relationships between branch mortality and parameters of light response curve were determined using point biserial correlation.

Parameters of Light Response Curve

Figure 2 shows a fitted light response curve and the derived parameters estimated from the curve. Light-saturated net assimilation rate (A_{max}) is the maximum photosynthetic capacity of a plant under high light intensity. Light saturation point (LSP) describes the point at which the photosynthetic rate reaches a plateau and does not increase commensurately with further increase in light intensity. Light compensation point (LCP) represents the equilibrium between energy production (photosynthesis) and consumption (respiration) in plants. It is the light intensity where the net assimilation rate becomes zero. Rate of respiration or dark respiration (R_d) is the

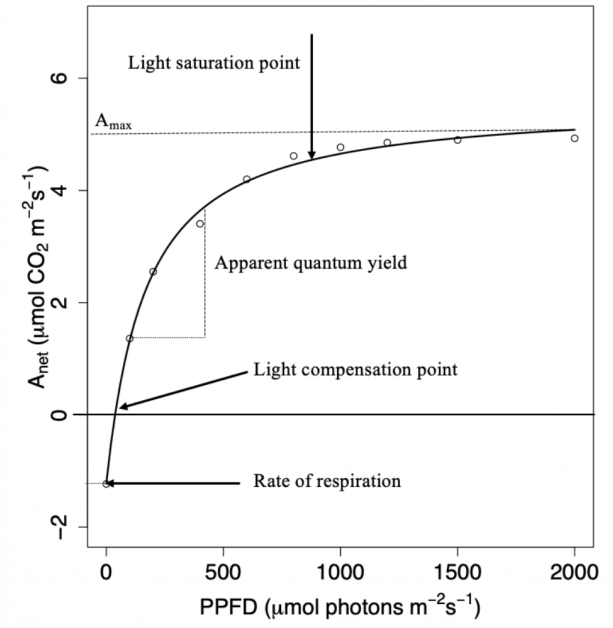

Figure 2—Fitted light response curve and the derived parameters. A_{max} = assimilation at light saturation, PPFD = photosynthetic photon flux density, A_{net} = net assimilation rate.

assimilation rate in the absence of light (i.e., when PPFD was zero).

RESULTS AND DISCUSSION

Observed Branch Mortality

At the end of the 2022 growing season, 11 branches had survived and 7 had died. Greatest mortality was observed in branches under the deepest shade (60 percent) where two of three branches died on both plots while branches under full light (0 percent) performed best with one death recorded on the northern plot. This is consistent with similar studies on whole-tree mortality that found improved growth and survivorship under high light intensity (Kunstler and others 2005, Lin and others 2002).

Relationship Between Branch Mortality and Parameters of Light Response Curve

Figure 3 shows the relationships between mortality and the derived parameters of light

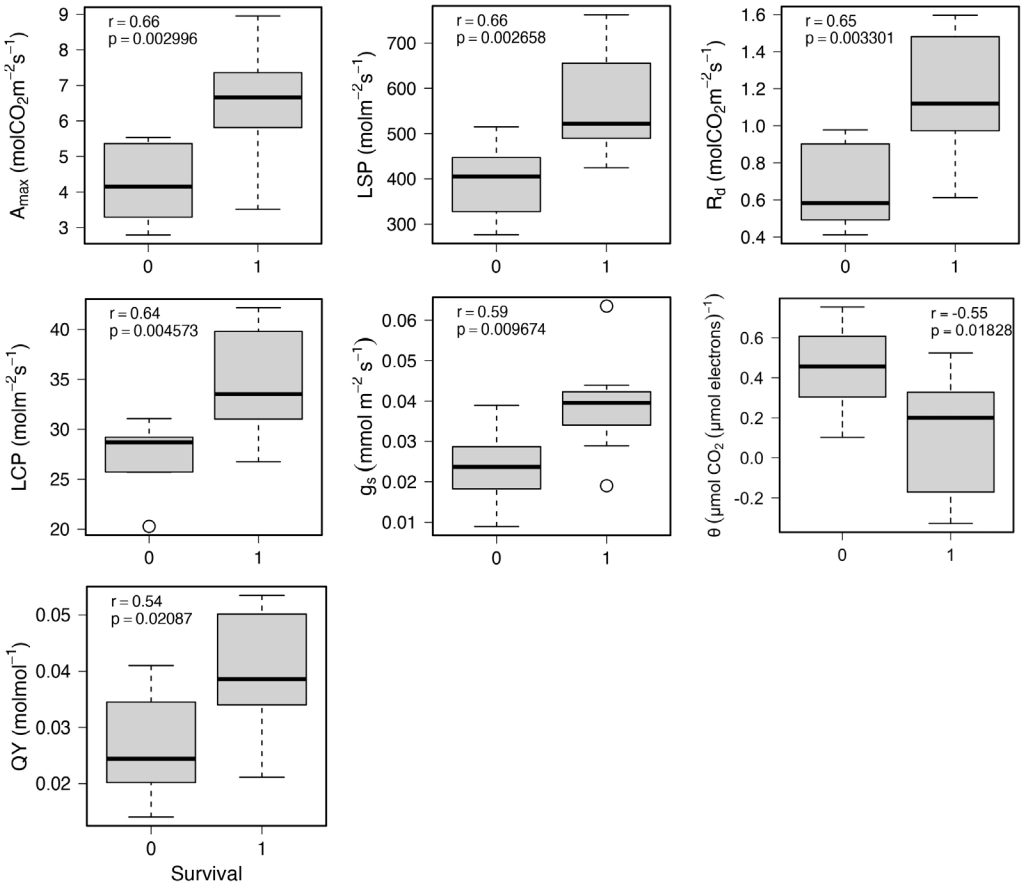

Figure 3—Biserial correlation between branch mortality and parameters of light response curve. 1 = branch alive (n = 11), 0 = branch died (n = 7). A_{max} = assimilation at light saturation, LSP = light saturation point, R_d = rate of respiration, LCP = light compensation point, g_s = stomatal conductance, θ = the convexity of the curve, QY = quantum yield.

response curves as well as stomatal conductance (g_s). Generally, A_{max}, LSP, LCP, and R_d are higher in plants grown under high light intensity (Bond and others 1999, Lewis and others 2000). Quantum yield (QY), which is the initial slope of the light response curve, describes the efficiency with which plants convert absorbed photons into fixed carbon through photosynthesis. Higher QY means more effective conversion of light energy to chemical energy while lower QY indicates lower photosynthetic efficiency (Genty and others 1989). Theta (θ) defines the convexity of the curve and how efficiently photosynthesis responds to changes in light intensity. When the convexity of light response curve is high, photosynthesis in the intermediate range is most effective (Ögren 1993).

In our study of planted loblolly pine, A_{max}, LSP, LCP, and R_d showed the strongest relationships with branch mortality while g_s, θ, and QY showed moderate correlation. In a drought response study, Mitchell and others (2013) found a strong relationship between A_{max}, and R_d and mortality in *Eucalyptus globulus*, *Eucalyptus smithii*, and *Pinus radiata* where they observed death in the three species 40, 50, and 115 days, respectively, after A_{max} reached 0 μmol m^{-2} s^{-1}. A similar trend of higher mean values in surviving branches was observed for all parameters except the convexity term (θ) where the opposite was the case. A similar pattern was observed in Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*), where convexity increased between the sun and shade needles, suggesting that photosynthetic efficiency at low light was higher (Lewis and others 2000). High θ values observed in the branches that died (which were mostly shade branches) suggests that these branches rapidly increased their photosynthetic rate with low light intensity, ensuring efficient light capture and utilization in limited light conditions (Ögren 1993).

CONCLUSION

Our study investigated the relationship between parameters of photosynthetic light response curves and mortality in loblolly pine branches grown under contrasting light conditions. The results revealed that branches under deepest shade experienced highest mortality rates while those under full sunlight showed lowest mortality. Among the derived parameters, A_{max}, LSP, LCP, and R_d exhibited the strongest correlation with branch mortality highlighting their significance in promoting growth and survival. Branches undergoing mortality exhibited decline in all these parameters. Shade reduces the chlorophyll content in leaves, which limits the branch's capacity to efficiently convert light energy to chemical energy and thus lead to a decline in A_{max}. Reduction in chlorophyll content also lowers the light absorption capacity of leaves and decreases the light compensation and saturation points. With declining A_{max}, energy (ATP) production is reduced and because dark respiration is dependent on the availability of ATP for cellular processes, dark respiration rate declined in dead branches. Our research contributes to a better understanding of the mechanism of branch mortality in trees by offering insightful information about the physiological responses of branches to shade conditions. It has the potential to improve predictions of tree crown size, growth, and biomass allocation. These improvements can enhance process-based forest growth models, ultimately supporting better forest management practices.

LITERATURE CITED

Anderegg, W.R.L.; Flint, A.; Huang, C-y. [and others]. 2015. Tree mortality predicted from drought-induced vascular damage. Nature Geoscience. 8: 367–371. https://doi.org/10.1038/ngeo2400.

Antos, J.A.; Parish, R.; Nigh, G.D. 2010. Effects of neighbours on crown length of *Abies lasiocarpa* and *Picea engelmannii* in two old-growth stands in British Columbia. Canadian Journal of Forest Research. 40: 638–647. https://doi.org/10.1139/X10-011.

Blazier, M.A.; Tyree, M.C.; Sword Sayer, M.A. [and others]. 2018. Gas exchange and productivity in temperate and droughty years of four eastern, elite loblolly pine genotypes grown in the Western Gulf Region. International Journal of Agronomy. 2018: 3942602. 10 p. https://doi.org/10.1155/2018/3942602.

Bond, B.J.; Farnsworth, B.T.; Coulombe, R.A.; Winner, W.E. 1999. Foliage physiology and biochemistry in response to light gradients in conifers with varying shade tolerance. Oecologia. 120: 183–192. https://doi.org/10.1007/s004420050847.

Dicus, C.A.; Dean, T.J. 2008. Tree-soil interactions affect production of loblolly and slash pine. Forest Science. 54(2): 134–139.

Genty, B.; Briantais, J.M.; Baker, N.R. 1989. The relationship between the quantum yield of photosynthetic electron transport and quenching of chlorophyll fluorescence. Biochimica et Biophysica Acta-General Subjects. 990(1): 87–92. https://doi.org/10.1016/S0304-4165(89)80016-9.

Givnish, T.J. 1988. Adaptation to sun and shade: a whole-plant perspective. Australian Journal of Plant Physiology. 15(2): 63–92. https://doi.org/10.1071/PP9880063.

Gonzalez-Benecke, C.A.; Gezan, S.A.; Samuelson, L.J. [and others]. 2014. Estimating *Pinus palustris* tree diameter and stem volume from tree height, crown area and stand-level parameters. Journal of Forestry Research. 25: 43–52. https://doi.org/10.1007/s11676-014-0427-4.

Jucker, T.; Avăcăriței, D.; Bărnoaiea, I. [and others]. 2016. Climate modulates the effects of tree diversity on forest productivity. Journal of Ecology. 104(2): 388–398. https://doi.org/10.1111/1365-2745.12522.

Kunstler, G.; Curt, T.; Bouchaud, M.; Lepart, J. 2005. Growth, mortality, and morphological response of European beech and downy oak along a light gradient in sub-Mediterranean forest. Canadian Journal of Forest Research. 35(7): 1657–1668. https://doi.org/10.1139/x05-097.

Lewis, J.D.; McKane, R.B.; Tingey, D.T.; Beedlow, P.A. 2000. Vertical gradients in photosynthetic light response within an old-growth Douglas-fir and western hemlock canopy. Tree Physiology. 20(7): 447–456. https://doi.org/10.1093/treephys/20.7.447.

Lin, J.; Harcombe, P.A.; Fulton, M.R.; Hall, R.W. 2002. Sapling growth and survivorship as a function of light in a mesic forest of southeast Texas, USA. Oecologia. 132: 428–435. https://doi.org/10.1007/s00442-002-0986-5.

Lone, P.M.; Khan, N.A. 2007. The effects of rate and timing of N fertilizer on growth, photosynthesis, N accumulation and yield of mustard (*Brassica juncea*) subjected to defoliation. Environmental and Experimental Botany. 60: 318–323. https://doi.org/10.1016/j.envexpbot.2006.12.013.

Macfarlane, C.; Arndt, S.K.; Livesley, S.J. [and others]. 2007. Estimation of leaf area index in eucalypt forest with vertical foliage, using cover and fullframe fisheye photography. Forest Ecology and Management. 242: 756–763. https://doi.org/10.1016/j.foreco.2007.02.021.

Marshall, B.; Biscoe, P.V. 1980. A model for C3 leaves describing the dependence of net photosynthesis on irradiance. Journal of Experimental Botany. 31(1): 29–39. https://doi.org/10.1093/jxb/31.1.29.

McDowell, N.; Pockman, W.T.; Allen, C.D. [and others]. 2008. Mechanisms of plant survival and mortality during drought: why do some plants survive while others succumb to drought? New Phytologist. 178: 719–739. https://doi.org/10.1111/j.1469-8137.2008.02436.x.

Mitchell, P.J.; O'Grady, A.P.; Tissue, D.T. [and others]. 2013. Drought response strategies define the relative contributions of hydraulic dysfunction and carbohydrate depletion during tree mortality. New Phytologist. 197(3): 862–872. https://doi.org/10.1111/nph.12064.

Niinemets, Ü. 2010. Responses of forest trees to single and multiple environmental stresses from seedlings to mature plants: past stress history, stress interactions, tolerance and acclimation. Forest Ecology and Management. 260(10): 1623–1639. https://doi.org/10.1016/j.foreco.2010.07.054.

Ögren, E. 1993. Convexity of the photosynthetic light-response curve in relation to intensity and direction of light during growth. Plant Physiology. 101(3): 1013–1019. https://doi.org/10.1104/pp.101.3.1013.

R Core Team. 2023. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Samuelson, L.J.; Stokes, T.A. 2012. Leaf physiological and morphological responses to shade in grass-stage seedlings and young trees of longleaf pine. Forests. 3(3): 684–699. https://doi.org/10.3390/f3030684.

Shao, Q.; Wang, H.; Guo, H. [and others]. 2014. Effects of shade treatments on photosynthetic characteristics, chloroplast ultrastructure, and physiology of *Anoectochilus roxburghii*. PLOS ONE. 9(2): e85996. 10 p. https://doi.org/10.1371/journal.pone.0085996.

Sharma, R.P.; Vacek, Z.; Vacek, S. [and others]. 2017. Modelling individual tree height to crown base of Norway spruce (*Picea abies* (L.) Karst.) and European beech (*Fagus sylvatica* L.). PLOS ONE. 12(10): e0186394. 23 p. https://doi.org/10.1371/journal.pone.0186394.

Sperry, J.S.; Love, D.M. 2015. What plant hydraulics can tell us about responses to climate-change droughts. New Phytologist. 207: 14–27. https://doi.org/10.1111/nph.13354.

Tang, X.L.; Jiang, J.; Jin, H.P. [and others]. 2019. Effects of shading on chlorophyll content and photosynthetic characteristics in leaves of *Phoebe bournei*. Journal of Applied Ecology. 30(9): 2941–2948. https://doi.org/10.13287/j.1001-9332.201909.002.

Teskey, R.O.; Shrestha, R.B. 1985. A relationship between carbon dioxide, photosynthetic efficiency and shade tolerance. Physiologia Plantarum. 63: 126–132. https://doi.org/10.1111/j.1399-3054.1985.tb02830.x.

Tinkham, W.T.; Hoffman, C.M.; Ex, S.A. [and others]. 2016. Ponderosa pine forest restoration treatment longevity: implications of regeneration on fire hazard. Forests. 7(7): 137. 19 p. https://doi.org/10.3390/f7070137.

Tsel'niker, Y.L. 1979. Resistances to CO_2 uptake at light saturation in forest tree seedlings of different adaptation to shade. Photosynthetica. 13: 124–129.

Venturas, M.D.; Sperry, J.S.; Love, D.M. [and others]. 2018. A stomatal control model based on optimization of carbon gain versus hydraulic risk predicts aspen sapling responses to drought. New Phytologist. 220: 836–850. https://doi.org/10.1111/nph.15333.

Warrington, I.J.; Rook, D.A.; Morgan, D.C.; Turnbull, H.L. 1989. The influence of simulated shadelight and daylight on growth, development and photosynthesis of *Pinus radiata*, *Agathis australis* and *Dacrydium cupressinum*. Plant, Cell and Environment. 12: 343–356. https://doi.org/10.1111/j.1365-3040.1989.tb01951.x.

CHARACTERIZING SPATIAL DEPENDENCIES OF COMPETING VEGETATION IN THINNED LOBLOLLY PINE PLANTATIONS

John B. Young, Bronson P. Bullock, and Cristian R. Montes

ABSTRACT

The spatial structure of vegetation influences the development of forest stands. Individual crop trees, which have either similar or dissimilar traits to their neighbors, are likely to be spatially dependent. In intensively managed plantations, the presence of competing vegetation has been shown to limit crop tree productivity, suggesting that the occurrence of undesired vegetation is also spatially dependent because it is competing for the same site resources. Silvicultural treatments applied throughout rotation systematically alter the spatial structure of forests to favor crop tree production, but any resultant competing vegetation growth is also influenced by changes in resource distribution. However, the spatial dependence of competing vegetation in intensively managed plantations is not well understood. A long-term regional study from the Southeastern United States which monitored the growth of intensively managed loblolly pine (*Pinus taeda*), as well as competing vegetation, was used to characterize the spatial dependence of competing vegetation following operational, mid-rotation silvicultural treatments. Competing vegetation was found to be spatially autocorrelated but was not found to differ between silvicultural treatments. Negative spatial autocorrelation, especially for herbaceous vegetation, occurred in response to postthin stand dynamics. Characterizing the spatial dependence of competing vegetation improves site-specific decisions for loblolly pine plantation management.

INTRODUCTION

Loblolly pine (*Pinus taeda*) plantations are the most important commercial pine resource in the Southern United States, covering nearly 50 million acres across the region (South and Harper 2016). The productivity of loblolly pine plantations has drastically increased throughout the last century due to the development of highly effective and site-specific silviculture (Fox and others 2007, Homyack and others 2022). Intensive plantation management often includes a combination of site preparation, fertilization, and the control of competing vegetation (Fox and others 2007). Mid-rotation silviculture treatments are common during intensive management and are typically applied in combination with thinning (Albaugh and others 2017, Liechty and Fristoe 2013). Plantation silviculture is primarily used to maximize the productivity of crop trees within a stand, and to reduce competition for site resources (Fox and others 2007).

Competition is an important factor contributing to the growth and yield of forest stands, and competition between individuals influences the development of stand attributes such as stem size and stand density (Liu and Burkhart 1994a, Miller and others 2003a, Pretzsch 1997). Competition occurs when resources are insufficient to meet the demands of all individuals within a stand. Measures of competition have most often been used to monitor the occurrence of interspecific competition at the stand level, represent some measure of stand density, and have been used to guide the timing and intensity of silvicultural treatments (Liu and Burkhart 1994b, Wagner and others 1996). The presence of competing

John B. Young, Graduate Student, Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA 30602

Bronson P. Bullock, Professor, Co-Director, Plantation Management Research Cooperative, Warnell School of Forestry and Natural Resources, Athens, GA 30602

Cristian R. Montes, Associate Professor, Co-Director, Plantation Management Research Cooperative, Warnell School of Forestry and Natural Resources, Athens, GA 30602

vegetation can also impact the productivity of crop trees, especially if not controlled during stand establishment (Miller and others 2003a,b). Periodic competition control may be applied throughout the rotation, but the impact of competing vegetation on stand development at mid-rotation is not as well documented (Albaugh and others 2012).

Significant competitive interactions imply dependency on the spatial distribution of site resources because individuals manipulate resources that are within their immediate spatial proximity (Cannell and others 1984, Mack and Harper 1977). When individual tree characteristics are either systematically similar or dissimilar to those of neighboring trees, spatial autocorrelation may be present (Reed and Burkhart 1985). The stem characteristics of plantation loblolly pines have been shown to be spatially autocorrelated, indicating that individual growth has a deterministic impact on neighborhood growth (Bullock and Burkhart 2005, Reed and Burkhart 1985). The presence of competing vegetation may also be spatially autocorrelated because its growth is also determined by the same site resource distribution. The spatial dependency of competing vegetation within intensively managed pine plantations has not yet been characterized. Characterizing the spatial autocorrelation of vegetation, in either crop or noncrop species, can increase the precision of stand growth and yield models by adjusting them to account for errors due to spatial variability (Bullock and Burkhart 2005).

A long-term, regional research field trial administered by the University of Georgia's Plantation Management Research Cooperative (PMRC) was used to assess the spatial autocorrelation of competing vegetation at mid-rotation. We hypothesized that competing vegetation in thinned pine plantations would have significant spatial autocorrelation. Additionally, we anticipated the degree of spatial autocorrelation to vary for different mid-rotation silvicultural treatments.

MATERIALS AND METHODS

Study Design

The mid-rotation treatment study was initiated in 2009 and consisted of 24 installations distributed evenly across the Lower Coastal Plain (LCP) and combined Upper Coastal Plain and Piedmont (UCPIE) physiographic regions of the Southeastern United States (fig. 1a). Installation locations were provided by PMRC member companies and site selection was based on an initial assessment of stem quality and heterogeneity. Sites were considered typical operational plantations within either region and were selected to have homogeneous stand characteristics. Each installation consisted of four treatment plots, with no replication at an installation. Treatments were thin only (T), T + chemical herbicide (H), T + fertilization (F), and a combined treatment of all three (T+F+H). Treatment plots were thinned to a residual basal area (BA) of 50, 70, or 90 square feet per acre and were a fifth-row thin with free thinning between rows to achieve the desired BA. Fertilization was applied at a rate of 225 pounds nitrogen per acre plus 25 pounds phosphorus per acre following thinning. The chemical vegetation control was broadcast sprayed following thinning and consisted of 3 ounces glyphosate and 3 ounces triclopyr (Garlon® 3A, Corteva Agriscience, Riverside, CA) mixed with 36 gallons of water per acre. Follow-up applications were applied to remove any remaining woody vegetation with a variable rate of triclopyr (Garlon® 4, Corteva Agriscience, Riverside, CA).

Measurement plots were 0.5 acres in size, and a total treatment plot size of 0.75 acres created a buffer zone to reduce the impact of edge effects from nontreated areas. Within each treatment plot, a competing vegetation survey was established (fig. 1b). The competing vegetation survey was twenty 4-foot-radius subplots arranged systematically between thinned rows and leave rows and represented roughly 5 percent of the plot area. At each subplot, measurements of multiple competing vegetation classes were collected (table 1). Large woody vegetation (>2 inches) was measured for diameter at breast

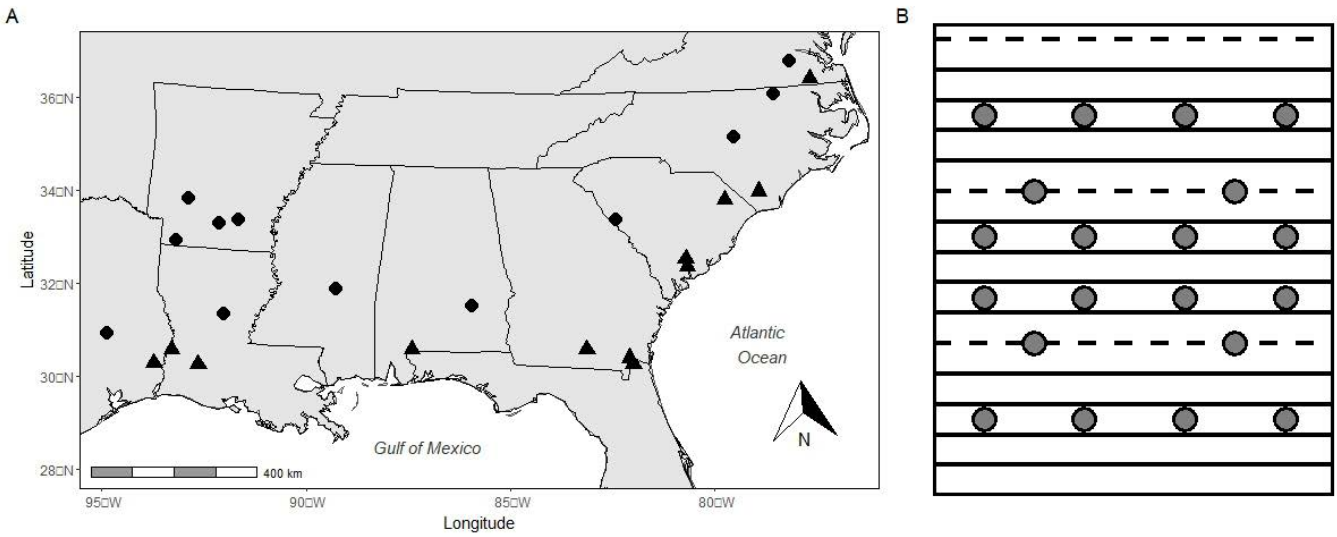

Figure 1—(A) Map of mid-rotation treatment study installation locations across the Southeastern United States. Triangles represent Lower Coastal Plain installations; circles represent Upper Coastal Plain/Piedmont installations. (B) An example measurement plot layout for each treatment plot within an installation. Solid lines represent leave rows, dashed lines represent take rows, and circles represent competing vegetation subplot locations. Subplot locations within a measurement plot were standardized at each location but differed between installations. The ratio of subplots located between leave rows and within take rows was the same for every installation.

height (d.b.h.) and height. A stem count and mean height by species groups was collected for small woody vegetation and shrub species. The groundcover percentage of herbaceous vegetation was ocularly estimated for broadleaf species, *Andropogon* spp., and grass species and further summed to estimate total herbaceous groundcover. From the competing vegetation measurements, three classes of competing vegetation were considered for this study: overstory hardwood, understory shrubs and hardwoods, and herbaceous vegetation. Preliminary analysis found that overstory hardwoods were present in only a small subset of subplots within either region, and were subsequently excluded from the remaining analysis. Competing vegetation measurements were taken every 2 years posttreatment, beginning in the second growing season after thinning.

Table 1—Summary of average competing vegetation density by treatment in each physiographic region

		Herbaceous density		Woody density	
Region	**Treatment**	**Groundcover**	**Primary vegetation**	**Crown volume**	**Primary species**
		percent		*cubic feet*	
Lower Coastal Plain	**Thin**	17.58	Broadleaf	68.30	*Ilex vomitoria*
	Thin + Herb	17.74	Broadleaf	44.71	*Ilex glabra*
	Thin + Fert	16.93	Broadleaf	73.02	*Ilex glabra*
	Thin + Fert + Herb	18.07	Broadleaf	76.46	*Ilex glabra*
Upper Coastal/ Piedmont	**Thin**	16.55	Broadleaf	135.20	*Liquidambar styraciflua*
	Thin + Herb	12.35	Broadleaf	113.34	*Callicarpa americana*
	Thin + Fert	16.58	Broadleaf	112.97	*Liquidambar styraciflua*
	Thin + Fert + Herb	16.19	Broadleaf	130.55	*Liquidambar styraciflua*

Herbaceous vegetation was categorized as either broadleaf, *Andropogon* spp., or grass during data collection
Values represent the mean across all subplot measurements collected and expanded to a per-acre basis.
Thin = thinning applied; Herb = chemical herbicide applied; Fert = fertilization applied.

Characterizing Spatial Dependence

Spatial dependence is present in forest stands when the characteristics of an individual tree are influenced by its local environment. Spatial autocorrelation in forest systems may either be positive or negative, depending on the proximity to neighboring trees and the underlying resource distribution. Positive spatial dependency indicates that the distribution of site resources has the greatest impact on individual development. For example, nutrient deposits at the toe slope of a hill may lead to a pocket of larger trees that have positively benefitted from favorable nutrient availability. Negative spatial autocorrelation occurs when competition between individuals is the primary influence on stand spatial structure. When individual stems are dissimilar to their neighbors, they are believed to be competing more vigorously for scarce resources. As trees age, a shift from positive to negative spatial correlation is likely (Bullock and Burkhart 2005, Reed and Burkhart 1985).

Spatial autocorrelation among competing vegetation was assessed using Moran's I (MI) statistic (Moran 1950). For a given set of spatial locations $S = \{s_i, ..., s_n\}$, MI is a global value representing the overall magnitude of spatial autocorrelation among observations for a desired characteristic of interest, $Z(s_i)$. For the set S, MI was calculated using the formula presented by Cliff and Ord (1973),

$$MI = \frac{N}{W} \frac{\sum_i^N \sum_j^N w_{ij}(x_i - \bar{x})(x_j - \bar{x})}{\sum_i^N (x_i - \bar{x})^2} \quad (1)$$

where

N is the total number of spatial observations in the set S

x is the variable of interest for either s_i or s_j

w_{ij} is the weight associated with relationship between the i^{th} and j^{th} observation when $i \neq j$

W is the sum of all weights

W was calculated as

$$W = \sum_i^N \sum_j^N w_{ij} . \quad (2)$$

The value of MI ranges between -1 and 1 and represents either perfectly clustered spatial data (1) or perfectly contrasted spatial data (-1). A value of 0 suggests the spatial variation in S is stochastic, indicating that observed values, $Z(s_i)$, are randomly distributed. An important consideration of the MI formulation presented by Cliff and Ord (1973) is the specification of the spatial weights, W. W accounts for a neighborhood structure between the observations in Z, adjusting the value to depend on the strength of their spatial proximity. Here, the individual weights, w_{ij}, make up the entries of W and are the Euclidean distances between each spatial observation (i.e., competing vegetation plot center) s_i and s_j, when $i \neq j$. Significance was determined using an α-level of 0.05.

MI was calculated for two competing vegetation groups, herbaceous and small woody vegetation (i.e., understory shrubs and hardwoods). Herbaceous vegetation was assessed using total groundcover percentage, and woody vegetation using total crown volume. Spatial coordinates were relative to the subplot layout within each treatment plot, and MI was calculated for every posttreatment measurement taken at each treatment and installation. A total of 205 observations in the LCP and 230 observations in the UCPIE were available for this analysis.

Silvicultural Assessment

To assess the impact of mid-rotation silvicultural treatments on the presence of spatial autocorrelation among competing vegetation, a two-way repeated measures analysis of variance (ANOVA) was fit to the MI values calculated for each physiographic region. The ANOVA model was

$$MI_{ijk} = \mu + I_{Herb} + I_{Fert} + (I_{Herb} \times I_{Fert}) + YST_k + \epsilon_{ijk} \quad (3)$$

Where MI_{ijk} is the MI value for each unique combination of measurement year (YST_k), installation and treatment. I_{Herb} and I_{Fert} are indicator values for each operational treatment, which constitutes a two-way factorial design at each installation.

RESULTS

MI for herbaceous vegetation in the LCP ranged from -0.145 to 0.203, and approximately 13 percent (27) of the 205 observations in the LCP were significantly spatially autocorrelated. MI in the UCPIE ranged between -0.144 and 0.130, and the proportion of significant observations for the UCPIE was slightly less than the LCP, with 12 percent (27) of 230 observations having significant autocorrelation. MI did not appear to be impacted by the different silvicultural treatments in either the LCP (fig. 2) or the UCPIE (fig. 3). Additionally, MI for any of the treatments did not appear to change with respect to time. The trends observed in figure 1 for herbaceous vegetation were confirmed by the ANOVA (table 2). The LCP results indicated that neither silvicultural treatment nor years since treatment significantly impacted the degree of spatial autocorrelation. In the UCPIE, treatment and years since treatment were also not significantly related to the magnitude of spatial autocorrelation. In both regions, the overall mean value for MI was significant (LCP, $F_{1,156} = 8.33$, $P = 0.004$; UCPIE, $F_{1,176} = 4.61$, $P = 0.033$).

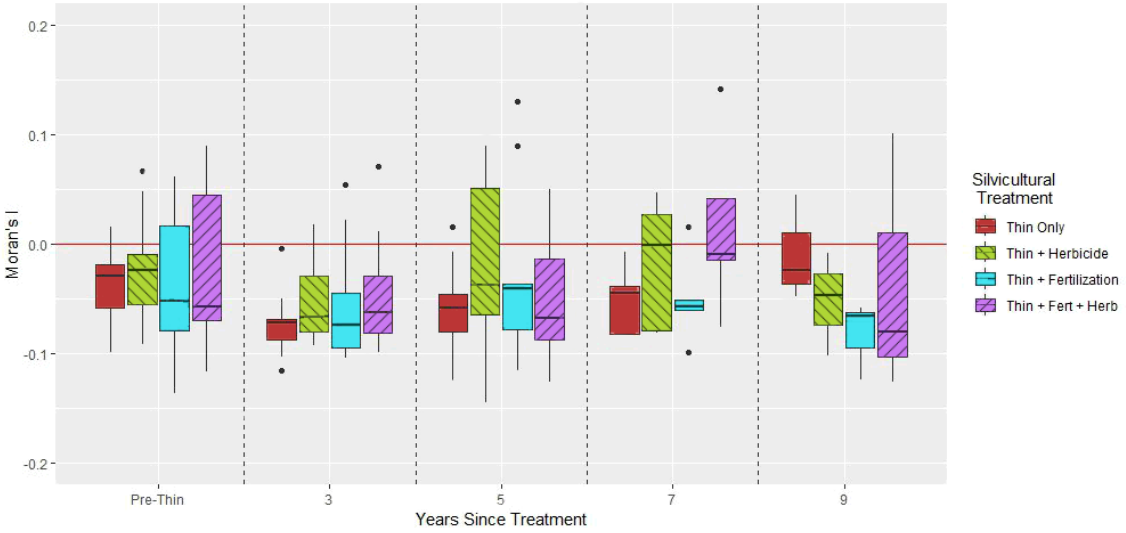

Figure 2—Distribution of Moran's I for herbaceous vegetation in the Lower Coastal Plain by treatment and measurement year (years since treatment). Moran's I values were calculated for each installation, with a possible range of -1 to 1. The red line at zero indicates complete spatial randomness for a treatment by years since treatment observation.

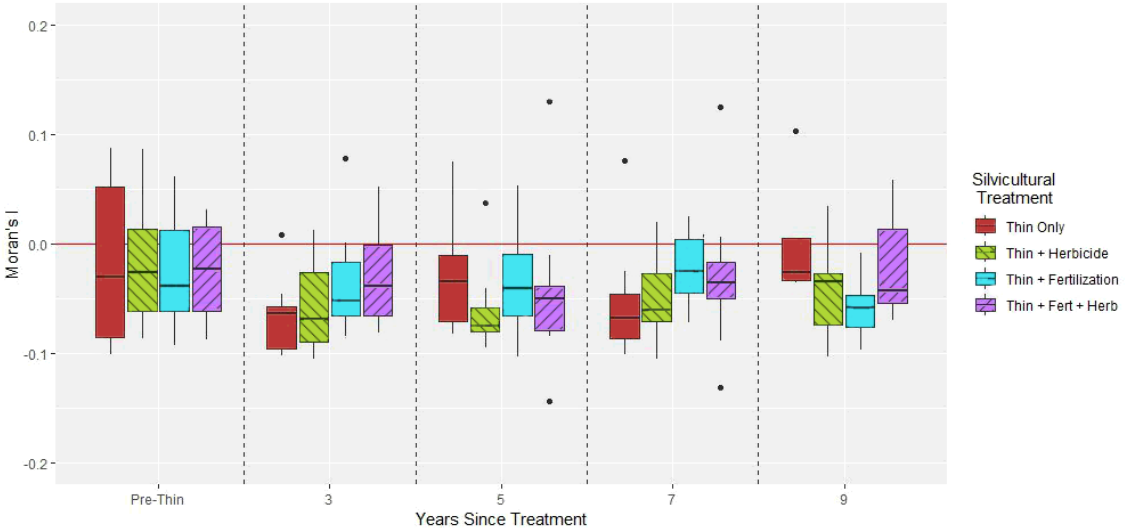

Figure 3—Distribution of Moran's I for herbaceous vegetation in the Upper Coastal Plain/Piedmont by treatment and measurement year (years since treatment). Moran's I values were calculated for each installation, with a possible range of -1 to 1. The red line at zero indicates complete spatial randomness for a treatment by years since treatment observation.

Table 2—Two-way, repeated measures analysis of variance results for Moran's I based on herbaceous vegetation

Region	Factor	Sum of squares	Degrees of freedom	F	*P*
Lower Coastal Plain	**Intercept**	0.029	1	8.33	0.004
	Herb	0.005	1	1.39	0.239
	Fert	0.000	1	0.00	0.978
	YST	0.026	4	1.85	0.122
	Herb*Fert	0.000	1	0.03	0.857
	Residual	0.548	156		
Upper Coastal/Piedmont	**Intercept**	0.012	1	4.61	0.033
	Herb	0.002	1	0.66	0.417
	Fert	0.000	1	0.11	0.736
	YST	0.018	4	1.70	0.152
	Herb*Fert	0.001	1	0.46	0.496
	Residual	0.468	176		

Significance was considered at the 95-percent confidence level (P <0.05).
YST = years since treatment; Herb = herbaceous vegetation control applied; Fert = fertilization applied.

MI trends among woody vegetation reflected those of herbaceous vegetation. Roughly 8 percent of observations were significant for woody vegetation for the LCP (16 of 203) and 9 percent for UCPIE (20 of 225). The range of MI values for woody vegetation were between -0.124 and 0.175 for the LCP, and between -0.116 and 0.156 for the UCPIE. As with herbaceous vegetation, there appeared to be no difference in MI values between silvicultural treatments or changes in values over time for the woody vegetation in either region (figs. 4 and 5). The ANOVA results were similar to those of the observed trends (table 3) for woody vegetation within both regions. The effects of treatment or time were not significant factors for spatial autocorrelation in either the LCP or UCPIE, respectively. The overall mean trend was still significant among woody vegetation in both the LCP ($F_{1,154}$ = 17.99, P <0.001) and UCPIE ($F_{1,167}$ = 20.365, P <0.001) and is consistent with the trend of negative spatial autocorrelation found for herbaceous vegetation.

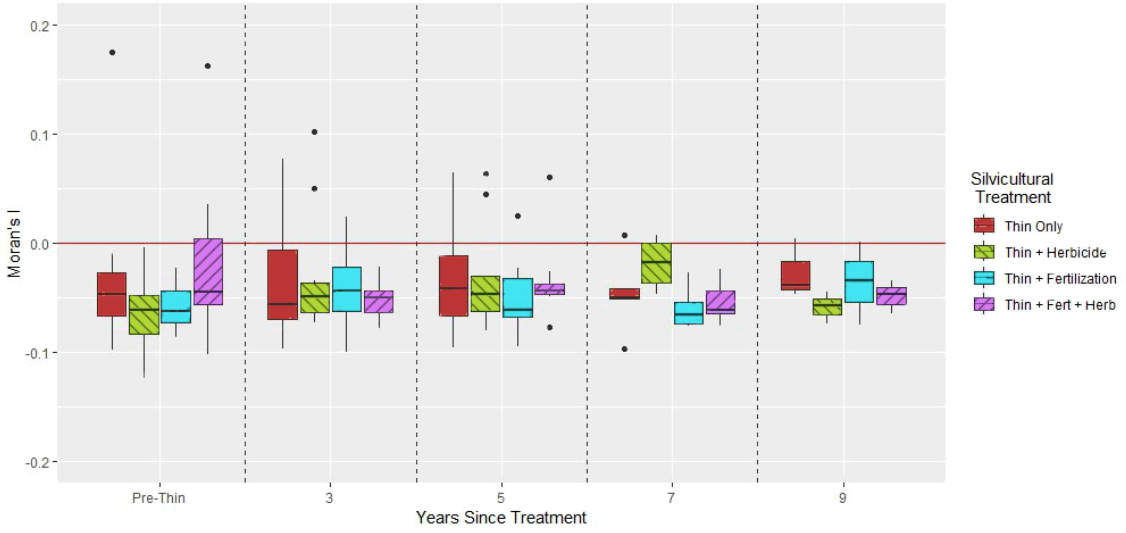

Figure 4—Distribution of Moran's I for woody vegetation in the Lower Coastal Plain by treatment and measurement year (years since treatment). Moran's I values were calculated for each installation, with a possible range of -1 to 1. The red line at zero indicates complete spatial randomness for a treatment by years since treatment observation.

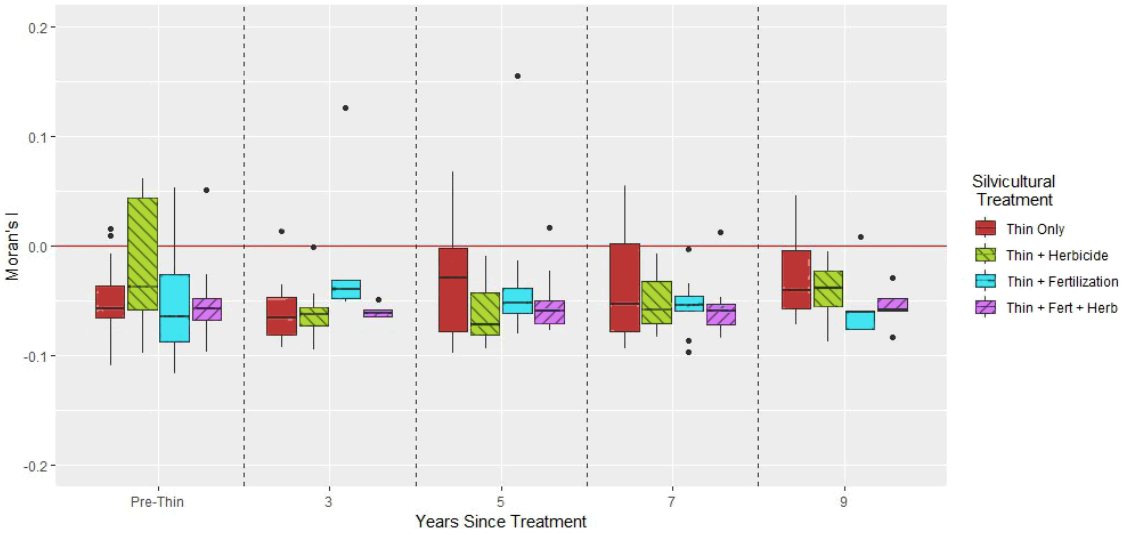

Figure 5—Distribution of Moran's I for woody vegetation in the Upper Coastal Plain/Piedmont by treatment and measurement year (years since treatment). Moran's I values were calculated for each installation, with a possible range of -1 to 1. The red line at zero indicates complete spatial randomness for a treatment by years since treatment observation.

Table 3—Two-way, repeated measures analysis of variance result for Moran's I based on woody vegetation

Region	Factor	Sum of squares	Degrees of freedom	F	*P*
Lower Coastal Plain	**Intercept**	0.036	1	17.99	0.000
	Herb	0.001	1	0.43	0.513
	Fert	0.005	1	2.29	0.132
	YST	0.001	4	0.17	0.952
	Herb*Fert	0.004	1	1.81	0.181
	Residual	0.309	154		
Upper Coastal/Piedmont	**Intercept**	0.038	1	20.92	0.000
	Herb	0.000	1	0.29	0.590
	Fert	0.000	1	0.05	0.823
	YST	0.002	4	0.21	0.931
	Herb*Fert	0.000	1	0.26	0.613
	Residual	0.307	171		

Significance was considered at the 95-percent confidence level ($P < 0.05$).
YST = years since treatment; Herb = herbaceous vegetation control applied; Fert = fertilization applied.

DISCUSSION

Competing vegetation was found to be spatially autocorrelated in thinned loblolly pine stands. For both the LCP and UCPIE, the number of significant MI values was relatively low but was higher for herbaceous vegetation (12–13 percent) than woody vegetation (8–9 percent). Observing higher degrees of spatial autocorrelation among herbaceous vegetation was not surprising. Herbaceous vegetation is known to be highly sensitive to light availability, and differences in spatial autocorrelation may have occurred in response to increased light availability following thinning (Anderson and others 1969). Herbaceous vegetation growth is also more impacted by seasonal fluctuation, and emergence patterns during the growing season may also contribute to the presence of spatial autocorrelation. Additionally, light availability beneath the main

canopy is altered during thinning, and patterns in the canopy may reflect spatial patterns in vegetation growth.

Mid-rotation silvicultural treatments strongly influence stand structure (Bailey and others 1989). Treatments such as row thinning also have an explicit spatial structure, and deterministic changes to stand structure systematically alter the distribution of site resources. It was anticipated these changes to stand structure would lead to a significant effect of mid-rotation silviculture on the spatial distribution of competing vegetation. However, treatments did not significantly impact spatial autocorrelation for either herbaceous or woody vegetation. For herbaceous vegetation, the need for increased light may reduce the influence of mid-rotation treatments on its spatial arrangement. All plots were thinned, which likely increased light availability within all plots along the same spatial pattern (i.e., within thinned rows). Effective broadcast applications of fertilizer and chemical herbicide should be uniformly distributed across space, and the resulting resource distribution should be uniform and promote consistent vegetation growth between treatment plots. Spatial autocorrelation among woody vegetation may be influenced by additional factors. Woody vegetation growth is not as seasonally influenced as herbaceous vegetation, and therefore changes in its spatial distribution over time may not be as easily observable. The thinning treatments in this study also reduced the amount of woody vegetation present within each stand, regardless of whether competition control was applied. Following thinning, woody vegetation growth did not recover to pretreatment levels at any location, whereas herbaceous vegetation was near pretreatment levels within 4 years posttreatment at most installations (results not shown).

Even though no difference in spatial autocorrelation was found between treatments, there was an overall trend of significant, negative spatial autocorrelation. Although the proportion of significant MI values were low for both herbaceous and woody vegetation, the average spatial autocorrelation was strongly significant for either population. Negative spatial autocorrelation is generally rare in practice but has been shown to be associated with plant competitive interactions (Griffith 2019, Mead 1968). Negative spatial autocorrelation between individuals implies competition because characteristics of neighbors are dissimilar (Griffith 2019). For the subplot data, negative spatial autocorrelation suggests that surrounding plots have either greater or less competition than neighboring subplots. One potential explanation of the negative spatial autocorrelation among subplots is vegetation response to thinning treatments and light availability. If thin rows are oriented to favor increased light availability following thinning (i.e., north-south vs. east-west), then subplots within thin rows may have higher competing vegetation due to increased light availability relative to neighboring subplots located in shade-dominated, nonthinned rows.

Another important consideration is the sampling design used to assess spatial autocorrelation. Competition for site resources occurs between individuals and should be used for modeling competitive interactions when possible (Burton 1993). However, density measures may be appropriate when production is described per unit of land area (Mack and Harper 1977). Here, only aggregate data at the subplot level is available. Although relative subplot locations still provide insight into the spatial distribution of understory vegetation, the spatial variation occurring at either finer or more coarse spatial resolutions was not detected. The results of this analysis indicate that competing vegetation is spatially dependent among subplots, but the degree of spatial autocorrelation between individuals and at the plot level was not determined.

Accounting for spatial autocorrelation among competing vegetation at mid-rotation can be beneficial. By understanding the connection between silvicultural treatments and their effect on stand dynamics, practitioners will be more informed of the impact that spatial relationships have on site-specific stand development. The presence of spatial autocorrelation means that stand structure is not random and neighboring observations are not independent (Bullock and Burkhart 2005). Growth and yield models are often used to make predictions of future stand growth. When incorporated into yield models, spatial autocorrelation can increase the precision of future yield predictions by

explaining the variation attributed to spatial relationships (Bullock and Burkhart 2005). Competing vegetation growth is generally variable and difficult to model (Liu and Burkhart 1994a). Modeling competition to account for spatial autocorrelation may reduce model variability, resulting in more reliable estimates of understory growth. Updated estimates of competing vegetation are likely to be more suitable for use as a covariate within pine plantation growth and yield models. Improving the estimation of covariates associated with pine productivity will subsequently improve site-specific management decisions.

CONCLUSIONS

Data from a long-term, regional research field trial in the Southeastern United States was used to characterize the spatial dependence of competing vegetation in thinned loblolly pine stands. Both herbaceous and woody vegetation was found to be negatively spatially autocorrelated. Stands receiving a mid-rotation silvicultural treatment application, namely fertilization, chemical herbicide, and the combination of the two, did not show significant differences in the degree of spatial autocorrelation present. Competition for light is a primary driver of herbaceous vegetation growth, and increased light availability following thinning is likely contributing to the spatial distribution of herbaceous vegetation following thinning. Factors contributing to the spatial distribution of woody vegetation are less clear and may be impacted by silvicultural treatment at the time of application. When present, the effects of spatial autocorrelation can be used to adjust growth and yield models and increase the precision of forest growth estimates. Including spatial variation may also improve models of competing vegetation growth. Future research is needed to explore methods for incorporating spatial autocorrelation and improved estimates of competing vegetation into growth and yield models used for pine plantation management.

LITERATURE CITED

Albaugh, T.J.; Fox, T.R.; Rubilar, R.A. [and others]. 2017. Post-thinning density and fertilization affect *Pinus taeda* stand and individual tree growth. Forest Ecology and Management. 396: 207–216. https://doi.org/10.1016/j.foreco.2017.04.030.

Albaugh, T.J.; Stape, J.L.; Fox, T.R. [and others]. 2012. Midrotation vegetation control and fertilization response in *Pinus taeda* and *Pinus elliottii* across the southeastern United States. Southern Journal of Applied Forestry. 36(1): 44–53. https://doi.org/10.5849/sjaf.10-042.

Anderson, R.C.; Loucks, O.L.; Swain, A.M. 1969. Herbaceous response to canopy cover, light intensity, and throughfall precipitation in coniferous forests. Ecology. 50(2): 255–263. https://doi.org/10.2307/1934853.

Bailey, R.L.; Burgan, T.M.; Jokela, E.J. 1989. Fertilized midrotation-aged slash pine plantations – stand structure and yield prediction models. Southern Journal of Applied Forestry. 13(2): 76–80. https://doi.org/10.1093/sjaf/13.2.76.

Bullock, B.P.; Burkhart, H.E. 2005. An evaluation of spatial dependency in juvenile loblolly pine stands using stem diameter. Forest Science. 51(2): 102–108. https://doi.org/10.1093/forestscience/51.2.102.

Burton, P.J. 1993. Some limitations inherent to static indices of plant competition. Canadian Journal of Forest Research. 23: 2141–2152. https://doi.org/10.1139/x93-267.

Cannell, M.G.R; Rothery, P.; Ford, E.D. 1984. Competition within stands of *Picea sitchensis* and *Pinus contorta*. Annals of Botany. 53: 349–362. https://doi.org/10.1093/oxfordjournals.aob.a086699.

Cliff, A.D.; Ord, J.K. 1973. Spatial autocorrelation. London: Pion. 178 p.

Fox, T.R.; Jokela, E.J.; Allen, H.L. 2007. The development of pine plantation silviculture in the southern United States. Journal of Forestry. 105(7): 337–347. https://doi.org/10.1093/jof/105.7.337.

Griffith, D.A. 2019. Negative spatial autocorrelation: one of the most neglected concepts in spatial statistics. Stats. 2: 388–415. https://doi.org/10.3390/stats2030027.

Homyack, J.; Sucre, E.; Maglaska, L.; Fox, T. 2022. Research and innovation in the private forestry sector: past successes and future opportunities. Journal of Forestry. 120(1): 106–120. https://doi.org/10.1093/jofore/fvab039.

Liechty, H.O.; Fristoe, C. 2013. Response of midrotation pine stands to fertilizer and herbicide application in the western Gulf Coastal Plain. Southern Journal of Applied Forestry. 37(2): 69–74. https://doi.org/10.5849/sjaf.11-044.

Liu, J.; Burkhart, H.E. 1994a. Modelling inter- and intra-specific competition in loblolly pine (*Pinus taeda* L.) plantations on cutover, site-prepared lands. Annals of Botany. 73(4): 429–435. https://doi.org/10.1006/anbo.1994.1053.

Liu, J.; Burkhart, H.E. 1994b. Spatial characteristics of diameter and total height in juvenile loblolly pine (*Pinus taeda* L.) plantations. Forest Science. 40(4): 774–786. https://doi.org/10.1093/forestscience/40.4.774.

Mack, R.N.; Harper, J.L. 1977. Interference in dune annuals: spatial pattern and neighbourhood effects. Journal of Ecology. 65(2): 345–363.

Mead, R. 1968. Measurement of competition between individual plants in a population. Journal of Ecology. 56(1): 35–45.

Miller, J.H.; Zutter, B.R.; Zedaker, S.M. [and others]. 2003a. Growth and yield relative to competition for loblolly pine plantations to midrotation—a southeastern United States regional study. Southern Journal of Applied Forestry. 27(4): 237–252. https://doi.org/10.1093/sjaf/27.4.237.

Miller, J.H.; Zutter, B.R.; Zedaker, S.M. [and others]. 2003b. Stand dynamics and plant associates of loblolly pine plantations to midrotation after early intensive vegetation management—a southeastern United States regional study. Southern Journal of Applied Forestry. 27(4): 221–236. https://doi.org/10.1093/sjaf/27.4.221.

Moran, P.A.P. 1950. A test for the serial independence of residuals. Biometrika. 37(1/2): 178–181. https://doi.org/10.1093/biomet/37.1-2.178.

Pretzsch, H. 1997. Analysis and modeling of spatial stand structures. Methodological considerations based on mixed beech-larch stands in Lower Saxony. Forest Ecology and Management. 97: 237–253. https://doi.org/10.1016/S0378-1127(97)00069-8.

Reed, D.D.; Burkhart, H.E. 1985. Spatial autocorrelation of individual tree characteristics in loblolly pine stands. Forest Science. 31(3): 575–587. https://doi.org/10.1093/forestscience/31.3.575.

South, D.B.; Harper, R.A. 2016. A decline in timberland continues for several southern yellow pines. Journal of Forestry. 114(2): 116–124. https://doi.org/10.5849/jof.15-006.

Wagner, R.G.; Noland, T.L.; Mohammed, G.H. 1996. Timing and duration of herbaceous vegetation control around four northern conifer species. New Zealand Journal of Forestry Science. 26(1/2): 39–52.

CLIMATE-SMART FORESTRY AND ITS IMPLICATIONS FOR SOUTHERN SILVICULTURE

G. Geoff Wang

ABSTRACT

Climate change is happening today and will continue in the future, producing many unexpected effects on our forests. In recent years, climate change has become the most critical driver for forest management, altering how we practice silviculture and forestry. Climate-Smart Forestry (CSF) is an emerging approach to sustainable forest management, with specific strategies and proactive practices aimed at promoting climate change adaptation to improve forest resilience for the benefits of neutralizing carbon emissions, sustaining productivity and provision of ecosystem services, and harmonizing environmental, social, and economic well-being. This new forestry paradigm will have significant implications for southern silviculture. In this paper, I first describe the recent rise of CSF. I then explain what is expected from CSF. Finally, I briefly discuss the implications of CSF to southern silviculture, using planted loblolly pine (*Pinus taeda*) forest management as an example.

WHAT IS CLIMATE-SMART FORESTRY (CSF)?

CSF is a new forestry paradigm that has recently emerged in response to contemporary greenhouse gas (GHG)-driven climate change. To understand how CSF differs from other forestry paradigms, I start by reviewing how forestry has evolved and then explain CSF.

Forestry has always been about managing forests to meet the needs of our society. However, society's need for forests and our perception of forests has always been changing. As a result, forestry has been an evolving concept over time. Forestry is rooted in timber management. For a long time, our society's needs from forests were timber—nothing else, which is best exemplified by this quote from President Theodore Roosevelt when he addressed the Society of American Foresters in 1903:

> The object of our forest policy is not to preserve the forests because they are beautiful . . . or because they are refuges for the wild creatures of the wilderness . . . but the making of prosperous homes (Roosevelt 1903).

Of course, this view has been long abandoned. Today, we value forests for so much more than their timber, and we care deeply about the health and integrity of our forests because forests are a critical part of the earth's life support system.

When examining its entire developmental history, forestry has experienced three paradigms: traditional forestry prior to the 1970s, ecological forestry from the 1980s to 2020, and CSF emerging around 2020. Traditional forestry was rooted in a human-centered, utilitarian view of forests, and silviculture was primarily focused on timber management. Ecological forestry was inspired by the modern environmental movement and great advances in ecology since the 1970s. Nature-based silviculture and ecosystem management approaches were developed to ensure the health and integrity of forest ecosystems and their associated goods and services. CSF is an emerging, new concept that directly responds to the threats from contemporary, GHG-driven climate change. It started in the European Union (Bowditch and others 2020, Verkerk and others 2020, Weatherall and others 2022) and has become popular in the United States in the last few years (Shephard and others 2022). I expect that CSF will promote forest

G. Geoff Wang, Clemson University, Department of Forestry and Environmental Conservation, Clemson, SC 29634

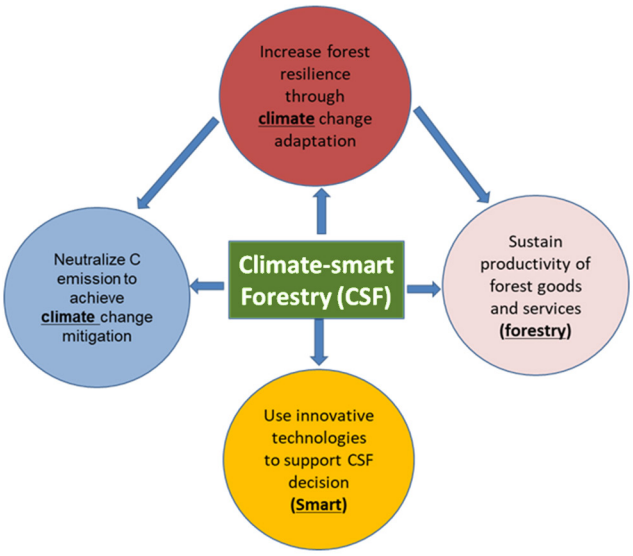

Figure 1—Climate-Smart Forestry (CSF) uses innovative technologies to support forestry decisionmaking to improve forest resilience, increase forest carbon sequestration capacity, and sustain all ecosystem goods and services for the benefit of society.

resilience through climate change adaptation so that forestry will become an important pathway for natural climate solutions (i.e., climate change mitigation).

CSF can be defined as an emerging approach to sustainable forest management, with specific strategies and proactive practices aimed at promoting climate change adaptation to improve forest resilience for the benefits of neutralizing carbon emissions, sustaining productivity and provision of ecosystem services, and harmonizing environmental, social, and economic well-being. As outlined in figure 1, the forestry part has never changed, which remains as providing goods and services to our society, but one service, climate change mitigation, is now taking the top priority. To meet this top priority, CSF must increase forest resilience through climate change adaptation while making appropriate tradeoffs with other ecosystem goods and services. To best deal with the limitations of data, the complexity of the system, and the uncertainty of climate change and its impacts, CSF must also apply new and innovative technologies to make informed management decisions. As a result, forestry has entered a new era known as CSF (fig. 1).

CSF is rooted in recent developments in climate forestry and smart forestry, and it can be viewed as a natural expansion from climate forestry with the use of new and innovative technologies (i.e., smart forestry). Climate forestry and carbon forestry are sometimes used interchangeably because they both aim at mitigating climate change through forest management. However, carbon forestry tends to have a narrower focus on carbon sequestration by planting large monocultures or managing existing forests in ways that maximize carbon storage so that carbon credits can be generated and sold on carbon markets to offset GHG emissions (Mitchell and others 2012). Climate forestry, on the other hand, manages forests to mitigate the impacts of climate change by enhancing the ability of forests to sequester carbon dioxide (CO_2) from the atmosphere, reducing GHG emissions from forest activities, and promoting the sustainable use of forest resources for environmental and socioeconomic benefits.

Smart forestry refers to the implementation of advanced digital technologies such as the Internet of Things, artificial intelligence (AI), big data analytics, and automation in the field of forestry, which is a natural evolution from digital forestry (Zhao and others 2005). It aims to optimize forestry practices by leveraging real-time data from sensors, drones, and other monitoring devices to make informed decisions that improve forest resilience, ecosystem services, and the communities that depend on them. The center of smart forestry is the development and application of forest digital twin (FDT), a virtual representation of a real-world forest ecosystem that uses real-time data to simulate the behavior and characteristics of the forest (Buonocore and others 2022), allowing the prediction of changes in stand dynamics by using early warning signals before the occurrence of unwanted events (e.g., forest decline).

WHY DO WE NEED CSF?

CSF is needed because forestry can expect to play a significant role in solving the climate change crisis. Specifically, practicing CSF can (1) improve forest resilience through climate

change adaptation, (2) increase forest carbon (C) sequestration to slow down climate change (i.e., climate change mitigation), and (3) make better management decisions under climate change uncertainty.

CSF: Improve Forest Resilience through Climate Change Adaptation

Climate change has significantly affected and will more likely disrupt forest resilience (Forzieri and others 2022). As a result, CSF must help improve forest resilience through proactive management under climate change uncertainty. To do so, CSF may promote forest resistance or recovery to maintain current conditions; alternatively, CSF may also manage for changes that promote a smooth transition from the current condition to a desired alternate state (fig. 2). Which strategy to use will depend on many factors, including tree species, site conditions, and climate change scenarios.

When future climate conditions and disturbances are within their historical range of variation, CSF will promote forest resistance or recovery to maintain current forest conditions (fig. 3). Should novel climate conditions and disturbances occur, CSF would promote a smooth transition to

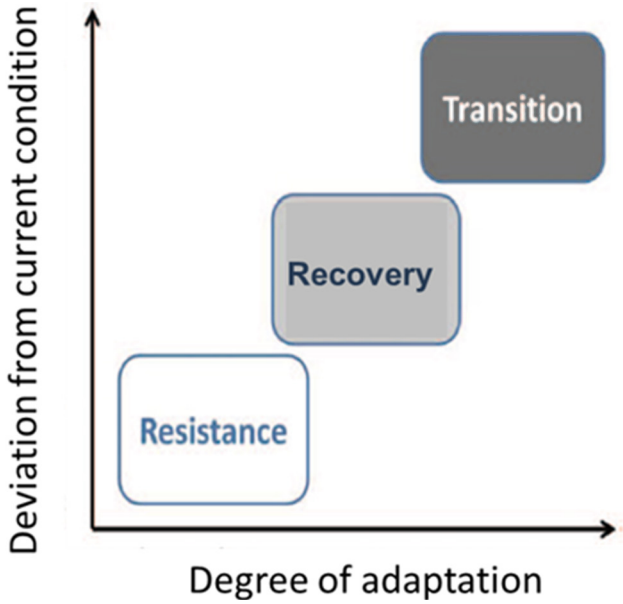

Figure 2—Forest resilience can be achieved by promoting resistance, recovery, and transition in response to climate change and disturbance. A larger deviation from the current forest condition would require a greater degree of adaptability. Modified from Nagel and others (2017).

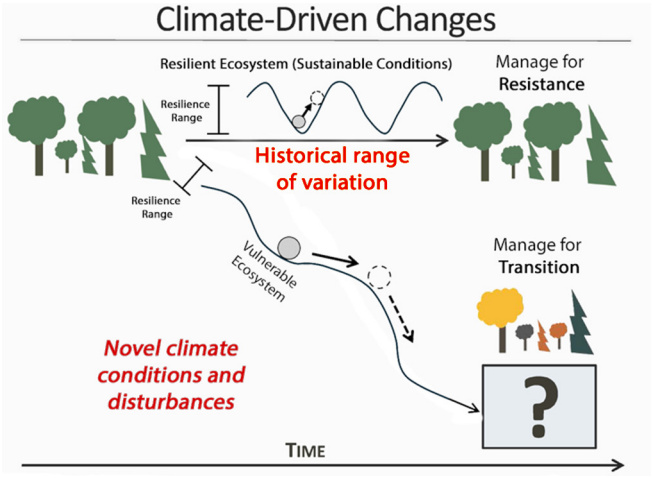

Figure 3—Hypothetical responses of current forests to climate change-driven disturbances. The resistance and/or recovery ability of current forests lead to the maintenance of current forest conditions when climate change is within the historical range of variation, but the forest will transition to a new alternate state in response to novel climate change (modified from https://www.adaptivesilviculture.org/silviculture-climate-adaptation).

desired future conditions. There is no doubt that novel climate conditions and disturbances are occurring and will continue to occur in many forests worldwide. Therefore, managing the transition becomes increasingly important, which is the most challenging aspect of CSF.

Millar and Stephenson (2015) described how management practices can influence the nature of transitions between forest types. Under novel climate conditions and disturbance, maintaining current forest conditions is not possible, and the current forest could collapse and degrade into an undesirable alternate state, resulting in the disruption of ecosystem goods and services. CSF anticipates that a change in the forest state is inevitable and takes necessary steps to ensure a smooth transition to desirable future conditions that maintain a continuous provision of ecosystem goods and services. To accomplish this desirable transition, CSF must first predict future climate conditions and disturbances, determine which suite of tree species (native or introduced) are suitable, and create a new forest community that will provide a similar structure and function to the existing forests. Each step represents a significant challenge that requires future research attention.

CSF: Increase Forest C Sequestration to Achieve Climate Change Mitigation

Terrestrial ecosystems are significant C sinks, absorbing 30 percent of our C emissions, mostly (90 percent) by forests (Keenan and Williams 2018). Therefore, a forest-based natural climate solution is an important strategy for climate change mitigation. Among the 20 natural climate solution pathways, forest pathways account for 50–70 percent of the total climate mitigation potential (Griscom and others 2017). However, realizing this potential will require increasing forest C sequestration through the help of CSF.

In general, CSF can increase forest C sequestration by following three strategies: (1) increase forest quantity (i.e., increasing afforestation and avoiding deforestation and degradation), (2) enhance forests' capacity to sequester and store carbon in situ within forest ecosystems (i.e., restoring C carrying capacity and mitigating C loss by disturbances), and (3) develop climate-smart commodities to store C *ex-situ* (i.e., increasing timber production and promoting wood utilization and substitution).

Increase forest quantity

An obvious and simple approach to increase forest C sink is to increase forest quantity in the world. This can be achieved first by increasing afforestation, which is effective but limited by competing land uses and the unsuitability of some lands for tree growth. Preventing deforestation is equally effective but requires good policies and their effective implementation. Today, deforestation remains a major problem in some countries and regions in the world, which requires international cooperation and support to incentivize forest protection. In the Southeastern United States, there is a great potential for losing forests to development, which exemplifies a significant challenge from competing land use. For example, the South Carolina Forestry Commission has estimated that about 1 million acres of forests (i.e., 7.8 percent of the total) may be lost to development by 2060 within the State[1].

Increase C storage in existent forests

Due to widespread timber exploitation and other historical human disturbances, many existent forests are currently recovering and have great C sequestration potential (CSP). In theory, the CSP of any forest can be calculated as the difference between the C carrying capacity (CCC) and the current C stock of the forest. CCC is the maximum amount of carbon capable of being stored by a forest ecosystem in a state of dynamic equilibrium under prevailing environmental conditions and natural disturbance regimes but excluding anthropogenic disturbance (Gupta and Rao 1994). For example, studies on forest CCC have been conducted at regional or national scales (Chen and others 2022, Liu and others 2014), but more research is urgently needed to determine the CCC of different forest types (Keith and others 2010) and, more importantly, to develop forestry practices to quickly realize the CSP, especially at stand and landscape scales.

Given the vast forest areas, only a small portion of forests may be actively managed in most countries in the world. Therefore, we should use a combination of passive and active restoration approaches to enhance forests' CSP. For those forests that are beyond our capacity of active management, we should protect them from disturbances, especially further human degradation. This passive forest restoration would allow natural succession to occur for the benefit of increasing forest C sequestration and storage. For those forests that can be actively managed, we should prioritize the management of stands or forest landscapes that have the most CSP, and developing stand- and landscape-specific CSF practices becomes the key to quickly realizing the CSP. Because heavily degraded forests, especially under favorable climate and site conditions, are often associated with high CSP, those stands should be first targeted for management. In the Southeastern United States, most forests that are actively managed are pine plantations. In South Carolina, pine plantations represent 24 percent of the total forest area. Therefore, how to better manage pine plantations becomes critically important to the success of CSF.

[1] Personal communication. 2023. S. Phillips, State Forester, South Carolina Forestry Commission, 5500 Broad River Road, Columbia, SC 29212.

In any forest, C storage will eventually approach the CCC, at which the forest will no longer function as a C sink. Indeed, the large C sink observed in recent years in the United States is a result of forest recovery from past forest destruction. It was reported that U.S. forests emitted 42,000 TgC (teragrams of carbon) between 1700 and 1935 due to forest clearing (agriculture, pasture, and development) but absorbed 15,000 TgC between 1935 and 2010 due to forest regrowth (Birdsey and others 2006). As this regrowth approaches the CCC, it is not clear if and how long the current rate of C sequestration will be maintained, which is another area of future research.

Under climate change, forests are likely to be subject to increasing disturbances. As a result, forest C storage is also facing a high risk of reversal (i.e., releasing CO_2 from forests to the atmosphere). More importantly, altered climate conditions and disturbance regimes could even lead to decreased CCC, forcing the release of stored C into the atmosphere during the process. In the Southeastern United States, significant stressors induced by climate change include more frequent and intense droughts and hurricanes, increased wildfire risk, and increased pressure from native and invasive pests, plants, and diseases. Therefore, more studies are needed to understand how climate-driven disturbances affect forest C storage and how this risk of reversal can be mitigated or at least factored in C-credit trading.

Ex-situ C storage through timber production, utilization, and substitution

In any stand or landscape, forest C storage would eventually saturate, reaching its C carrying capacity. At this point, further C sequestration is stopped, and the existent C stock is subject to C leakage due to natural disturbances. However, timber harvest and wood utilization can move the carbon sequestered by forests to *ex-situ* storage while increasing the CSP of the harvested forest at the same time. This represents a great opportunity for actively managed forests or plantations. To maximize this opportunity, we would (1) use innovative, climate-smart (i.e., minimizing C emission) silviculture to increase the production of timber (and/or other products), (2) promote timber and woody biomass utilization that allow long-term C storage in the products, and (3) substitute C emission-intensive products to obtain added C benefit. Previous studies have suggested that timber harvest and wood utilization, especially with the wood substitution for C emission-intensive products such as concrete and steel, can greatly increase forest C sequestration per unit area (Lippke and others 2011). However, adopting this solution would require more than silviculture and forestry.

CSF: Better Decisionmaking Empowered by AI, Intelligent Data Acquisition, and Digital Twin

To achieve CSF, we must take advantage of new and innovative technologies. Forestry decisionmaking is a complex process, especially under climate change uncertainty, which can benefit greatly from artificial intelligence (AI) (fig. 4). For example, AI can improve our predictions of future forest states by better overcoming

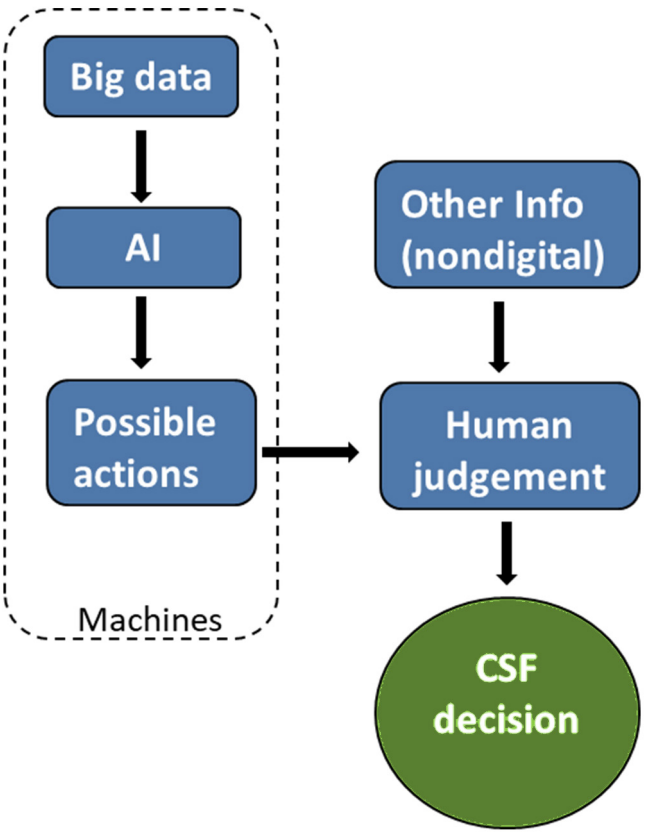

Figure 4—A Climate-Smart Forestry decisionmaking model that combines AI and human judgment [modified from Colson (2019) at https://hbr.org/2019/07/what-ai-driven-decision-making-looks-like].

data limitations, incomplete knowledge and understanding, and uncertainty in future climate change, when compared to existent modeling approaches. To be effective, CSF should adopt an evidence-based practice (EBP) that has been widely used in medicine. EBP integrates the best available research evidence, local management expertise, and landowners' values and needs into CSF decisionmaking. AI can enable efficient, objective, and systematic knowledge synthesis in the era of information explosion so that the best available research evidence is readily for CSF decisionmaking.

Better decisionmaking can only become possible with good data. CSF must develop and deploy innovative technologies in data collection. For example, a forest intelligent data acquisition system (FIDAS) powered by AI and robotics can be developed to acquire data in real time through active remote sensing. Specifically, autonomous drones with various specialty cameras and sensors onboard collect data both above and below the forest canopy with accuracy and efficiency in real time, which will greatly enrich the sparse ground data (e.g., forest inventory data) and low-resolution satellite data. It is expected that the development and application of FIDAS will revolutionize forest data collection and significantly advance forest research and management.

Better decisionmaking also needs better predictions under future climate change uncertainty. CSF can develop and apply a FDT platform to facilitate data-driven, real-time predictions. FDT is a set of virtual models that mimics the structure, composition, dynamics, and states of the physical forest, which can be constructed at any given spatial scale (from stand to global scales). The FDT platform will produce better predictions of future forest states under different scenarios of climate change and climate-driven disturbance over different temporal resolutions. By combining results from simulations and predictions with 3D virtual reality, FDT can help forest researchers and managers to visualize the states of future forests in real time. By integrating with the explainable AI method, FDT can identify and illustrate the factors that are important to the development of CSF management policies, strategies, and practices.

IMPLICATIONS OF CSF FOR SOUTHERN SILVICULTURE

CSF requires us to modify how we practice southern silviculture. In the Southeastern United States, active forest management is mostly conducted for southern pine, mainly planted loblolly pine (*Pinus taeda*). Loblolly pine plantations are one of the most globally productive timberland systems, representing 22 percent of the total forest area and an astonishing 51 percent of total forest growth in the southern region (Oswalt and others 2019). Under CSF, southern planted pine forests can be managed effectively to achieve synergy between economic goals and natural climate solutions. Therefore, planted loblolly pine forest management is used as an example to illustrate some possible changes in silviculture under CSF.

Manage for Forest Resilience

Maintaining forest health and productivity is of paramount importance under climate change. Therefore, CSF must promote forest resilience under climate change uncertainty. To do so, current silvicultural practices must be adapted to major stressors under climate change [e.g., temperature rise, drought, fire, and southern pine beetle (*Dendroctonus frontalis*) outbreaks]. Using a landscape-level, instead of stand-level, management approach, silvicultural practices developed under CSF should aim to 1) increase species and structural diversity (stands of different ages or structures), 2) improve functional complexity (diversity and redundancy of tree functional types), and 3) optimize spatial connectivity of forest stands in terms of seed dispersal and establishment in forested landscapes (Messier and others 2019).

For example, we can improve the functional complexity of a planted loblolly forest-dominated landscape in two ways. First, we may increase the presence of other native tree species with different functional attributes. Specific silvicultural practices would include 1) promoting live oak (*Quercus virginiana*) and slash pine

Figure 5—An illustration of forest resilience in two forested landscapes with different functional complexity (diversity and redundancy of tree functional types). Modified from Messier and others (2019).

(*Pinus elliottii*) in coastal areas to increase forest resilience in response to salt spray and sea-level rise, or 2) increasing longleaf pine (*P. palustris*) component to better cope with increasing activities of drought, fire, and southern pine beetles. Second, we may introduce new functional attributes not currently present in the landscape. Specific silvicultural practices would include 1) expanding slash pine northward in anticipating temperature rise, or 2) promoting the northern expansion of shortleaf pine (*P. echinata*) to cope with temperature rise and increased fire frequency.

It has been conceptualized that improving functional complexity in forested landscapes would increase forest resilience (fig. 5). A landscape dominated by planted loblolly pine forests may be vulnerable to climate change, but a landscape with planted pine and many other stand types, when mixed in a desired proportion and spatial pattern, is likely resilient to climate change. For example, under a mean fire frequency of 3 years, a fire-free interval of 15 years would only occur once every 1,000 years based on the equations of Hoffmann and others (2012). Given the specific fire-free interval required for canopy recruitment of different tree species in southern forests, an increase in wildfire frequency to ≤4 years due to climate change would practically eliminate loblolly pine, oaks, and other hardwood forests in the landscape. These forests would transition to longleaf, shortleaf, and slash pine forests if these species were present in the landscape. As a result, the functional complexity of a forest landscape becomes a prerequisite to this transition under increased fire frequency induced by climate change.

Figure 6—An example illustrating that functional complexity promoted forest resilience under a novel disturbance: American chestnut-dominated forests are replaced by oak-dominated forests due to the exotic invasive species— chestnut blight.

A good example showing how functional complexity promotes forest resilience in response to a novel disturbance is the transition of American chestnut (*Castanea dentata*)-dominated forests to oak (*Quercus* spp.)-dominated forests because of a novel disturbance—chestnut blight (fig. 6). The example demonstrated the inherent resilience of eastern hardwood forests, probably due to their functional diversity and redundancy of tree species. Therefore, it is of great importance for climate-smart silviculture to promote functional diversity at both stand and landscape scales.

Manage for Forest C Sequestration

One of the key CSF objectives is leveraging forest management as a natural climate solution. Under CSF, planted loblolly pine forest management must emphasize C sequestration. Enhancing forest C sinks of planted southern pine forests can help our nation to meet emissions-related goals such as the Paris Climate Agreement (Grassi and others 2017). To do so, we can build on our comprehensive understanding of planted southern pine forest C pools and fluxes (Kinerson and others 1977).

Increase timber production

Planted loblolly pine forest management has been focused on increasing timber production through tree improvement and innovative silvicultural practices. For the 60 years from 1940 to 2000, silvicultural treatment and tree improvement increased timber production of planted loblolly pine forests by about 400 percent (Fox and others 2007). Therefore, continuing to improve on our past success in loblolly pine silviculture and tree improvement is critical for increasing C sequestration.

Develop climate-smart practices

To maximize timber production, even-aged management with short rotations has been widely applied to planted loblolly pine forests. However, following a clearcut, the stand initiation can be a significant C source for up to 10 years. To minimize this C source, we may want to explore if there are better alternatives to this popular practice. For example, future studies should examine the potential C benefit by using a longer rotation with even-aged management or by implementing uneven-aged management.

Develop climate-smart commodities

How we use the harvested timber can significantly impact the potential of planted southern pine forests as a C sink. For example, the use of mass timber in building construction can achieve both C substitute and long-term C storage. Mass timber can substitute concrete and steel, reducing C emissions from fossil fuels. The C contained in the mass timber can also be stored in the built environment for the long term when compared with other timber uses. Similarly, easily decomposable woody biomass in forests is a major C source, and its conversion to biofuel and biochar could significantly enhance forests' capacity as a natural climate solution. In addition to significantly reducing forest decomposition, biofuel is a source of renewable energy and can substitute fossil fuel, which reduces C emissions. The application of biochar can improve soil properties and potentially reduce the use of fertilizers, another way of C substitution that reduces C emission. Biochar C can also be stored long term in soil, increasing soil C storage.

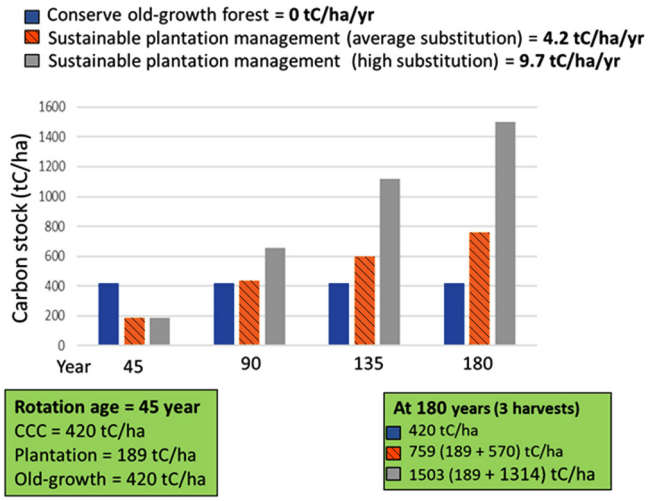

Figure 7—The impact of management and wood-use alternatives on total carbon mitigation. Carbon storage through timber utilization and substitution (i.e., *ex-situ* storage) greatly increases the capacity of forest carbon sinks. This figure is reconstructed from the data of table 1 of Lippke and others (2011), assuming the old-growth forest has a carbon stock at the carrying capacity (CCC) of 420 tC/ha, and the sustainably managed plantation has a carbon stock of 189 tC/ha at the rotation age of 45. The three scenarios are 1) conserving the old-growth forest, 2) sustainably managing the plantation with average substitution, and 3) sustainably managing the plantation with high substitution. At 180 years after three harvests, *in-situ* carbon stock is 420, 189, and 189 tC/ha, *ex-situ* carbon stock is 0, 570, and 1314 tC/ha, and carbon sequestration rate is 0, 4.2, 9.7 tC/ha/year.

Lippke and others (2011) illustrated the potential of *ex-situ* C storage by comparing an old-growth forest and a managed 45-year rotation planted forest established on similar climate and site conditions (fig. 7). The old-growth forest had already reached its CCC (the maximum C stock) at 420 tC/ha, so it would no longer sequester any more C. The sustainably managed planted forest had a much lower *in-situ* C storage at the rotation age of 45, which was 189 tC/ha. However, the plantation was harvested once every 45 years, and the *ex-situ* C storage of the planted forest is realized due to substitution and long-term storage after each timber harvest. As a result, compared to land consistently maintained as old growth, the land used for timber management (i.e., the sustainably managed planted forest) would have created an additional C sequestration at 4.2 (average substitution) and 9.7 (high substitution) tC/ha/year for the next 135 years (after three harvests) at the time when the fourth harvest is conducted (fig. 7).

CONCLUSIONS

CSF is a new forestry paradigm that is motivated by our desire to use forests as a natural climate solution and empowered by innovative technologies such as AI, intelligent data acquisition systems, and FDT.

CSF emphasizes increasing forest C sequestration to slow down future climate change (i.e., climate

change mitigation) but must do so to ensure forest resilience through climate change adaptation to sustain all ecosystem services.

Because land availability limits the increase of forest areas and the inherent CCC and natural disturbances limit the *in-situ* forest C storage in existent forests, *ex-situ* C storage must be encouraged through timber production, utilization, and substitution.

CSF will change how we practice silviculture in the Southeastern United States. With some modification and development, planted loblolly pine forest management is well positioned to help achieve CSF.

ACKNOWLEDGMENTS

This paper has benefitted from discussions with many colleagues, especially Feng Luo (Clemson University), Vijay Kumar (University of Pennsylvania), Mark Ashton (Yale University), and Ling Liu (Georgia Institute of Technology), while preparing a team proposal of "AI Institute for Climate-Smart Forestry," and Jiaojun Zhu (Institute of Applied Ecology, Chinese Academy of Sciences). Lauren Pile Knapp, Deliang Lu, and Brian Oswald have reviewed the draft and provided helpful suggestions for improvement.

REFERENCES

Birdsey, R.; Pregitzer, K.; Lucier, A. 2006. Forest carbon management in the United States: 1600–2100. Journal of Environmental Quality. 35(4): 1461–1469. https://doi.org/10.2134/jeq2005.0162.

Bowditch, E.; Santopuoli, G.; Binder, F. [and others]. 2020. What is climate-smart forestry? A definition from a multinational collaborative process focused on mountain regions of Europe. Ecosystem Services. 43: 101113. 13 p. https://doi.org/10.1016/j.ecoser.2020.101113.

Buonocore, L.; Yates, J.; Valentini, R. 2022. A proposal for a forest digital twin framework and its perspectives. Forests. 13(4): 498. 19 p. https://doi.org/10.3390/f13040498.

Chen, S.; Lu, N.; Fu, B. [and others] 2022. Current and future carbon stocks of natural forests in China. Forest Ecology and Management. 511: 120137. 11 p. https://doi.org/10.1016/j.foreco.2022.120137.

Colson, E. 2019. What AI-driven decision making looks like. Harvard Business Review: 1–9.

Forzieri, G.; Dakos, V.; McDowell, N.G. [and others]. 2022. Emerging signals of declining forest resilience under climate change. Nature. 608(7923): 534–539. https://doi.org/10.1038/s41586-022-04959-9.

Fox, T.R.; Jokela, E.J.; Allen, H.L. 2007. The development of pine plantation silviculture in the Southern United States. Journal of Forestry. 105(7): 337–347. https://doi.org/10.1093/jof/105.7.337.

Grassi, G.; House, J.; Dentener, F. [and others]. 2017. The key role of forests in meeting climate targets requires science for credible mitigation. Nature Climate Change. 7: 220–226. https://doi.org/10.1038/nclimate3227.

Griscom, B.W.; Adams, J.; Ellis, P.W. [and others]. 2017. Natural climate solutions. Proceedings of the National Academy of Sciences. 114: 11645–11650. https://doi.org/10.1073/pnas.1710465114.

Gupta, R.K.; Rao, D.L.N. 1994. Potential of wastelands for sequestering carbon by reforestation. Current Science. 66: 378–380. https://www.currentscience.ac.in/Volumes/66/05/0378.pdf.

Hoffmann, W.A.; Geiger, E.L.; Gotsch, S.G. [and others]. 2012. Ecological thresholds at the savanna-forest boundary: how plant traits, resources and fire govern the distribution of tropical biomes. Ecology Letters. 15(7): 759–768. https://doi.org/10.1111/j.1461-0248.2012.01789.x.

Keenan, T.F.; Williams, C.A. 2018. The terrestrial carbon sink. Annual Review of Environment and Resources. 43: 219–243. https://doi.org/10.1146/annurev-environ-102017-030204.

Keith, H.; Mackey, B.; Berry, S. [and others]. 2010. Estimating carbon carrying capacity in natural forest ecosystems across heterogeneous landscapes: addressing sources of error. Global Change Biology. 16(11): 2971–2989. https://doi.org/10.1111/j.1365-2486.2009.02146.x.

Kinerson, R.S.; Ralston, C.W.; Wells, C.G. 1977. Carbon cycling in a loblolly pine plantation. Oecologia. 29: 1–10. https://doi.org/10.1007/BF00345358.

Lippke, B.; Oneil, E.; Harrison, R. [and others]. 2011. Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. Carbon Management. 2(3): 303–333. https://doi.org/10.4155/CMT.11.24.

Liu, Y.C.; Yu, G.R.; Wang, Q.F. [and others]. 2014. Carbon carry capacity and carbon sequestration potential in China based on an integrated analysis of mature forest biomass. Science China Life Sciences. 57: 1218–1229. https://doi.org/10.1007/s11427-014-4776-1.

Messier, C.; Bauhus, J.; Doyon, F. [and others]. 2019. The functional complex network approach to foster forest resilience to global changes. Forest Ecosystems. 6(1): 21. 16 p. https://doi.org/10.1186/s40663-019-0166-2.

Millar, C.I.; Stephenson, N.L. 2015. Temperate forest health in an era of emerging megadisturbance. Science. 349(6250): 823–826. https://doi.org/10.1126/science.aaa9933.

Mitchell, C.D.; Harper, R.J.; Keenan, R.J. 2012. Current status and future prospects for carbon forestry in Australia. Australian Forestry. 75(3): 200212. https://doi.org/10.1080/00049158.2012.10676402.

Nagel, L.M.; Palik, B.J.; Battaglia, M.A. [and others]. 2017. Adaptive silviculture for climate change: a national experiment in manager-scientist partnerships to apply an adaptation framework. Journal of Forestry. 115: 167–178. https://doi.org/10.5849/jof.16-039.

Oswalt, S.N.; Smith, W.B.; Miles, P.D.; Pugh, S.A., coords. 2019. Forest resources of the United States, 2017: a technical document supporting the Forest Service 2020 RPA assessment. Gen. Tech. Rep. WO-97. Washington, DC: U.S. Department of Agriculture, Forest Service. 223 p. https://doi.org/10.2737/WO-GTR-97.

Roosevelt, T. 1903. Remarks at a meeting of the Society of American Foresters at the residence of Mr. Gifford Pinchot. The American Presidency Project. https://www.presidency.ucsb.edu/node/343450 [Date accessed: October 25, 2023].

Shephard, N.T.; Narine, L.; Peng, Y.; Maggard, A. 2022. Climate Smart Forestry in the Southern United States. Forests. 13(9): 1460. 20 p. https://doi.org/10.3390/f13091460.

Verkerk, P.J.; Costanza, R.; Hetemäki, L. [and others]. 2020. Climate-Smart Forestry: the missing link. Forest Policy and Economics. 115: 102164. 4 p. https://doi.org/10.1016/j.forpol.2020.102164.

Weatherall, A.; Nabuurs, G.J.; Velikova, V. [and others]. 2022. Defining Climate-Smart Forestry. In: Tognetti, R.; Smith, M.; Panzacchi, P., eds. Climate-Smart Forestry in mountain regions. Managing Forest Ecosystems 40. Cham, Switzerland: Springer: 35-58. https://doi.org/10.1007/978-3-030-80767-2_1.

Zhao, G.; Shao, G.; Reynolds, K.M. [and others]. 2005. Digital forestry: a white paper. Journal of Forestry. 103(1): 47-50. https://doi.org/10.1093/jof/103.1.47.

Intermediate Treatments/ Competition Control

AMINOCYCLOPYRACHLOR CUT STEM APPLICATIONS COMPARED TO STANDARD HERBICIDES FOR EUCALYPTUS CONTROL

Patrick J. Minogue and Kimberly A. Lorentz

ABSTRACT

Eucalyptus (*Eucalyptus* spp.) plantations are grown on more than 44 million acres worldwide, yet some species are considered invasive. A new herbicide, aminocyclopyrachlor (AMCP) was evaluated to determine rate response and effect of stem diameter for control of planted eucalyptus trees by cut stem treatment compared to standard applications of imazapyr or triclopyr herbicides. Using an AMCP product containing 1.0-pound acid equivalent (ae) per gallon, the lowest rate tested, 12.5 percent in water, resulted in 100 percent eucalyptus crown reduction and greater control than the standard aqueous preparations of 8 percent imazapyr product containing 2.0-pound ae per gallon or 50 percent triclopyr amine product containing 3.0-pounds ae per gallon at 6 and 12 months after treatment. Increases in stem height and diameter at 24 months after treatment suggest that control obtained using imazapyr and triclopyr cut stem treatments may be impermanent. AMCP treated trees did not have regrowth by 24 months after treatment.

INTRODUCTION

Members of the diverse *Eucalyptus* genus are native to Australia and the bordering islands of Polynesia (González-Orozco and others 2014) and more than 70 species are naturalized elsewhere (Rejmánek and Richardson 2011). Because of their adaptability and fast growth rates, *Eucalyptus* species and hybrids have been grown for fiber, fuel, landscaping mulch, essential oils, phytoremediation, and as an ornamental (Rockwood 2012) on more than 44 million acres worldwide (Pires and others 2013). In the Southeastern United States, eucalyptus has historically been used for pulp, mulch, and windbreaks. There is interest in planting cold tolerant species, hybrids, and genetically modified stock to supply potential bioenergy markets. However, concerns have been raised over the potential for eucalyptus to become invasive in the Southeastern United States (Callaham and others 2013, Gordon and others 2011, Lorentz and Minogue 2015).

Existing recommendations for chemical control of naturalized eucalyptus generally entail cut stump, cut stem (hack and squirt), or basal bark applications using concentrated herbicide solutions or emulsions containing the active ingredients triclopyr, imazapyr, or glyphosate. Individual stem treatments using these herbicides are preferred for their ease of use and targeted application. Triclopyr and glyphosate may be applied to the stems of target vegetation with minimal impact to nearby vegetation because these herbicides have limited soil activity (Senseman 2007). However, imazapyr is a persistent soil-active herbicide, thus injury to nontarget vegetation is more likely.

Cut stump applications of these herbicides are also used in eucalyptus plantations for silviculture objectives such as managing tree disease and replacing rootstocks with genetically improved cultivars, with harvesting cycles at stand age 5 to 10 years. Most *Eucalyptus* species planted in commercial plantations resprout vigorously following felling and may be coppiced. Cold-hardy *Eucalyptus* species such as *E. amplifolia*, *E. benthamii*, and *E. macarthurii*, widely planted in the Southeastern United States, southern Brazil, and South Africa (Minogue

Patrick J. Minogue, Associate Professor, North Florida Research and Education Center, University of Florida, Quincy, FL 32351
Kimberly A. Lorentz, Graduate Student, School of Forest, Fisheries, and Geomatics Science, University of Florida, Gainesville, FL 32611

and others 2018), are proving difficult to kill by existing cut stump methods (Little and van den Berg 2007).

Aminocyclopyrachlor (AMCP) is a relatively new herbicide developed by DuPont (Wilmington, DE) for use in noncrop areas such as rights-of-way, turf, range, and natural areas (E.I. du Pont de Nemours and Company 2009). AMCP has good selectivity to established cool- and warm-season perennial grasses (Westra and others 2009) and shows other positive stewardship attributes including low volatility, the absence of bioaccumulation in animals, and low toxicity to terrestrial and aquatic organisms (E.I. du Pont de Nemours and Company 2009). It has shown effectiveness at rates as low as 2-ounces acid equivalent (ae) per acre for difficult species to control such as kudzu (*Pueraria montana*) (Minogue and others 2011). However, its use in southern landscapes has severely damaged trees and other ornamentals (Wherley and others 2013).

In an early screening, Yeiser and others (2012) compared four rates from 2.5 to 15 percent AMCP (1-pound ae per gallon) to 30 percent triclopyr ester product (4-pounds ae per gallon) and combinations of these herbicides with 1-percent imazapyr (2-pounds ae per gallon) in basal oil for cut-stump treatments of Chinese tallow (*Triadica sebifera*), sweetgum (*Liquidambar styraciflua*), and yaupon (*Ilex vomitoria*). At 18 months after treatment (MAT) 10 percent AMCP, AMCP plus imazapyr, and triclopyr alone treatments provided 100 percent control of Chinese tallow. Whereas for sweetgum, the least sprouting and best control was obtained with triclopyr alone. For yaupon, a difficult species to control with herbicides, at 18 MAT 100 percent control was observed with triclopyr, triclopyr plus imazapyr, and AMCP plus imazapyr.

Eucalyptus benthamii is among the more promising eucalyptus species for widescale planting in the Southeastern United States because of its cold hardiness, success in outplantings under a variety of conditions, and fast growth rates (Zalesny and others 2011). This species is not among the eucalyptus species considered invasive (Rejmánek and Richardson 2013), but the ability of this species to withstand cold weather could contribute to its potential invasiveness at a geographic scale.

We initiated a study in a young *E. benthamii* plantation to evaluate the efficacy of AMCP for eucalyptus control. The objectives of this research were 1) to compare the effectiveness of four rates of AMCP to imazapyr and triclopyr amine operational standard herbicides for control of eucalyptus using cut stem applications, and 2) to determine the effects of herbicide treatments on nearby nontarget eucalyptus trees.

METHODS

Study Location

Our study was established in a 28-month *Eucalyptus benthamii* plantation at the University of Florida, North Florida Research and Education Center, south of Quincy (30.55° N, 84.60° W) at approximately 240 feet elevation. This location has a temperate climate with highest temperatures in July (mean 80.6 °F), lowest temperatures in January (mean 50 °F) and 56.3 inches mean annual precipitation (NOAA 2023). The prevalent soil series is Orangeburg fine sandy loam (fine-loamy, kaolinitic, thermic Typic Kandiudults) (USDA NRCS 2023).

Treatments and Experimental Design

Healthy trees with suitable diameters were ranked by basal stem diameter (b.s.d.) at 1 foot height and sequentially divided into eight groups of seven trees with like diameters. The seven treatments (table 1) were assigned in a randomized complete block design with eight replications. A hatchet was used to make cup-like downward incisions at b.s.d. One cut was made per inch of b.s.d. Cuts were evenly spaced around the stem circumference. A volumetric syringe was used to apply 0.03 ounces (1 ml) of aqueous herbicide mixture to each cut on November 5, 2011. Trees in the cut-only control were cut with a hatchet in the same manner as other treatments, but no herbicides were applied.

Table 1—Cut stem treatments for *Eucalyptus benthamii* control using aminocyclopyrachlor (AMCP) and standard herbicides in aqueous solution

Herbicide	Formulation tested	Formulation concentration	Applied formulation concentration in water	b.s.d. specific dose
		pounds ae per gallon	*percent*	*ounces ae 10^{-4}*
AMCP	DPX MAT28-159	1.0	12.5	5.3
AMCP	DPX MAT28-159	1.0	25.0	10.6
AMCP	DPX MAT28-159	1.0	50.0	21.2
AMCP	DPX MAT28-159	1.0	100.0	42.3
Imazapyr	Stalker®	2.0	7.8	6.7
Triclopyr amine	Garlon® 3A	3.0	50.0	63.5
Cut-only control	Cut-only	0.0	0.0	0.0

One cut was made per 1-inch basal stem diameter (b.s.d.) at 1-foot height with 0.03 ounces (1 ml) solution applied per cut.
b.s.d. specific dose is ounces 10^{-4} acid equivalent (ae) of active ingredient per inch of stem diameter at 1 foot above groundline.

AMCP (1-pound ae per gallon, DPX MAT28-159) was applied at four rates, 12.5, 25, 50, and 100 percent formulated product in water (vol:vol) to determine rate response. Standard cut stem treatments for eucalyptus control included aqueous preparations of 7.8 percent imazapyr (isopropylamine salt of imazapyr, 2-pounds ae per gallon, Stalker®, BASF, Research Park, NC) and 50 percent triclopyr amine (triethylamine salt of triclopyr, 3 pounds ae per gallon, Garlon® 3A, Dow AgroSciences, Indianapolis, IN) following label guidelines (BASF Corporation 2008, Dow AgroSciences LLC 2003). Herbicide treatments were compared to the cut-only method control.

Tree Assessments

Pretreatment b.s.d. (1 foot height) was measured to the nearest 0.0394 inch (1 mm) to determine b.s.d. and diameter at breast height (d.b.h. 4.5 feet from groundline), in October 2011. Pretreatment stem height was measured to the nearest inch using a height pole. To quantify stem dieback or growth, the live height was measured again 12 and 24 MAT. Additionally, at 2, 6, and 12 MAT, each tree was assessed for percent crown reduction. This standard variable (Miller and Glover 1991) is an ocular estimate of reduction relative to the pretreatment condition, and considers stem dieback, leaf necrosis, and defoliation. Estimates were made to the nearest 5 percent for values between 0 and 10 percent, and 90 and 100 percent; and to the nearest 10 percent for values between 10 and 90 percent. In place of crown dieback, live d.b.h. was measured as a more appropriate response variable at 24 MAT because some rootstocks recovered by that time. Additionally, phytotoxicity symptoms (mortality, foliar necrosis, defoliation, red foliage, chlorosis, stem sap flow, epinasty, basal sprouting, and adventitious buds on stems) were noted as present or not present for each treatment and buffer tree at 2, 6, and 12 MAT.

Herbicide Impacts to Nontarget Trees

To measure the potential for herbicide flashback via root grafting or injury through soil uptake by nonherbicide-treated trees, at various distances from treated trees, symptoms of phytotoxicity were assessed for both cut-only control trees (n = 56) and buffer trees (n = 64) in the stand. Buffer trees were those directly adjacent to herbicide-treated trees on the row (5 feet away), whereas cut-only control trees were randomly assigned to trees that were not adjacent to herbicide-treated trees (≥10 feet away). For each tree assessment date given above, crown reduction, live stem height, and live diameter were measured.

Statistical Analysis

Data for stem live height, diameter, and crown reduction were analyzed using SAS PROC GLM (SAS v. 9.3, SAS Institute Inc., Cary, NC) to determine differences among treatment groups

for change in stem live height at 12 and 24 MAT and for change in d.b.h. at 24 MAT. Tukey's HSD test was used to compare means at $\alpha = 0.05$. Crown reduction response values were primarily near the fixed limits of 0 and 100 percent. Homogeneity of variances and normality could not be achieved by standard data transformations (arcsine, arcsine square root, and log), so traditional parametric and nonparametric tests could not be used to determine if there were differences among treatment groups. Instead, Welch's analysis of variance on ranked data was performed using SAS PROC GLM to determine if there were differences between treatments. The Ryan-Einot-Gabriel-Welsch multiple range test was performed to compare treatment means at $\alpha = 0.05$. These procedures are recommended for analysis of data that is simultaneously heteroscedastic and nonnormal, and they have been shown to provide good power and acceptable control for type I error rates (Cribbie and others 2007). To assess herbicide impacts to nontarget trees data for buffer trees were compared to cut-only control trees using Fisher's exact test for absent versus present responses.

RESULTS

Crown Reduction

The treatment effect for crown reduction was highly significant (P <0.0001) and all herbicide treatments resulted in greater crown reduction than the cut-only control at each assessment (table 2). Crown reduction did not differ among the AMCP rates tested and provided 100 percent control by 6 MAT. AMCP resulted in greater crown reduction than imazapyr at each assessment with imazapyr providing 58-percent crown reduction at 12 MAT. Triclopyr amine resulted in 49-percent crown reduction at 2 MAT which was less than observed with AMCP treatments (91–100 percent) but not different from the results using imazapyr (19 percent). However, at the 6 and 12 MAT assessments crown reduction obtained with triclopyr amine (80–81 percent) did not differ from AMCP treatments (100 percent) but was greater than observed for imazapyr (49–58 percent).

Table 2—Crown reduction, stem live height change, and change in diameter at breast height (d.b.h.) of *Eucalyptus benthamii* from pretreatment to assessments at 2, 6, 12, and 24 months after treatment (MAT) with herbicide

		Crown reduction			Stem live height change		d.b.h. change
Herbicide	b.s.d. dose	2 MAT	6 MAT	12 MAT	12 MAT	24 MAT	24 MAT
	ounce 10^{-4}	~~~~~~~~~~ *percent* ~~~~~~~~~~			~~~~~~~*feet*~~~~~~~		*inch*
AMCP	5.3	91a	100a	100a	-26.6a	-26.6a	-2.9ab
AMCP	10.6	100a	100a	100a	-23.9a	-23.9a	-2.5ab
AMCP	21.2	99a	100a	100a	-26.9a	-26.9a	-3.0a
AMCP	42.3	100a	100a	100a	-25.6a	-25.6a	-2.7ab
Imazapyr	6.7	19bc	49b	58b	-9.4b	-6.7b	0.8c
Triclopyr amine	63.5	49b	80a	81a	-11.4b	-6.0b	-0.9bc
Cut-only control	0.0	3c	0c	2c	14.0c	31.4c	3.6d

b.s.d. specific dose, ounce 10^{-4} acid equivalent of active ingredient per inch of stem diameter at 1 foot above groundline.

Crown reduction and stem live height change treatment means in a column followed by the same letter were not significantly different according to the Ryan-Einot-Gabriel-Welsch multiple range test on ranked data at $\alpha = 0.05$.

Change in d.b.h. treatment means within a column followed by the same letter were not significantly different according to Tukey's HSD at $\alpha = 0.05$.

b.s.d. = basal stem diameter; d.b.h. = diameter at breast height; MAT = months after treatment; AMCP = aminocyclopyrachlor.

Stem Live Height

The treatment effect on stem live height change at 12 and 24 MAT was highly significant (P <0.0001). All herbicide treatments reduced stem height due to stem dieback and apparent mortality, whereas cut-only control trees grew 31.4 feet by 24 MAT (table 2). At 12 MAT all AMCP treatments had no live stem height, with the change from pretreatment stem height ranging from -23.9 to -26.6 feet, and there was no regrowth at 24 MAT. Imazapyr and triclopyr amine standards had less stem height change than AMCP treatments and did not differ. Stem dieback continued from 12 to 24 MAT for the standards with -6.7- and -6.0-foot height change for imazapyr and triclopyr amine, respectively.

Stem Live Diameter

The treatment effect on stem live d.b.h. change at 24 MAT was highly significant (P <0.0001) and all herbicide treatments reduced d.b.h. relative to the cut-only control (table 2). AMCP treatments had no live d.b.h. and the change from pretreatment ranged from -2.5 to -3.0 inches, as compared to 3.6 inches growth for the cut-only control. The imazapyr standard treatment grew 0.8 inches and differed from all treatments except the -0.9-inch change observed for the triclopyr standard treatment.

Herbicide Impacts to Nontarget Trees

There were no significant treatment differences in crown reduction at 12 MAT or for the change in stem live height and diameter at 24 MAT when buffer trees (5 feet away from herbicide treated trees) were compared to cut-only control trees (≥10 feet away from herbicide treated trees) (data not shown). By 24 MAT, buffer trees grew 25.9 feet taller and gained 2.9 inches in diameter. Similarly, cut-only control trees grew 25.2 feet taller and gained 2.7 inches in diameter. The most frequently observed phytotoxicity symptoms for both buffer and cut-only control trees were foliar necrosis, defoliation, chlorosis, and red foliage. Other symptoms observed included epinasty, a symptom characteristic of the growth regulator herbicides triclopyr and AMCP. In general, symptoms in buffer and cut-only control trees began to appear by 6 MAT, but remained uncommon throughout the study period. At 6 MAT the presence of foliar necrosis was greater (P = 0.0236) for adjacent buffer trees (28 percent of trees) than for cut-only control (13 percent of trees), but no differences were observed for any other symptoms or assessment (data not shown). At 6 MAT mortality was observed in one buffer tree bordered by AMCP treatments on both sides, but no mortality occurred in cut-only control trees.

DISCUSSION

Whereas no other literature reporting results for AMCP cut stem application for the control of woody plants was found, this application method, also known as stem injection or "hack and squirt," is described by many herbicide product labels and widely used because of the ease of use and low cost (Miller and others 2015). Our results demonstrate that AMCP cut stem treatment is an effective alternative to more labor-intensive cut stump applications, which have been the focus of several studies (Little and van den Berg 2007, Yeiser and others 2012). In addition, the determined cut stem AMCP b.s.d. dose (5.3 ounces 10^{-4}) for complete control demonstrated the greater method efficiency as compared to the basal spray AMCP b.s.d. dose (21.2 ounces 10^{-4}) for complete control of *Eucalyptus benthamii* (Minogue and Lorentz 2021).

Herbicide injury was observed in buffer and nontreated control trees as expected for AMCP and imazapyr, which are absorbed from the soil by roots. This may be of concern when making applications in very close proximity to desirable trees. These plantations were established at high density (871 trees per acre) for short-rotation bioenergy production, and the risk of herbicide injury to crop trees may be less for more typical wide spacings. However, the use of triclopyr amine, which has negligible soil activity, would provide greater safety to residual trees in thinning or cultivar replacement. While we did not quantify impacts to herbaceous vegetation in this study, no obvious symptoms of injury such as necrosis or chlorosis were observed for herbaceous vegetation near treated stems. Moreover, during the study several

newly established pine and oak seedlings were discovered throughout the *E. benthamii* understory.

CONCLUSIONS

While there have been serious concerns about the effects of AMCP on nontarget trees and vegetation (Durkin 2012), this may not be a concern for stand eradication of nuisance or invasive eucalyptus species. Our results suggest that the risk of injury to nontarget vegetation may be reduced when low rate, low volume directed applications are used away from desired trees. Based on this research, cut stem applications at 1 foot above groundline using 0.03 ounces (1 ml) of 12.5-percent AMCP solution (diluted 1-pound ae AMCP per gallon product) per inch of basal diameter provides complete control, greater than observed for standard treatments. This rate equates to 5.3 ounces 10^{-4} ae AMCP per inch stem diameter, and there is a clear need to examine lower doses for eucalyptus control and other tree species.

Although we tested AMCP rate response in comparison to the efficacy of standard herbicides, the potentially important effects of application timing and eucalyptus species susceptibility were not determined. This would require experiments across multiple seasons for commonly grown *Eucalyptus* species, such as *E. camaldulensis*, *E. grandis*, *E. urograndis*, *E. amplifolia*, and others. The strong control of *E. benthamii* in this study represents significant progress in the development of effective herbicide recommendations for eucalyptus management and indicates that future research with different species and timings would be worthwhile.

REFERENCES

BASF Corporation. 2008. Stalker® herbicide product label. Research Triangle Park, NC. 9 p.

Callaham, M.A., Jr.; Stanturf, J.A.; Hammond, W.J. [and others]. 2013. Survey to evaluate escape of *Eucalyptus* spp. seedlings from plantations in southeastern USA. International Journal of Forestry Research. 2013: 946374. 10 p. http://dx.doi.org/10.1155/2013/946374.

Cribbie, R.A.; Wilcox, R.R.; Bewell, C.; Keselman, H.J. 2007. Tests for treatment group equality when data are nonnormal and heteroscedastic. Journal of Modern Applied Statistical Methods. 6: 117–132. https://doi.org/10.22237/jmasm/1177992660.

Dow AgroSciences LLC. 2003. Garlon® 3A herbicide product label. LOES No. 010-00084. Indianapolis, IN. 9 p.

Durkin, P.R. 2012. Aminocyclopyrachlor: Human health and ecological risk assessment. Final report. TR-056-01-03a. Manlius, NY: Syracuse Environmental Research Associates, Inc. 212 p. https://www.fs.usda.gov/foresthealth/pesticide/pdfs/Aminocyclopyrachlor.pdf.

E.I. du Pont de Nemours and Company. 2009. DuPont DPX-MAT28 herbicide. Technical Bulletin No. K15023. Wilmington, DE. 7 p.

González-Orozco, C.E.; Thornhill, A.H.; Knerr, N. [and others]. 2014. Biogeographical regions and phytogeography of the eucalypts. Diversity and Distributions. 20: 46–58. https://doi.org/10.1111/ddi.12129.

Gordon, D.R.; Tancig, K.J.; Onderdonk, D.A.; Gantz, C.A. 2011. Assessing the invasive potential of biofuel species proposed for Florida and the United States using the Australian Weed Risk Assessment. Biomass and Bioenergy. 35: 74–79. https://doi.org/10.1016/j.biombioe.2010.08.029.

Little, K.M.; van den Berg, G.J. 2007. Comparison of different herbicides for single stem *Eucalyptus marcarthurii* cut stump control. Journal of Tropical Forest Science. 19(1): 13–17.

Lorentz, K.A.; Minogue, P.J. 2015. Exotic *Eucalyptus* plantations in the southeastern US: risk assessment, management and policy approaches. Biological Invasions. 17: 1581–1593. https://doi.org/10.1007/s10530-015-0844-0.

Miller, J.H.; Glover, G.R. eds. 1991. Standard methods for forest herbicide research. Champaign, IL: Southern Weed Science Society. 68 p.

Miller, J.H.; Manning, S.T.; Enloe, S.F. 2015. Revised. A management guide for invasive plants in southern forests. Gen. Tech. Rep. SRS-131. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 120 p. https://doi.org/10.2737/SRS-GTR-131.

Minogue, P.J.; Enloe, S.F.; Osiecka, A.; Lauer, D.K. 2011. Comparison of aminocyclopyrachlor to common herbicides for kudzu (*Pueraria montana*) management. Invasive Plant Science and Management. 4: 419–426. https://doi.org/10.1614/IPSM-D-11-00024.1.

Minogue, P.J.; Lorentz, K.A. 2021. Comparison of aminocyclopyrachlor to standard herbicides for basal stem treatment of *Eucalyptus benthamii*. Weed Technology. 35: 304–308. https://doi.org/10.1017/wet.2020.111.

Minogue, P.J.; Osiecka, A.; Lauer, D.K. 2018. Selective herbicides for establishment of *Eucalyptus benthamii* plantations. New Forests. 49:5 29–550. https://doi.org/10.1007/s11056-018-9637-5.

National Oceanic and Atmospheric Administration [NOAA]. 2023. Monthly station normals of temperature, precipitation, and heating and cooling degree days 1981 to 2010. In: Climatography of the United States. No. 81.08–Florida. Asheville, NC: NOAA National Climatic Data Center and National Environmental Satellite, Data, and Information Service.

Pires, R.N.; Pereira, F.C.M.; Nepomuceno, M.P.; Alves, P.L.C.A. 2013. Effects of the simulated drift of ripeners on *Eucalyptus urograndis*. Journal of Agricultural Science. 5(12): 78–86. http://dx.doi.org/10.5539/jas.v5n12p78.

Rejmánek, M.; Richardson, D.M. 2011. Eucalypts. In: Simberloff, D.; Rejmánek, M., eds. Encyclopedia of biological invasions. Berkeley, CA: University of California Press: 203–209.

Rejmánek, M.; Richardson, D.M. 2013. Trees and shrubs as invasive alien species – 2013 update of the global database. Diversity and Distributions. 19: 1093–1094. https://doi.org/0.1111/ddi.12075.

Rockwood, D.L. 2012. History and status of *Eucalyptus* improvement in Florida. International Journal of Forestry Research. 2012: 607879. 10 p. http://dx.doi.org/10.1155/2012/607879.

Senseman, S.A., ed. 2007. Herbicide handbook. 9th ed. Lawrence, KS: Weed Science Society of America. 458 p.

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2023. Web soil survey. https://websoilsurvey.sc.egov.usda.gov. [Date accessed: September 5, 2023].

Westra, P.; Nissen, S.; Shaner, D. [and others]. 2009. Invasive weed management with aminocyclopyrachlor in the Central Great Plains. Abstract of presentation at the 49th annual meeting of the Weed Science Society of America. https://wssa.net/meeting/meeting-abstracts/.

Wherley, B.; Pannkuk, T.; Cabrera, R.I.; Campos, C. 2013. Off-target injury to southern landscape species following aminocyclopyrachlor applications. Journal of Environmental Horticulture. 31(3): 189–193. https://doi.org/10.24266/0738-2898.31.3.189.

Yeiser, J.L.; Link, M.; Grogan, J. 2012. Screening cut-stump control of Chinese tallowtree, sweetgum, and yaupon with aminocyclopyrachlor. e-Gen. Tech. Rep. SRS-156. In: Butnor, J.R., ed. Proceedings of the 16th biennial southern silvicultural research conference. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 389–393.

Zalesny, R.S., Jr.; Cunningham, M.W.; Hall, R.B. [and others]. 2011. Woody biomass from short rotation energy crops. In: Zhu, J., ed. Sustainable production of fuels, chemicals, and fibers from forest biomass. ACS Symposium Series Vol. 1067. Washington, DC: American Chemical Society: 27–63. https://doi.org/10.1021/bk-2011-1067.ch002.

A COMPARISON OF GARLON® 4 ULTRA, GARLON® XRT, VISTA® XRT, VASTLAN®, CHOPPER GEN2®, FREELEXX® OUST® EXTRA, AND TERRAVUE® IN SITE PREPARATION APPLICATIONS

A. Brady Self, J. Eric Ezell, and Andrew W. Ezell

ABSTRACT

Site preparation continues to be a major focus of vegetation management programs in pine (*Pinus* spp.) plantation management in the South. Evaluation of new products applied alone or in mixture must be completed to test cost efficacy. In this study, 10 treatments (including an untreated control) were applied to a recent cutover area in northern Mississippi. All treatments were replicated three times in an October 2021 application. Plots were evaluated at 4 months after treatment (MAT), 8 MAT, and 1 year after treatment. Results indicate that most tested mixtures provided excellent control of hardwoods, except hickory (*Carya* spp.). Greatest efficacy was observed in treatment plots containing higher rates of Chopper GEN2®.

INTRODUCTION

Chemical site preparation is a critical step in establishment of southern pine (*Pinus* spp.) plantations. While cost efficacy is always a consideration, in recent history, class action lawsuit settlements over perceived glyphosate concerns have resulted in a push to use different herbicide compounds in forest management efforts. Chemical site preparation is the largest end use for herbicides in forestry. Consequently, there is strong interest in using new herbicide formulations as alternatives to glyphosate in this role.

Different competition scenarios require different mixtures for optimal results. New herbicide formulations offer the chance to evaluate new possibilities for site preparation tank mixtures as land managers attempt to develop alternative cost-efficient treatment alternatives. The objective of this study was to evaluate efficacy of different herbicide mixtures for control of hardwoods in site preparation applications.

METHODS

The study was installed on the Mississippi State University, John Starr Memorial Forest near Starkville, MS. The site was harvested 1 year prior to study installation. Soil series was Falkner silt loam with an average pH of 5.3. Hickory (*Carya* spp.) was the dominant woody genus onsite with oak (*Quercus* spp.), red maple (*Acer rubrum*), and common persimmon (*Diospyros virginiana*) in sufficient quantities for testing. The stems ranged between 6 inches and 5 feet in height. Other woody and herbaceous species made up a minor vegetative component.

A complete list of treatments is found in table 1. Plots were rectangular areas measuring 10 feet by 100 feet, marked at each end by vinyl pin flags with string stretched along the plot midline between pin flags. Treatments were applied using a CO_2-powered backpack sprayer with pole extension and a KLC-9 spray tip to simulate aerial spraying. Total spray volume was 10 gallons per acre and all treatments were applied October 7, 2021.

A. Brady Self, Associate Extension Professor, Mississippi State University, Department of Forestry, Grenada, MS 38901
J. Eric Ezell, Assistant Clinical Professor, Mississippi State University, Department of Geosciences, Starkville, MS 39762
Andrew W. Ezell, Professor and Department Head (Emeritus), Mississippi State University, Department of Forestry, Starkville, MS 39762

Table 1—List of treatments applied in 2021 Corteva site preparation study

Treatment	Herbicide (rate per acre)
1	Garlon® 4 Ultra (32 ounces) + TerraVue® (2.85 ounces) + Chopper GEN2® (32 ounces)
2	Vastlan® (32 ounces) + TerraVue® (2.85 ounces) + Chopper GEN2® (32 ounces)
3	Garlon® XRT (60 ounces) + TerraVue® 2.85 ounces) + Chopper GEN2® (16 ounces)
4	Garlon® XRT (21 ounces) + TerraVue® (5.7 ounces) + Chopper GEN2® (16 ounces)
5	Vista® XRT (21 ounces) + TerraVue® (2.85 ounces) + Chopper GEN2® (32 ounces)
6	Vista® XRT (21 ounces) + Vastlan® (32 ounces) + Chopper GEN2® (32 ounces)
7	Freelexx® (128 ounces) + TerraVue® (2.85 ounces) + Chopper GEN2® (32 ounces)
8	Oust® Extra (4 ounces)
9	Untreated check
10	Garlon® XRT (60 ounces) + Chopper GEN2® (32 ounces) + Oust® Extra (4 ounces)

Chopper GEN2® (BASF, Florham Park, NJ) is imazapyr.
Garlon® 4 Ultra and Garlon® XRT (Corteva Agriscience, Riverside, CA) are triclopyr ester.
TerraVue® (Corteva Agriscience™, Riverside, CA) is aminopyralid + florpyrauxifen-benzyl.
Vastlan® (Corteva Agriscience™, Riverside, CA) is triclopyr choline.
Vista® XRT (Corteva Agriscience™, Riverside, CA) is fluroxypyr.
Freelexx® (Corteva Agriscience™, Riverside, CA) is 2,4-D choline.
Oust® Extra (Bayer CropScience, St Louis, MO) is sulfometuron methyl + metsulfuron methyl.

Vegetation assessments were completed on a sample area 10 feet by 80 feet centered in the treatment plot. Prior to treatment, hardwood seedlings were identified to species and recorded by 1-foot height classes. Control was evaluated at 4 months after treatment, 8 months after treatment, and finally, 1 year after treatment.

Treatments were replicated three times in a randomized complete block design. Cumulative heights were calculated by multiplying the number of stems by respective height classes. Percent brownout averages were calculated and are presented herein. Percent values were arcsine transformed, but for ease of interpretation, actual values are presented. Means separation was performed using Duncan's New Multiple Range Test at $\alpha = 0.05$.

RESULTS

Average percentages of control 1 year after treatment are presented in table 2. Land managers expect to obtain ≥90.0 percent control of hardwood stems when employing chemical site preparation and none of the treatments tested provided this level of control on hickory. Excepting treatments 8 and 9 (35.7 and 37.2 percentage of control, respectively), control of oak was good ranging from 85.7 to 100.0 percent. It is noteworthy that plots receiving treatments 3 and 4 Chopper GEN2® at 16 ounces compared to 32 ounces in treatments 1, 2, 5 through 7, and 10) exhibited less control. This is not unexpected and correlates to the industry standard of 40–48 ounces of Chopper GEN2® in site preparation tank mixes.

With the exception of treatments 8 and 9, all treatments provided 100.0 percent control of red maple. Similar results were observed for common persimmon and subarborescent species with control of 92.3 percent observed in persimmon and ≥81.0 percent control noted in subarborescents. Adequate control was not expected from treatments 8 or 9 for any of the species tested in this study.

CONCLUSIONS

Chemical site preparation has a decades-long standing of use in pine plantation establishment and will continue to be utilized in most of these efforts in the South for the foreseeable future. As new product development offers different herbicide formulations, it is important to learn

Table 2—Percentage of control averages of hardwood groups in 2021 Corteva site preparation study at 1 year after treatment

Treatment	Hickory	Oak	Red maple	Persimmon	Subarborescents	Total
1	41.5c	100.0a	100.0a	100.0	97.1a	77.9b
2	39.2c	92.3a	100.0a	100.0	91.2a	77.4b
3	62.2b	87.2ab	100.0a	92.3	91.1a	78.0b
4	63.0b	85.7b	100.0a	93.8	90.0a	84.4ab
5	40.9c	100.0a	100.0a	100.0	93.1a	90.8a
6	81.8a	95.8a	100.0a	100.0	95.3a	95.2a
7	NA	NA	100.0a	NA	81.0b	83.9ab
8	48.3c	35.7c	18.9c	NA	45.7c	38.5c
9	0.0d	37.2c	71.7b	NA	+22.3d	26.6c
10	83.0a	100.0a	100.0a	100.0	99.2a	91.1a

Values in a column followed by the same letter do not differ at α = 0.05.
NA = insufficient stems for analysis.
+ indicates an increase in stems.

how site preparation efficacy can vary depending on what species are targeted for control. Though several of the mixtures in this study provided excellent control of most hardwood species, overall hardwood control was strongly influenced by the lack of hickory control. Other herbicides should be considered for sites where hickory is of concern for site preparation work. Finally, if Chopper GEN2® is used, control of oak will be substantially greater with rates exceeding 16 ounces per acre.

A COMPARISON OF GLUFOSINATE PRODUCTS FOR THE CONTROL OF NATURAL PINES

John Ezell, Andrew Self, and Andrew Ezell

ABSTRACT

A total of 15 treatments were applied to wilding loblolly pines (*Pinus taeda*) on two utility right-of-way sites. One site was sprayed in July and the second site was sprayed in September. Five different glufosinate products were included in the treatments. Total spray volume of 15 gallons per acre and three replications of all treatments were utilized at both sites. Brownout was evaluated at 14, 28, and 56 days after treatment. Pine control was evaluated at 1 year after treatment. Pine control ranged from 98.4 to 100.0 percent among the treatments. There was no significant difference among the glufosinate products for pine control at either site. Glufosinate exhibits excellent potential for the control of natural pines in forestry site preparation applications.

INTRODUCTION

Site preparation continues to be the largest use of herbicides in forestry applications in the Southeastern United States. One of the major challenges on many sites is the control of natural pines. These pines can range from 6 inches to 6 feet in height and may be encountered in many thousands of stems per acre. Concerns over the use of glyphosate have resulted in many land managers seeking a substitute material that would be effective in controlling natural pines. Glufosinate is a chemical that has shown promise for such applications.

METHODS

The study site for the July applications was an electric utility right-of-way in Oktibbeha County, MS. The site was well stocked with natural loblolly pines 1–6 feet tall with the majority being 1–3 feet tall. The average number of pines on this site was 9,300 per acre. The study site for the September applications was also an electric utility right-of-way in Oktibbeha County, MS. The site had dense pine coverage of seedlings in the 1-foot height class with the average number of pines being 230,490 per acre.

A total of 45 plots were utilized at each site. Each plot was a rectangular area 30 feet by 50 feet. The sample area for the July treatment plots was a swath 10 feet by 50 feet centered in the treatment plot. Pine sampling for the September plots was completed using three circular subplots located along the center line of the treatment plot. Each subplot was marked with a pin flag at the center and had a 1-foot radius.

A total of 15 treatments were utilized in the study (table 1). The same treatments were applied on both sites and included five different glufosinate products, one imazapyr product, one glyphosate product, and three products being tested to evaluate the potential to enhance the performance of the glufosinate.

The July treatments were applied July 28, 2021, using a CO_2-powered backpack sprayer equipped with a pole extension and KLC-9 nozzle. This equipment simulates aerial application. September treatments were applied September 24, 2021, using the same equipment. Total spray volume was 15 gallons per acre for all applications and all treatments were replicated three times.

All pines in the sample areas were recorded by height class prior to treatment application and

John Ezell, Assistant Clinical Professor, Mississippi State University, Starkville, MS 39762
Andrew Self, Associate Extension Professor, Mississippi State University, Grenada, MS 38901
Andrew Ezell, Professor and Department Head (Emeritus), Mississippi State University, Starkville, MS 39762

Table 1—List of treatments and percentage reduction in the number of loblolly pine seedlings at 1 year after treatment by application timing

Treatment	Herbicide (rate per acre)	July	September
1	Untreated	+64.3d	20.6b
2	Chopper GEN2® (48 ounces)	1.1c	28.8b
3	Finale® VU (150 ounces)	100.0a	100.0a
4	Finale® XL (64 ounces)	100.0a	100.0a
5	Numbered Glufosinate Cpd. (64 ounces)	100.0a	100.0a
6	Cheetah® Pro (64 ounces)	100.0a	100.0a
7	Interline® (64 ounces)	100.0a	100.0a
8	Accord® XRT II (64 ounces)	83.5b	100.0a
9	Finale® XL (64 ounces) + Detail® (2 ounces)	100.0a	100.0a
10	Finale® XL (64 ounces) + Numbered Enhancer Cpd. (1.83 ounces)	98.9a	100.0a
11	Finale® XL (64 ounces) + Ammonium Sulfate (3 pounds)	100.0a	100.0a
12	Finale® XL (64 ounces) + Chopper GEN2® (48 ounces)	98.4a	100.0a
13	Numbered Glufosinate Cpd. (64 ounces) + Chopper GEN2® (48 ounces)	100.0a	100.0a
14	Cheetah® Pro (64 ounces) + Chopper GEN2® (48 ounces)	98.9a	100.0a
15	Interline® (64 ounces) + Chopper GEN2® (48 ounces)	100.0a	100.0a

All treatments had Velocity (2 ounces per acre) and Elite Supreme (0.25 percent volume to volume) added.
Values in a column followed by the same letter do not differ significantly at $\alpha = 0.05$.
+ = an increase in stems.
Chopper GEN2® (BASF, Florham Park, NJ) is imazapyr.
Finale® VU (BASF, Florham Park, NJ) is glufosinate-ammonium.
Finale® XL (BASF, Florham Park, NJ) is glufosinate-ammonium.
Cheetah® Pro (Nufarm, Laverton North, Victoria, Australia) is glufosinate-ammonium.
Interline® (United Phosphorus, Inc., King of Prussia, PA) is glufosinate-ammonium.
Accord® XRT II (Corteva Agriscience, Indianapolis, IN) is glyphosate.
Detail® (BASF, Florham Park, NJ) is saflufenacil.

1 year after treatment (1 YAT). Brownout was recorded at 14, 28, and 56 days after treatment (DAT). Pine control was based on the reduction in the number of stems based on a comparison of the pretreatment and 1 YAT data.

RESULTS

July Treatments

The July application date represents a potential extension of the site preparation window in the Southeastern United States, which is generally considered to be August through December, depending on location. Pine response to the glufosinate treatments was rapid with significant brownout noted at 7 DAT. The brownout was thorough and consistent at the 14, 28, and 56 DAT evaluations. Very little or no control of pines was expected in the untreated or Chopper GEN2®-only treatments (treatments 1 and 2) and such a response was observed (table 1). There was no significant difference in pine control among the different glufosinate products (treatments 3 through 7), all of which provided 100 percent control. With control this high, it is understandable that the three products being tested for enhancing the glufosinate performance did not improve control (treatments 9 through 11). The very slight reduction in pine control in treatments of mixtures of glufosinate and imazapyr is attributed to the occasional shading of a small pine by larger vegetation. A significant difference in pine control was observed in the glyphosate treatment (treatment 8), but it should be noted that the rate per acre was only 64 ounces. Glyphosate is typically applied at 128 ounces per acre (or more) in site preparation work.

September Treatments

Again, the untreated and Chopper GEN2®-only treatments had much less control and the reduction in pine seedlings in these plots was attributed to natural mortality, which is expected in dense populations of small seedlings (table 1). There was no difference in pine control among treatments using different glufosinate products. All provided 100 percent control applied alone or in mixtures with other products. The glyphosate treatment (treatment 8) also provided 100 percent control of these small seedlings, which were fully exposed.

CONCLUSIONS

Glufosinate demonstrated excellent potential for control of natural pine seedlings when applied at the rates utilized in this study. All glufosinate products tested in this work performed equally well and no difference in control was observed when comparing the two application timings. While these results are encouraging, additional testing is needed to fully evaluate the efficacy of glufosinate for control of hardwood species.

SEASONAL LIGHT RESPONSE CURVES OF *LIGUSTRUM SINENSE*: A TEST OF EXTENDED LEAF PHENOLOGY

Trisha M. Markus, Lauren S. Pile Knapp, David R. Coyle, and G. Geoff Wang

ABSTRACT

Chinese privet (*Ligustrum sinense*, hereafter: privet) is an invasive shrub common throughout the Eastern and Southeastern United States. The foremost concern of privet is its highly competitive nature; it displaces native plants, their associated pollinators, and wildlife communities. Extended leaf phenology (ELP) is an adaptation which allows understory plants in deciduous forests to take advantage of higher sunlight levels during the dormant season when canopy foliage is absent and has been associated with many invasive plant species. To measure the ELP potential of privet, we created light response curves for privet along two habitat types (forest edge and forest interior) using a gas exchange analyzer. Privet actively photosynthesized after native canopy leaf-off, and carbon assimilation rates did not differ between habitats. We observed habitat and seasonal variability in privet leaf shedding and leaf replacement with privet retaining leaves in both habitats throughout October 2019 to February 2020. Individuals along the forest edge shed leaves in November 2020, and privet in the interior habitat did not shed leaves until February 2021. Our results provide evidence of the competitive ELP potential of privet and suggest potential issues for planning winter foliar herbicide applications based on current management recommendations.

INTRODUCTION

Invasive species are an economic concern, costing more than $26 billion per year since 2010 and totaling more than $34 billion in the forest sector from 1960 to 2017 (Crystal-Ornelas and others 2021). They are also an ecological concern, wreaking havoc on native species abundance and diversity (Crystal-Ornelas and others 2021). Additionally, the horticulture industry's $38 billion economic contribution in 2021 suggests an increase in the propagule pressure of many invasive plant species land managers try to control, such as privets (*Ligustrum*), silverberries (*Elaeagnus*), and Callery/Bradford pear (*Pyrus calleryana*), as well as many others (Beaury and others 2021, Wei and others 2023).

Chinese privet (*Ligustrum sinense*, hereafter: privet) is a common invasive shrub in the Eastern and Southeastern United States that is still sold worldwide in nurseries and online as a hedgerow shrub for its attractive white flowers. The foremost concern about the invasion of privet is its ability to outcompete native plants for resources in the invaded range. Like many invasive plants, privet thrives in edge habitats where disturbances are common and light availability is high, such as roadsides and along stream systems or in depressions where moisture supply is abundant (Ward 2002). However, its high shade tolerance also allows it to live in deciduous hardwood understories and develop dense thickets that disrupt native species' growth and prevent tree regeneration (Merriam and Feil 2002), but it seldom invades upland evergreen conifer forests such as natural or plantation pine (*Pinus* spp.) stands (Tiller 2015).

Understanding the physiology of an invasive plant is vital to determine best management practices for control and eradication. Several hypotheses have been proposed to explain the relative success of nonnative plant species; for

Trisha M. Markus, Graduate Student, Clemson University, Clemson, SC 29634
Lauren S. Pile Knapp, Research Ecologist, U.S. Department of Agriculture, Forest Service, Northern Research Station, Columbia, MO 65211
David R. Coyle, Assistant Professor, Clemson University, Clemson, SC 29634
G. Geoff Wang, Professor, Clemson University, Clemson, SC 29634

example, intermediate disturbance, empty niche, enemy release, and novel weapons (Callaway and Ridenour 2004, Eggeling 1947, Elton 1927, Grinnell 1917, Watt 1947). One such hypothesis unique to plants is extended leaf phenology (ELP). ELP is an adaptation of evergreen and semi-evergreen plants growing under the forest canopy, which allows them to take advantage of the absence of deciduous canopy foliage during their dormant season (Smith 2013, Smith and Hall 2016). While not unique to invasive plants, ELP allows them to gain carbon (C) through photosynthesis during what is traditionally the dormant season, giving these species an advantage over those that do not photosynthesize during this time. The measurement of C production of plants requires a measure of photosynthetic activity (Lazar 2003) and determining the seasonal change in photosynthetic rates throughout the year for invasive plants may indicate how these plants compete with native flora by acquiring C when the native overstory canopy is absent (Feng and others 2007).

The overall goal of this study was to understand the competitive ability of using ELP by privet to improve management recommendations. For example, further understanding privet's photosynthetic activity and physiological adaptations should allow land managers to plan herbicide applications seasonally to align with the time when the plants translocate resources via the phloem for foliar application or xylem for soil applications (Cobb and Reade 2010, Higgins and Richardson 2014). Our objectives were to 1) measure the photosynthetic activity of privet during the native deciduous dormant season (October through March), 2) compare light saturation points (LSP) and light concentration points (LCP) between interior and edge habitats containing privet, and 3) compare the monthly photosynthetic rate of privet as the canopy closes in the spring.

MATERIALS AND METHODS

Study Species

Privet is a member of the Oleaceae family and is native to southeast China, Taiwan, and Vietnam. There are several *Ligustrum* species in the United States and the following is a description for *L. sinense*. It is a shrub or tree reaching average heights of 1.5 to 4.5 m but it can grow as tall as 9 m. The leaves are oppositely arranged approximately 2.5 cm along the stem and are evergreen or semideciduous. Leaf blades are elliptical with entire margins, up to 2.5 cm wide and 5 cm long. Flowers occur prolifically throughout the shrub in 5- to 10-cm-long panicles. Each flower has four white to off-white petals and two stamens attached to a fused petal that make up the corolla. A blue-black, berry-like fruit is formed from a single pistil (Kirkman and others 2007). While privet produces a large mast of berries that are consumed and spread primarily by birds, it also spreads by reproducing asexually via root sprouting (Wang and Grant 2012).

Study Area

We selected two sites within the Clemson Experimental Forest (CEF), a 7,082-ha multi-use forest for recreation, research, and timber management in upstate South Carolina. Historically, the land was used for cotton farming until land reclamation attempts occurred in 1934, when timber and kudzu were planted (Sorrells 1984). Both sites consisted of native deciduous forests with an actively spreading infestation of privet in the understory. Site 1 was in the New Hope (NH) Wildlife Management Area (WMA) and site 2 was in the Fant's Grove (FG) area of the CEF at the Butch Kennedy's Trail Head (fig. 1). Both sites were split into an understory (interior: >15 m from forest road or trail) and a forest edge (<15 m from road or trail) plot. The soil type for the interior NH was Pacelot, fine sandy loam, and Cecil clay loam for the exterior plots; the soil type for the FG area included Cecil, sandy loam for the interior and exterior sites (USDA NRCS 2020). The common overstory species at each location were oaks (*Quercus*), hickories (*Carya*), yellow-poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), loblolly pine (*Pinus taeda*), and Virginia pine (*P. virginiana*). Midstory species included red maple (*Acer rubrum*), river birch (*Betula nigra*), American holly (*Ilex opaca*), and winged elm (*Ulmus alata*). Other invasive species at both sites included silverthorn (*Elaeagnus pungens*), chinaberry (*Melia azedarach*), Japanese honeysuckle (*Lonicera japonica*), sacred bamboo

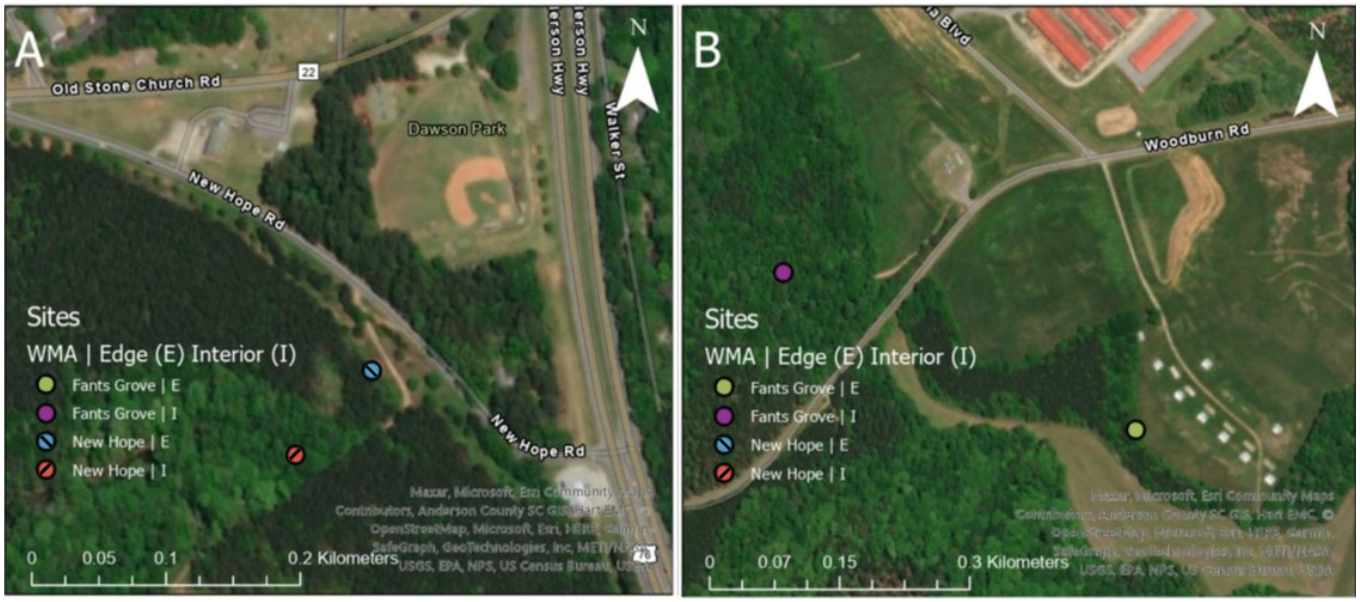

Figure 1—Site locations in the Clemson Experimental Forest of (a) New Hope (NH) and (b) Fant's Grove (FG) edge and interior habitats.

(*Nandina domestica*), and multiflora rose (*Rosa multiflora*).

Gas Exchange Measurements

Gas exchange was measured using the CIRAS-2 with the PLC cuvette attachment (P.P. Systems, Amesbury, MA). A total of six privet shrubs were selected at each site, three in the interior plot and three in the edge plot, with each shrub within a plot 10–20 m apart. The CIRAS-2, via the PLC-6U cuvette, was attached to an individual leaf, free of defects and securely attached to the stem, near the top of the canopy of each of the six shrubs per site once per month. For accurate measurements, and to prevent water damage to the equipment, gas exchange recordings were taken only on sunny days from November 2019 to January 2021. The cuvette, stabilized by a tripod, served to maintain a constant CO_2, temperature, and relative humidity around the leaf to standardize photosynthetic rate measurements (Lazar 2003). The selected leaf was acclimated for 2 minutes to the cuvette chamber conditions before measurement. The CIRAS-2 then measured the photosynthetic rate (P_N [µmol (CO_2) m^{-2} s^{-1}]) at light transmittance levels [photosynthetic active radiation (PAR)] from high to low [2000, 1500, 1000, 500, 100, 50, 25, 0 µmol (photon) m^{-2} s^{-1}] with an acclimation period of 60 seconds between recordings (Ralph and Gademann 2005). CO_2 was set as nonlimiting based on the atmospheric CO_2 levels at the time of the study (410 ppm, NOAA 2019). Relative humidity was held constant at 50 percent and temperature was held constant at 25 °C.

Statistical Analysis

All analyses were conducted using R (version 4.3.0, R Core Team 2023). We fit light response curves using a nonrectangular hyperbola-based model (equation 1) (Marshall and Biscoe 1980).

$$P_N = \frac{\Phi_{(I_0)} \times PAR + P_{gmax} - \sqrt{(\Phi_{(I_0)} \times PAR + P_{gmax})^2 - 4\theta \times \Phi_{(I_0)} \times PAR \times P_{gmax}}}{2\theta} - R_D \quad (1)$$

where

P_N = net photosynthetic rate [µmol (CO_2) m^{-2} s^{-1}]

$\phi_{(Io)}$ = quantum yield with irradiance equal to 0 [µmol (CO_2) µmol (photon)$^{-1}$]

PAR = photosynthetic active radiation [µmol (photon) m^{-2} s^{-1}]

P_{gmax} = asymptotic estimate of the max gross photosynthetic rate [µmol (CO_2) m^{-2} s^{-1}]

θ = convexity (dimensionless)

R_D = dark respiration rate [µmol (CO_2) m^{-2} s^{-1}]

We confirmed that our data did not violate the assumptions of normality and equal variance using a Shapiro-Wilk test and equal variance F-test, respectively (α = 0.05). We then compared curve coefficients from the nonrectangular

hyperbola-based model (equation 1) and individual light levels between sites and habitat (i.e., forest edge and forest interior), and among months using analysis of variance and Tukey's posthoc test for pairwise comparisons.

Unfortunately, data from November 2019 through September 2020 had to be removed from the analysis due to issues with the gas exchange analyzer needing to be calibrated and troubleshooting that throughout the study period. Continuous measurements were attempted from November 2019 to February 2020 on all samples because leaves were present and in good standing to be measured. However, we were unable to have consistent measurements for an entire year to compare spring and summer photosynthetic activity with fall and winter.

RESULTS

There were significant differences in P_N between sites for interior habitats with NH having lower P_N than FG but no significant P_N difference between sites for the edge habitats in October 2020 (F = 8.333 and 0.021; P = 0.006 and 0.885, respectively; fig. 2). We also saw a significant difference between sites for all interior plots with NH having lower P_N than FG (F = 51.324, P <0.001). Thus, the sites were analyzed separately for individual PAR

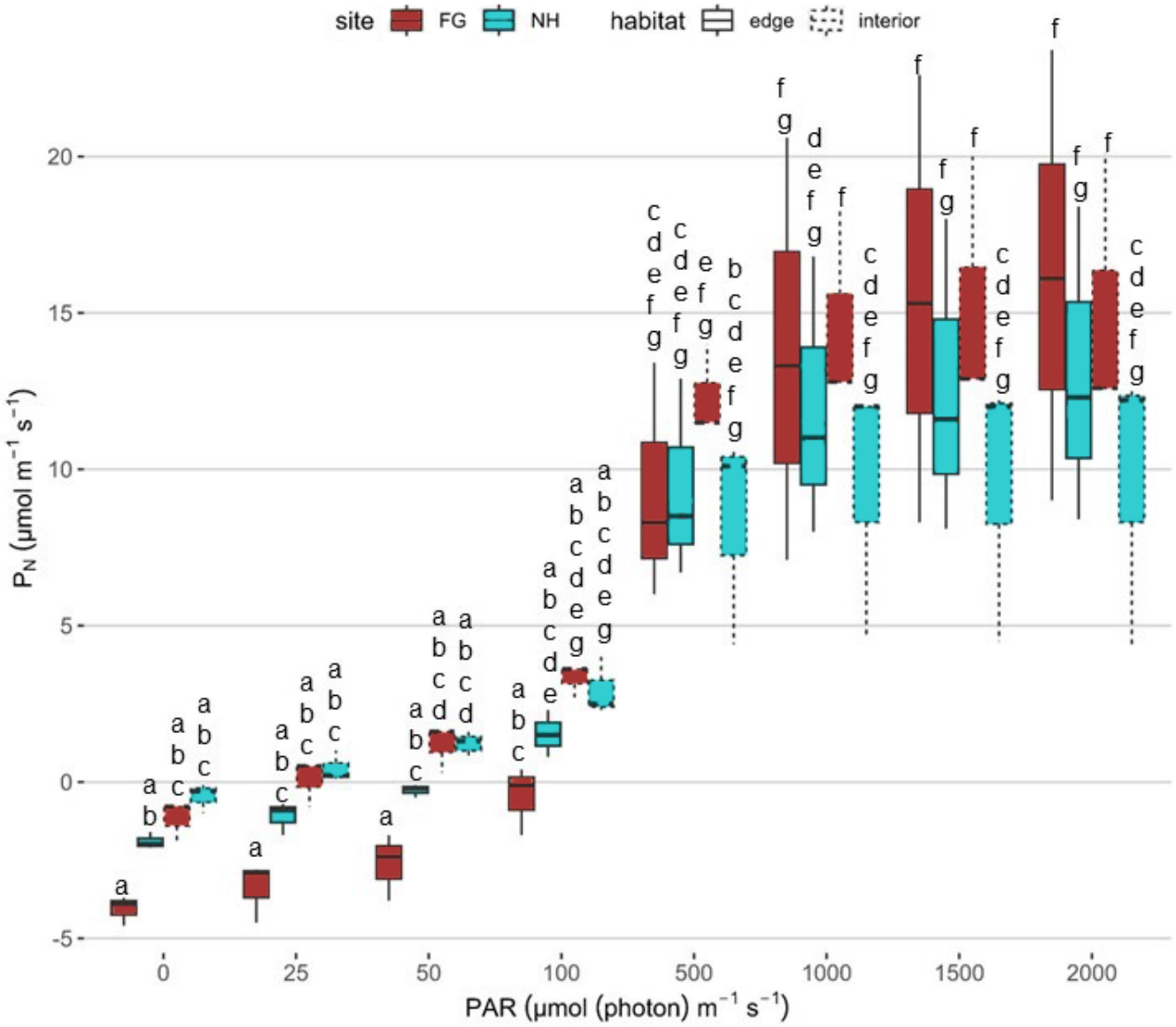

Figure 2—October 2020 light level comparisons of each site, New Hope (NH) and Fant's Grove (FG) for interior and edge habitat types. There was a significant difference among sites for interior habitat (P = 0.006) but no difference among sites for edge habitats (P = 0.885). Different letters represent significant differences with α = 0.05.

[0–2000 µmol (photon) m^{-2} s^{-1}] levels. When examining each PAR level for the NH site, P_N differed among months with October having greater P_N rates than December and January (F = 3.051, *P* = 0.035, fig. 3) and all others not significantly different. At the FG site, individual PAR level P_N did not differ among months (F = 1.802, *P* = 0.176, fig. 4).

The calculated curve coefficients using the nonrectangular hyperbola model (equation 1) showed no significant differences between sites or habitats for the month of October for LSP (P_{gmax}) (F = 3.654 and 2.254; *P* = 0.088 and 0.167, respectively), LCP ($\phi_{(Io)}$) (F = 3.271 and 1.965; *P* = 0.104 and 0.195, respectively), curve convexity (θ) (F = 0.081 and 2.277; *P* = 0.783 and 0.166, respectively). There was a significant difference in curve coefficients for dark respiration (R_D) with NH having a lower value than FG (F = 12.625, *P* = 0.006) and interior habitat having a lower value than edge habitat (F = 31.096, *P* <0.001).

The interior habitat curve coefficient for LSP (P_{gmax}) had significant differences between sites with NH having a lower LSP value than the FG site (F = 10.460, *P* = 0.005) with no differences among months (F = 0.236, *P* = 0.870). There was

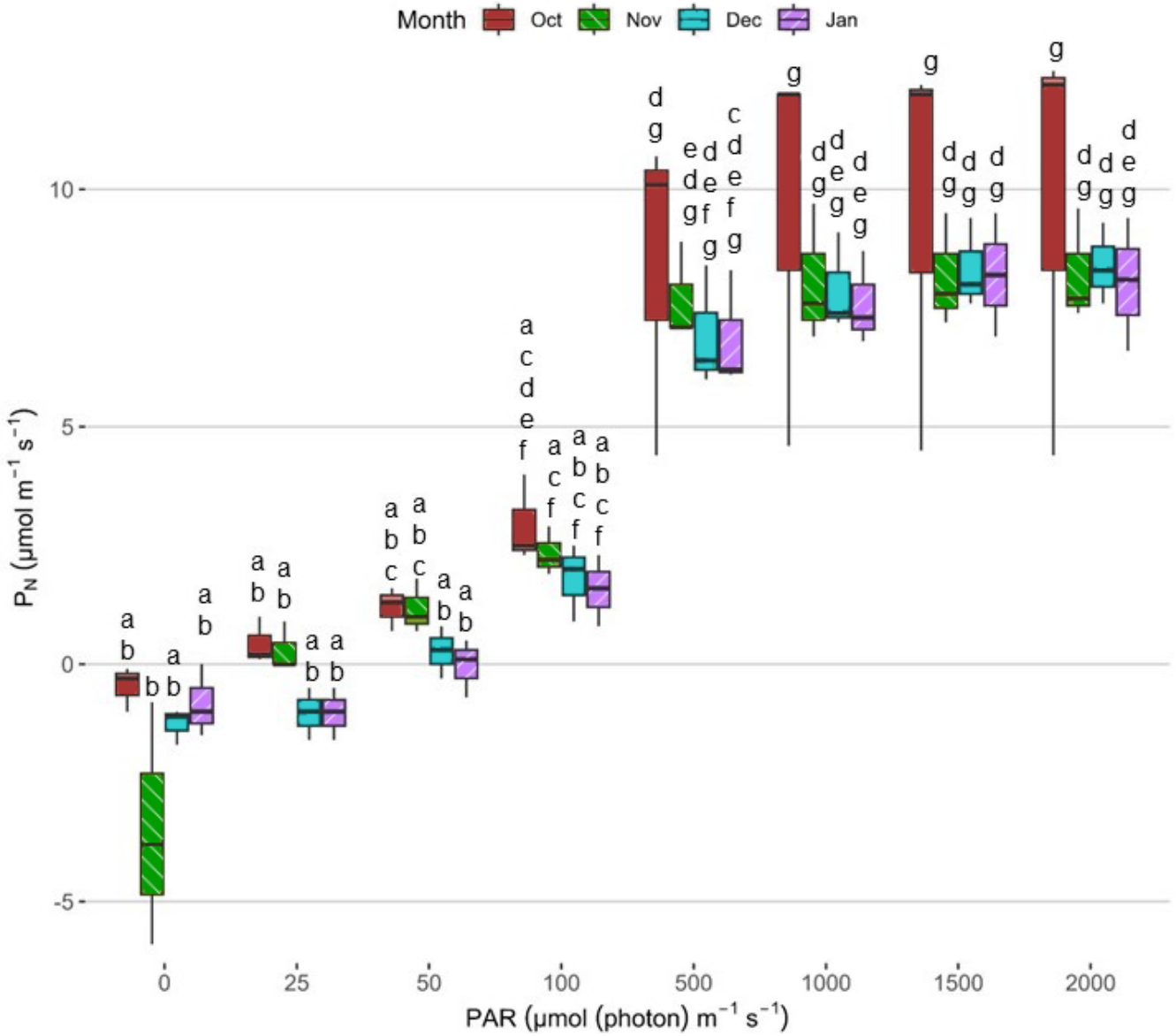

Figure 3—Light level comparison for New Hope (NH) interior habitats from October 2020 to January 2021. P_N rates were not significantly different other than between October and December (*P* = 0.043) and October and January (*P* = 0.022) with October having significantly higher P_N rates than December and January. Different letters denote significant differences with α = 0.05.

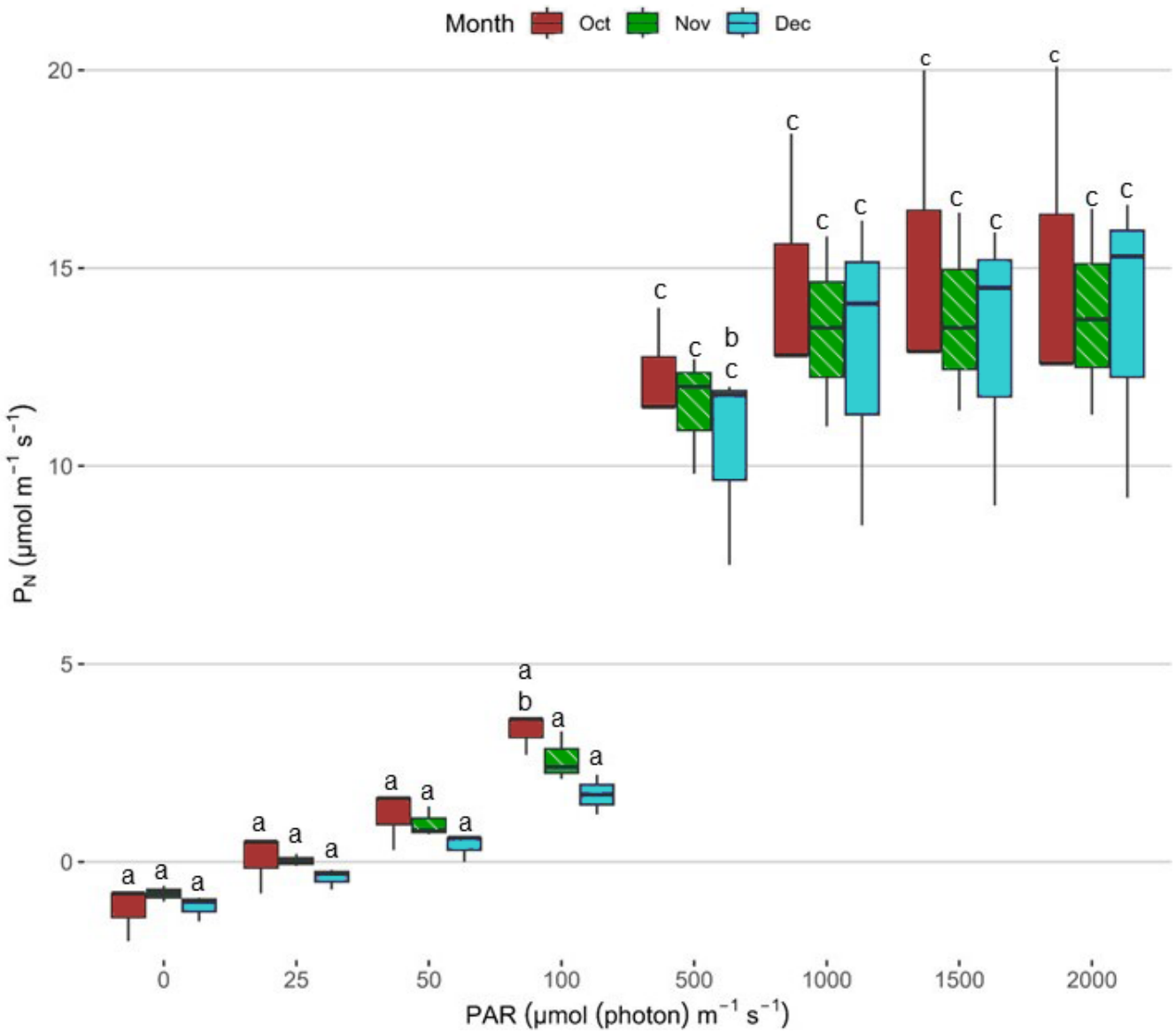

Figure 4—Light level comparison for Fant's Grove (FG) interior habitats from October 2020 to January 2021. P_N rates did not significantly differ among months (P = 0.176). Different letters denote significant differences with α = 0.05.

no significant difference between sites or among months for LCP ($\phi_{(Io)}$) (F = 1.212 and 1.970; P = 0.287 and 0.159, respectively), curve convexity (θ) (F = 3.399 and 1.526; P = 0.839 and 0.246, respectively), and dark respiration (R_D) (F = 1.530 and 1.236; P = 0.0.234 and 0.330, respectively).

In March 2020, leaf senescence and new leaf emergence occurred for all sites and habitats. Edge shrub leaf senescence occurred in November 2020, thus excluding data for this habitat from November 2020 through January 2021 and leaves were too small to measure in February 2021. The interior habitat leaves were in good standing until leaf senescence occurred in February 2021.

DISCUSSION

Our data provides evidence that privet actively photosynthesizes in the fall and winter months when the canopy is absent in deciduous forests. Photosynthetic rates continued in the FG sites as the winter progressed, and senescence eventually occurred in January. However, photosynthetic rates fell in December and January in the NH interior habitat until senescence occurred in February 2021. Still, any C assimilation occurring while most native species are dormant gives invasive plants a competitive advantage via ELP (Smith and Hall 2016). Privet seedling survival can occur at light levels as low as 1 to 5 percent

full sun but long-term survival is limited (Wang and Grant 2014). Winter photosynthesis would be significant to the success of privet as an invasive shrub and may explain why the species is more common in deciduous forests than in pine forests (Tiller 2015).

There was variability in when the plant was considered evergreen to semi-evergreen. Based on visual observations throughout the study in the winter of 2019–2020, privet was evergreen whereas in the winter of 2020–2021, all sampled privet were semi-evergreen, with the plants in the edge sites behaving more like deciduous species (i.e., most leaves dropped in November 2020). Species with the ability to adapt to various site characteristics will show differences in morphology, assimilation rates, and chemical production, which explains privet's ability to switch from evergreen to semi-evergreen based on environmental conditions (Fan and others 2019, Oduor and others 2016, Xiong and others 2021).

Seasonal variability in photosynthetic rates of semi-evergreen and evergreen species is expected (Rothstein and Zak 2001). Rothstein and Zak (2001) saw variability in photosynthesis and found limited growth in privet during canopy closure, while growth increased when the canopy opened. Photosynthetic rates depend on several factors, including water availability, temperature, and humidity, which cause the seasonal variability of assimilation rates. However, some species are better adapted to withstand the season's variability, allowing them to be highly competitive with deciduous species (Feng and others 2007). By having extended winter phenology (i.e., becoming semi-evergreen) or remaining evergreen during the entire winter, privet could accumulate a significant amount of carbon at a time when light is readily available due to the absence of canopy foliage given that air and soil temperatures are above freezing (Ensminger and others 2006).

Understanding the seasonal photosynthetic rates and morphological differences in privet can help determine management solutions for invaded ranges and sensitive ecosystems. Current management recommendations for privet include broad-spectrum application of foliar glyphosate in the fall (late October to early November) after desirable plants have senesced (Enloe and others 2018). Foliar herbicides are transported throughout the plant via C assimilation, having a better understanding of photosynthetic capabilities during the winter aids in recommendations for application timing and chemical formulation and rate (Draber and others 1991, Tiaz and others 2015). Results from our study suggest the phenology of privet may vary from year to year, and close monitoring of phenology is warranted when foliar herbicide applications are prescribed as treatment. Glyphosate can kill nontarget native evergreen species that grow in the forest understory [e.g., American holly, Christmas fern (*Polystichum acrostichoides*)]. Attempting to control privet with native evergreen species present may complicate control efforts. Finally, a further understanding of the photosynthetic ability of privet may also assist in finding more targeted herbicides for its control.

CONCLUSIONS

This study could only complete measurements for part of the year, thus preventing comparisons of understory privet photosynthetic rates in open- versus closed-canopy conditions. However, the snapshot of information gathered during the project did show that privet actively photosynthesizes during canopy absence. Additionally, this study discovered a variability in leaf senescence from one year to the next, posing issues with current herbicide prescription recommendations for homeowners and land managers and reducing the economic impact of treating privet. This study can be further improved by adding more elements to data collection, including measuring the growth rate, soil water availability, precipitation, ambient air and soil temperature, and environmental light levels during the time of measurement. The scope of the study would also benefit by comparing photosynthetic rates of privet with native and invasive species at each site to better understand its competitive ability.

ACKNOWLEDGMENTS

We would like to acknowledge Pat Layton for guidance and support. We would also like to thank the two reviewers, Jess Hartshorn and Kelly Oten (Clemson University and North Carolina State University, respectively) for their feedback along with lab mates (past and present), Bridget Bloodwood, Calvin Norman, and Ryan Bohannon, for their assistance on previous versions of this manuscript.

LITERATURE CITED

Beaury, E.M.; Patrick, M.; Bradley, B.A. 2021. Invaders for sale: the ongoing spread of invasive species by the plant trade industry. Frontiers in Ecology and the Environment. 19(10): 550–556. https://doi.org/10.1002/fee.2392.

Callaway, R.M.; Ridenour, W.M. 2004. Novel weapons: invasive success and the evolution of increased competitive ability. Frontiers in Ecology and the Environment. 2(8): 436–443. https://doi.org/10.2307/3868432.

Cobb A.H.; Reade J.P.H. 2010. Herbicides and plant physiology. Second ed. West Sussex, UK: Wiley-Blackwell. 286 p.

Crystal-Ornelas, R; Hudgins, E.J.; Cuthbert, R.N. [and others]. 2021. Economic costs of biological invasions within North America. NeoBiota. 67: 485–510. https://doi.org/10.3897/neobiota.67.58038.

Draber, W.; Tietjen, K.; Kluth, J.F.; Trebst, A. 1991. Herbicides in photosynthesis research. Angewandte Chemie. International Edition in English. 30(12): 1621–1633. https://doi.org/10.1002/anie.199116211.

Eggeling, W.J. 1947. Observations on the ecology of the Budongo rain forest, Uganda. Journal of Ecology. 34: 20–87. https://doi.org/10.2307/2256760.

Elton, C.S. 1927. Animal ecology. London, England: Sidgwick & Jackson, Ltd. 207 p.

Enders, M.; Havemann, F.; Ruland, F. [and others]. 2020. A conceptual map of invasion biology: integrating hypotheses into a consensus network. Global Ecology and Biogeography. 29(6): 978–991. https://doi.org/10.1111/geb.13082.

Enloe, S.F.; O'Sullivan, S.E.; Loewenstein, N.J. [and others]. 2018. The influence of treatment timing and shrub size on Chinese privet (*Ligustrum sinense*) control with cut stump herbicide treatments in the southeastern United States. Invasive Plant Science and Management. 11(1): 49–55. https://doi.org/10.1017/inp.2018.3.

Ensminger, I.; Busch, F.; Huner, N.P.A. 2006. Photostasis and cold acclimation: sensing low temperature through photosynthesis. Physiologia Plantarum. 126(1): 28–44. https://doi.org/10.1111/j.1399-3054.2006.00627.x.

Fan, Y., Chen, J.; Wang, Z. [and others]. 2019. Soybean (*Glycine max* L. Merr.) seedlings response to shading: leaf structure, photosynthesis and proteomic analysis. BMC Plant Biology. 19(1): 34. 12 p. https://doi.org/10.1186/s12870-019-1633-1.

Feng, Y.; Wang, J.; Sang, W. 2007. Biomass allocation, morphology and photosynthesis of invasive and noninvasive exotic species grown at four irradiance levels. Acta Oecologica. 31(1): 40-47. https://doi.org/10.1016/j.actao.2006.03.009.

Grinnell, J. 1917. The niche-relationships of the California thrasher. Auk. 34(4): 427–433.

Higgins, S.I.; Richardson, D.M. 2014. Invasive plants have broader physiological niches. Proceedings of the National Academy of Sciences. 111(29): 10610–10614. https://doi.org/10.1073/pnas.140607511.

Kirkman, L.K.; Brown, C.L.; Leopold, D.J. 2007. Native trees of the Southeast. Portland, OR: Timber Press. 370 p.

Lazar, T. 2003. Taiz, L. and Zeiger, E. Plant physiology. 3rd ed. Annals of Botany. 91(6): 750–751. https://doi.org/10.1093/aob/mcg079.

Marshall, B.; Biscoe, P.V. 1980. A model for C3 leaves describing the dependence of net photosynthesis on irradiance. Journal of Experimental Botany. 31(1): 29–39. https://doi.org/10.1093/jxb/31.1.29.

Merriam, R.W.; Feil, E. 2002. The potential impact of an introduced shrub on native plant diversity and forest regeneration. Biological Invasions. 4(4): 369–373. https://doi.org/10.1023/A:1023668101805.

National Oceanic and Atmospheric Administration [NOAA]. 2019. Trends in atmospheric carbon dioxide. Global Monitoring Laboratory. https://www.esrl.noaa.gov/gmd/ccgg/trends/global.html. [Date accessed: May 15, 2019].

Oduor, A.M.O.; Leimu, R.; van Kleunen, M. 2016. Invasive plant species are locally adapted just as frequently and at least as strongly as native plant species. Journal of Ecology. 104(4): 957–968. https://doi.org/10.1111/1365-2745.12578.

R Core Team (2023). R: a language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. https://www.R-project.org/.

Ralph, P.J.; Gademann, R. 2005. Rapid light curves: a powerful tool to assess photosynthetic activity. Aquatic Botany. 82(3): 222–237. https://doi.org/10.1016/j.aquabot.2005.02.006.

Rothstein, D.E.; Zak, D.R. 2001. Photosynthetic adaptation and acclimation to exploit seasonal periods of direct irradiance in three temperate, deciduous-forest herbs. Functional Ecology. 15(6): 722–731. https://doi.org/10.1046/j.0269-8463.2001.00584.x.

Smith, L.M. 2013. Extended leaf phenology in deciduous forest invaders: mechanisms of impact on native communities. Journal of Vegetation Science. 24(6): 979–987. https://doi.org/10.1111/jvs.12087.

Smith, L.M.; Hall, S. 2016. Extended leaf phenology may drive plant invasion through direct and apparent competition. Oikos. 125(6): 839–848. https://doi.org/10.1111/oik.02529.

Sorrells, R.T. 1984. The Clemson Experimental Forest: its first fifty years. Clemson, SC: Clemson University, College of Forest and Recreation Resources. 52 p.

Taiz, L.; Zeiger, E.; Møller, I.M.; Murphy, A. 2015. Plant physiology and development. 6th ed. Sunderland, MA: Sinauer Associates, Inc. 761 p.

Tiller, M.B. 2015. Effects of yaupon, Chinese privet, and Chinese tallow on understory fuel flammability in east Texas hardwood and pine ecosystems. M.S. thesis. Nacogdoches, TX: Stephen F. Austin State University. 152 p.

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2020. Web soil survey. https://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm. [Date accessed: September 17, 2020].

Wang, H.-H.; Grant, W.E. 2012. Determinants of Chinese and European privet (*Ligustrum sinense* and *Ligustrum vulgare*) invasion and likelihood of further invasion in southern U.S. forestlands. Invasive Plant Science and Management. 5(4): 454–463. https://doi.org/10.1614/IPSM-D-12-00038.1.

Wang, H.-H.; Grant, W.E. 2014. Invasion of eastern Texas forestlands by Chinese privet: efficacy of alternative management strategies. Diversity. 6(4): 652–664. https://doi.org/10.3390/d6040652.

Ward, R.W. 2002. Extent and dispersal rates of Chinese privet (*Ligustrum sinense*) invasion on the upper Oconee River floodplain, north Georgia. Southeastern Geographer. 42(1): 29–48.

Watt, A.S. 1947. Pattern and process in the plant community. Journal of Ecology. 35(1/2): 1–22. https://doi.org/10.2307/2256497.

Wei, X.; Khachatryan, H.; Hodges, A. [and others]. 2023. Exploring market choices in the US ornamental horticulture industry. Agribusiness. 39: 65–109. https://doi.org/10.1002/agr.21769.

Xiong, F.; Nie, X.; Yang, L. [and others]. 2021. Non-target metabolomics revealed the differences between *Rh. tanguticum* plants growing under canopy and open habitats. BMC Plant Biology. 21: 119. 13 p. https://doi.org/10.1186/s12870-021-02897-8.

EFFICACY OF MID-ROTATION BRUSH CONTROL APPLICATIONS USING IMAZAPYR AND GLUFOSINATE

Andrew Ezell, Andrew Self, and John Ezell

ABSTRACT

A total of 18 treatments were applied to the understory vegetation in a 31-year-old loblolly pine (*Pinus taeda*) plantation. The study site was on the Mississippi State University John Starr Forest in Oktibbeha County, Mississippi. Plots were evaluated at 14, 28, 56, 90, and 120 days after treatment with final evaluation at 1 year after treatment. Control of red oak species (Section *Quercus*) and red maple (*Acer rubrum*) was generally less than desirable unless imazapyr was in the treatment. Glufosinate performed better on white oak species (Section *Lobatae*), persimmon (*Diospyros virginiana*) and subarborescent species. The only treatments which provided desirable levels of control when all hardwood species were considered were those which contained imazapyr alone or those with mixtures of imazapyr and glyphosate.

INTRODUCTION

Mid-rotation brush control has been used more frequently in loblolly pine (*Pinus taeda*) management since the introduction of imazapyr to the market. Both aerial and ground applications are used. The use of ground application permits the addition of other materials which may be desirable if natural pine regeneration or species resistant to imazapyr are to be controlled. Glufosinate has demonstrated potential for use in controlling natural pine, but additional information is needed regarding the efficacy of glufosinate in controlling hardwood species.

METHODS

The study was conducted in a 31-year-old loblolly pine plantation located on the Mississippi State University John Starr Forest in Oktibbeha County, MS. The plantation had been thinned twice and had a well-established understory of 15 hardwood species, 4 subarborescent species, and vines.

A total of 54 plots were utilized in the study. Each plot was a rectangular area 30 feet by 50 feet. The sample area in each plot was a swath 10 feet by 50 feet centered in the treatment plot.

A total of 18 treatments were utilized in the study (table 1). These included four glufosinate products, two imazapyr products, one glyphosate product, and an untreated check.

All treatments were applied August 23, 2021, using a CO_2-powered backpack sprayer equipped with a pole extension and KLC-9 nozzle. The equipment simulates the application from either a helicopter or skidder. Total spray volume was 30 gallons per acre to reflect the typical application level used in ground applications. Each treatment was replicated three times in a completely randomized design.

All stems of woody species in the sample areas were recorded by species and height class prior to treatment application and 1 year after treatment. Brownout was recorded by vegetation group (grasses, forbs, hardwoods) at 14, 28, 56, 90, and 120 days after treatment. Control is based on the reduction in stems comparing the pretreatment and 1-year-after-treatment data.

Andrew Ezell, Professor and Department Head (Emeritus), Mississippi State University, Starkville, MS 39762
Andrew Self, Associate Extension Professor, Mississippi State University, Grenada, MS 38901
John Ezell, Assistant Clinical Professor, Mississippi State University, Starkville, MS 39762

Table 1—List of treatments in 2021 mid-rotation brush control study

Treatment	Herbicide (rate/acre)
1	Untreated check
2	Finale® XL (54.8 ounces)
3	BASF glufosinate (54.8 ounces)
4	Cheetah® Pro (54.8 ounces)
5	Interline (54.8 ounces)
6	Finale® XL (82.2 ounces)
7	BASF glufosinate (82.2 ounces)
8	Cheetah® Pro (82.2 ounces)
9	Interline (82.2 ounces)
10	Chopper® GEN 2 (48.0 ounces)
11	Arsenal® AC (20.0 ounces)
12	Finale® XL (54.8 oz) + Chopper® GEN 2 (48.0 ounces)
13	Finale® XL (54.8 oz) + Arsenal® AC (20.0 ounces)
14	Finale® XL (82.2 oz) + Chopper® GEN 2 (48.0 ounces)
15	Finale® XL (82.2 oz) + Arsenal® AC (20.0 ounces)
16	Accord® XRT II (48.0 ounces)
17	Accord® XRT II (48.0 ounces) + Chopper® GEN 2 (48.0 ounces)
18	Accord® XRT II (48.0 ounces) + Arsenal® AC (20.0 ounces)

All products made by BASF Corporation, Florham Park, NJ.
All treatments included Elite Imperial (19.2 ounces per acre) and Elite Supreme (10.0 ounces per acre).

RESULTS

Control of red oak species, which included cherrybark (*Quercus pagoda*), water (*Q. nigra*), willow (*Q. phellos*), and Shumard (*Q. shumardii*) oaks was generally less than desirable unless imazapyr was included in the treatment (table 2). Control of red maple was very low unless imazapyr was included in the treatment. Glufosinate alone provided better control on white oak species including white (*Q. alba*) and post (*Q. stellata*) oaks, and persimmon, but overall best results were for treatments containing imazapyr. While not the primary target of mid-rotation brush control applications, most of the treatments in the study provided good to excellent control of subarborescent species in the plots. Of principal interest in mid-rotation brush control applications is total control resulting from the treatment. In this consideration, only those treatments containing the imazapyr alone (treatments 10 and 11) and those with imazapyr and glyphosate (treatments 17 and 18) provided desirable levels of control. An item of concern was the reduction in total control, which was observed in treatments which had glufosinate added to the imazapyr (treatments 12, 13, 14, and 15) as compared to the treatments with imazapyr alone on those with imazapyr and glyphosate. Additional testing is being conducted to further evaluate the latter observation.

Table 2—Percentage of stem reduction in 2021 mid-rotation brush control study

Treatment	RED	REM	PER	WHO	SUB	Total
1	23.0de	+16.7e	+16.7e	–	18.1e	0.0f
2	+12.5f	+24.3e	33.3d	100.0a	82.4b	29.0e
3	0.0e	+13.8e	0.0e	100.0a	74.6c	+7.2f
4	7.1e	45.0d	–	–	60.2d	23.3ef
5	20.6de	14.1d	73.7bc	75.0b	27.6e	27.7e
6	29.4de	17.0d	75.0bc	80.0b	63.6d	29.0e
7	50.0d	15.0d	100.0a	100.0a	84.8b	38.6e
8	100.0a	+40.0f	57.1c	–	73.2c	20.0ef
9	+66.7f	3.3d	84.6b	100.0a	42.5e	32.7e
10	100.0a	100.0a	100.0a	100.0a	89.6ab	100.0a
11	95.7a	96.6a	100.0a	100.0a	87.7ab	97.8a
12	91.3a	70.6c	60.0c	100.0a	100.0a	81.7b
13	66.7d	100.0a	100.0a	100.0a	91.1a	82.1b
14	100.0a	85.3b	100.0a	100.0a	95.8a	78.9b
15	93.3a	69.2c	100.0a	100.0a	92.4a	84.1b
16	85.7b	68.4c	100.0a	–	98.8a	76.7b
17	100.0a	89.6ab	100.0a	100.0a	97.7a	96.1a
18	100.0a	100.0a	100.0a	–	100.0a	95.3a

See table 1 for treatment descriptions.
Values in a column followed by the same letter do not differ at α = 0.05.
– = insufficient stems for analysis.
+ = increase in stems.
RED = red oaks (cherrybark, water, willow, shumard); REM = red maple; PER = persimmon; WHO = white oaks (white, post); SUB = subarborescents (winged sumac, American beautyberry, Devil's walkingstick, Chinese privet); Total = all hardwood species (no subarborescents).

EVALUATING DORMANT-SEASON HERBICIDE APPLICATIONS FOR THE CONTROL OF CHINESE PRIVET (*LIGUSTRUM SINENSE*)

Karen Vaughn, Lauren S. Pile Knapp, and G. Geoff Wang

ABSTRACT

Chinese privet (*Ligustrum sinense*) is an aggressive, nonnative shrub of the Southeastern United States. Its evergreen leaves allow for herbicide treatment in the dormant season when the impact to native species can be minimized. The purpose of this study was to determine which combination of chemical herbicide (glyphosate, metsulfuron, or a mixture of both) and application method (backpack sprayer or mistblower) applied during the dormant season would result in the greatest control of Chinese privet with minimal nontarget impacts. Study results suggest that herbicide, regardless of the chemical formulation, will kill more than 50 percent of the basal area of Chinese privet. However, glyphosate did impact native species that remained green and resulted in an increase in Japanese stiltgrass (*Microstegium vimineum*). Mistblowers were more effective at controlling Chinese privet than backpack sprayers regardless of herbicide. Although no differences in native species richness were found between the method of application, mistblowers also had fewer impacts on native flora, including tree and shrub reproduction, than backpack sprayers. Mistblowers are not commonly used for controlling invasive shrubs; however, their utility for controlling large and tall shrub infestations should be considered.

INTRODUCTION

Chinese privet (*Ligustrum sinense*) is a nonnative invasive shrub that has become ubiquitous throughout the Southeastern United States, especially in the region's floodplain and bottomland forests. Once prized as an ornamental hedge and favored for its glossy, often evergreen foliage and abundant panicles of white flowers, its bird- and water-dispersed fruit, rapid growth, and generalist habitat requirements have allowed Chinese privet to spread rapidly and form dense thickets, often impacting native flora and tree regeneration (Greene and Blossey 2014, Merriam and Feil 2002, Turner and others 2022, Wilcox and Beck 2007). Controlling large and dense invasive populations of Chinese privet has become a management priority for many natural areas in the Southeastern United States. However, its size and density can challenge resources, and subsequently, controlling it can have adverse consequences for native flora. For example, any individual stem control method, such as cut-stump applications or mechanical removal, is often extremely laborious and practically unfeasible at large scales.

Although generally considered evergreen to semi-evergreen, leaf retention on Chinese privet is variable and likely dependent on site and environmental factors, including shading, drought, and soil and air temperature (Munger 2003). Extended leaf phenology, whereby understory plants assimilate carbon prior to overstory canopy leaf expansion and after canopy senescence, is hypothesized as a trait of invasiveness (Fridley 2012). The evergreenness of Chinese privet and subsequently its extended leaf phenology are factors contributing to its success in deciduous hardwood forest understories. Extended leaf phenology may also confer an advantage for control with foliar herbicides. Because Chinese privet is capable of year-round photosynthesis (Morris and others 2002) and herbicide transport generally follows the

Karen Vaughn, Associate Division Director, American Conservation Experience, U.S. Fish and Wildlife Service Division, Asheville, NC 28803
Lauren S. Pile Knapp, Research Ecologist, U.S. Department of Agriculture, Forest Service, Northern Research Station, Columbia, MO 65240
G. Geoff Wang, Professor, Forestry and Environmental Conservation, Clemson University, Clemson, SC 29634

direction of carbohydrates, which are moving toward the roots in winter, herbicide applications during this period may be quite effective (Harrington and Miller 2005, Miller 2005). However, low winter temperatures can slow herbicide uptake and transport (Frey and others 2007). Although foliar herbicide applications are less labor intensive than other methods (e.g., cut-stump, mechanical removal), broadcast herbicide applications are more likely to result in undesirable impacts on nontarget species, which suggests that applying foliar herbicides in winter may reduce undesirable nontarget impacts because most plants are dormant and leafless.

In field studies, foliar glyphosate applications demonstrate a high level of efficacy for the control of Chinese privet. Miller (2005) tested eight common herbicides and found glyphosate to be the most effective for growing-season Chinese privet control, followed by imazapyr and metsulfuron methyl (MSM). Harrington and Miller (2005) found that foliar glyphosate treatments at rates ranging from 1.7 to 6.7 kg acid equivalent per hectare provided up to 100 percent control in both fall (October and December) and spring (April) treatments. Ahuja (2003) also achieved 100 percent control with a December application of glyphosate. Glyphosate binds tightly and rapidly to soil particles, minimizing leaching or residual impacts to plants (Vereecken 2005) and several glyphosate formulations are approved for use near surface waters. However, glyphosate is highly nonselective and will kill or damage most plant types (Franz and others 1997). MSM has also been shown to be highly effective for controlling Chinese privet (Evans and others 2008, Miller 2005, Nespeca and Kemp 2006), although it has been less extensively tested in the dormant season. Several plants, including many grasses, have some resistance to MSM, which may be beneficial in broadcast applications. However, it does not bind as tightly to the soil as glyphosate and cannot be used near surface waters (Getsinger and others 2011). It does have the potential to cause residual impacts on some nontarget species, including canopy trees (Evans and others 2008). Subsequently, contrasting these herbicides for dormant-season applications of Chinese privet control and their resulting impact on nontarget flora is important for evaluating management trade-offs.

Backpack sprayers are commonly used for foliar herbicide applications, but backpack-mounted mistblowers may be able to provide similar control using a lower volume of herbicide (Nespeca and Kemp 2006). Additionally, mistblowers produce smaller, more regulated droplets than backpack sprayers, resulting in greater penetration of dense canopies and greater leaf interception (Devine and others 1992). However, mistblowers are heavy and require the transport of fuel and their smaller droplet size is more likely to result in herbicide drift. Backpack sprayers have the advantage of being lighter and smaller but are harder to maintain constant pressure and their larger droplets are more likely to fall through foliage and encounter ground flora (Devine and others 1992). Further, evaluating the outcome of foliar herbicide treatment by applicator type may improve the control of dense and large Chinese privet infestations while reducing nontarget impacts.

This study sought to answer the following management questions: 1) Can we use dormant-season foliar herbicide to control Chinese privet with less impact on native flora? 2) Could a mistblower be more effective for controlling large and dense Chinese privet infestations than a traditional backpack sprayer? 3) Would MSM provide better results, both in terms of Chinese privet control and reduced nontarget impacts, than glyphosate? To address these questions, we developed the following study objectives: 1) compare bottomland hardwood forest sites invaded by Chinese privet to similar but less invaded sites to establish a reference condition, 2) compare treatment combinations (application type plus herbicide formula) for the control of Chinese privet, and 3) determine which treatment combination has the least impact to native plants.

MATERIALS AND METHODS

Congaree National Park (33°47’59”N, 80°47’18”W) is located in the upper coastal plain of South Carolina, about 32 km southeast of Columbia. Mean monthly temperatures range from 7.7 °C in winter to 26.9 °C in summer, with significant year-to-year variation (Doyle 2009). The mean monthly precipitation ranges from 6.0 cm in winter to 13.8 cm in summer. The park is situated in the floodplain of the Congaree River, and it

experiences flooding an average of 10 times per year with an average of one flood per year that covers the majority of the park (Doyle 2009, Patterson and others 1985). The Congaree floodplain is a highly productive system, with trees exhibiting high growth rates and reaching a very large size (Doyle 2009). Much of the area has a history of agriculture and logging, with some salvage logging occurring as recently as 1990 following Hurricane Hugo.

Our study sites were located between 40 and 215 m from the Congaree River (fig. 1), primarily within the natural levee zone of the floodplain, becoming inundated only during the highest flooding events and usually for only a few days (Doyle 2009). Study sites were characterized by floodplain soils, primarily Congaree loam, with a small amount of Toccoa loam (Soil Survey Staff 2013). The forest is characterized by bottomland hardwood vegetation. Common tree species were boxelder (*Acer negundo*), sugarberry (*Celtis laevigata*), elms (*Ulmus* spp.), sweetgum (*Liquidambar styraciflua*), and pawpaw (*Asimina triloba*). Common understory species included false nettle (*Boehmeria cylindrica*), sedges (*Carex* spp.), Japanese stiltgrass (*Microstegium vimineum*), and lianas such as grapes (*Vitis* spp.), crossvine (*Bignonia capreolata*), and poison ivy (*Toxicodendron radicans*). Chinese privet distribution was variable within study sites. Some areas had dense Chinese privet thickets with a closed canopy and little understory vegetation, while others were more widely spaced, allowing for abundant growth of herbaceous species.

This study was designed as a randomized complete block experiment, with five blocks located in large Chinese privet populations (fig. 1). Within each block, seven 20-m by 20-m vegetation plots were installed. An additional plot within each block was established outside (uninvaded)

Figure 1—Study location and site locations for the Chinese privet project located within (above) the Congaree National Park with sites located in (below) the floodplain forest of the Congaree River in South Carolina.

of the Chinese privet infestation in an area similar to the invaded plots. Plots were quartered into four 10-m by 10-m modules. Corners were permanently marked with steel conduit stakes. All data were recorded on a per-module basis and divided into an 'herbaceous stratum' (height <50 cm) and a 'shrub stratum' (heights >50 cm, up to the maximum height of Chinese privet shrubs in the module). A visual estimate of cover was made for each species, including Chinese privet, by stratum. Woody stems >50 cm in height and rooted within the plot were tallied by size class and species. Any stem that branched from the main stem below 50 cm was considered an individual. Stems below breast height (137 cm) were tallied by height class (50–100 cm and >100–137 cm) and stems above breast height were tallied by diameter at breast height (d.b.h.) class (0–1, >1–2.5, >2.5–5, >5–10 cm, and 5 cm increments thereafter). For trees >40 cm d.b.h., individual measurements were recorded. Invaded plots were surveyed in the summer of 2011 prior to treatment. Posttreatment invaded plots and uninvaded plots were surveyed in the summer of 2012, 4 to 6 months following herbicide treatment.

Objective 1: Establish Reference Condition

To characterize differences between invaded and uninvaded communities, we compared plot averages from the midpoint of each cover class by species. We used total plot stem densities to compare woody abundance by species. We calculated species richness as the average number of species per module. Diversity was calculated with Simpson's Index ($D = \Sigma p_i^2$ where p_i is the proportion of total cover made up by species i). Simpson's Index represents the probability that two randomly selected individuals will be of the same species, and the reciprocal ($1/D$) was used here so that the index increased with increasing diversity (Magurran 2021). Herbaceous-stratum species were grouped by growth form (annual herbs, perennial herbs, trees and shrubs, vines, ferns, sedges, and native grasses). Japanese stiltgrass was analyzed individually. Canopy tree species were analyzed as a group to investigate impacts on regeneration. This group included boxelder, bitternut hickory (*Carya cordiformis*), sugarberry, green ash (*Fraxinus pennsylvanica*), sweetgum, sycamore (*Platanus occidentalis*), laurel oak (*Quercus laurifolia*), cherrybark oak (*Quercus pagoda*), and elms. Analysis of variance (ANOVA) was used to compare structure of uninvaded plots to pretreatment data from invaded plots. Site was designated as a random effect and invasion status as a fixed effect. A correlation analysis was used to investigate relationships between Chinese privet abundance and the abundance and diversity of nontarget plants.

Objective 2: Compare Treatment Outcomes for Chinese Privet

Within each block, the seven plots were randomly assigned an herbicide by applicator treatment combination or as an untreated control. Herbicide treatments were applied on January 11, 2012, by an experienced contract crew. Mistblowers were Stihl SR model 450, which is a backpack-mounted unit with a 14-L capacity. Their specified range is 14.5 horizontal meters and 13.0 vertical meters, with a droplet size generally ranging from 60 to 130 µm (Jessop and Bateman 2007). Backpack sprayers used were SP Systems Yard Tender model 101 with a 20-L capacity, which was pressurized by a hand-pump. Spray range and droplet size varied according to pressure. Glyphosate was mixed in a 5 percent volume to volume (v/v) solution, 49.53 g/L of Rodeo® liquid concentrate [53.8 percent active ingredient (a.i.)]. The MSM solution contained 0.47 g/L of AmTide MSM 60DF® powder concentrate (60 percent a.i.). The combination treatment contained 49.53 g glyphosate and 0.47 g MSM/3.78 L (1 gallon). The water conditioner Choice (Loveland Industries, Inc., Loveland, CO, 0.25 percent by volume), and the surfactant Rebound (Red River Specialties, Inc., Shreveport, LA, 0.5 percent v/v) were added to all spray mixes. The volume of spray applied to each plot varied depending on the density of Chinese privet and privet was sprayed to wet. Spray volumes ranged from 5.68 to 9.46 L per 400 m^2 plot, or approximately 142 to 236 L/ha. During the 2012 surveys, Chinese privet stems were assessed as either dead or alive. Two-way ANOVAs were used to analyze treatment effectiveness based on a factorial model with site, applicator, herbicide, and herbicide x applicator interaction as model effects. Treatment effectiveness was

quantified as a percentage control to account for differences in pretreatment privet densities. ANOVAs were run for percentage control of privet basal area, canopy cover, and stem density of small (50–137 cm height), medium (0–5 cm d.b.h.), and large stems (>5–15 cm d.b.h.). Due to treatment irregularities, nine modules and one whole plot were excluded from the analysis (Vaughn 2013).

Objective 3: Evaluate Treatment Outcomes for Native Plants

For diversity (richness and Simpson's Index) and abundance variables, pretreatment data were subtracted from posttreatment data to analyze the change. Herbaceous-stratum species were grouped by growth form (annual herbs, perennial herbs, trees and shrubs, vines, ferns, sedges, native grasses, and invasive plants) and plants that remain green in the winter (wintergreen) to analyze whether herbicide treatments disproportionately affected a particular plant type, based on the sum of the change in percentage coverage for all species in each group. Diversity and abundance variables were analyzed with ANOVAs with site and treatment as model effects. Linear contrasts were applied to test for effects of applicator, herbicide, and applicator x herbicide interaction.

RESULTS

Chinese Privet Invasion Alters Forest Structure and Composition

Regeneration of native trees differed between invaded and uninvaded plots. Generally, uninvaded plots had greater densities of tree reproduction than invaded plots, with significantly more stems in the >1- to 2.5-cm- and >2.5- to 5cm-d.b.h. size classes (table 1). The density of shrubs (excluding Chinese privet) and reproduction of understory trees was greater in uninvaded plots than in invaded plots, except for the largest size class (>5–10 cm d.b.h.). However, when Chinese privet was included, stem densities were greatest in invaded plots except for the >1- to 2.5-cm d.b.h. class. Herbaceous coverage, sedges, and the cover of Japanese stiltgrass were greater in the uninvaded plots (table 2). As the basal area (BA) and coverage of Chinese privet increased, herbaceous cover (BA: r = -0.41, P <0.01; cover: r = -0.34, P = 0.03), species richness (BA: r = -0.40, P <0.01; cover: r = -0.48, P <0.01), and the density of overstory tree reproduction (>1-5 cm d.b.h.) (BA: r = -0.34, P = 0.03; cover: r = -0.41, P <0.01) decreased. Simpson's Index was not significantly correlated with Chinese privet abundance.

Effectiveness of Privet Control

The interaction of herbicide by applicator was not significant for the control of Chinese privet, except for small Chinese privet stems (table 3). There was no difference detected for the control of Chinese privet between glyphosate or MSM for other variables. Conversely, the control of Chinese privet was affected by applicator type, with mistblowers being more effective at reducing Chinese privet abundance than backpack sprayers (fig. 2). Control plots showed no change or a slight increase in Chinese privet density, indicating that the decrease found in treatment plots was directly related to herbicide application.

Community Impacts from Chinese Privet Control

For most of the variables tested, individual treatments did not differ from the control. However, the backpack sprayer-glyphosate treatment showed a significant decrease in cover for sedges and a concomitant increase in Japanese stiltgrass (fig. 3). The backpack-MSM and backpack-glyphosate treatments showed a significant decrease in the coverage of tree and shrub seedlings (fig. 3). Linear contrasts did detect some overall herbicide and applicator effects. Specifically, backpack sprayer plots had larger decreases in cover than mistblower plots for trees and shrubs (P = 0.05). Glyphosate plots had larger decreases in sedge cover than MSM (P = 0.01) or combination plots (P = 0.03). Glyphosate (P = 0.02) plots showed greater decreases in the cover of wintergreen species than MSM plots. There were nonsignificant treatment effects on species richness, annual herb cover, perennial herb cover, native grass cover, nonnative cover,

Table 1—Density (stems/400 m^2) of overstory tree reproduction, shrub stems (Chinese privet excluded), understory tree reproduction, and all woody stems by size class for Chinese privet invaded and uninvaded plots at Congaree National Park

	Size class	Invaded	Uninvaded	*P*
Overstory tree reproduction[1]	50 cm height, 1.0 cm d.b.h.	0.3	0.2	0.8846
	>1.0–2.5 cm d.b.h.	0.7	3.4	0.0052
	>2.5–5.0 cm d.b.h.	1.3	4.8	0.0060
	>5.0–10.0 cm d.b.h.	3.1	3.8	0.5931
	Total	5.4	12.2	0.0218
Shrub (Chinese privet excluded) stems and understory tree reproduction[2]	50–100 cm height	17.7	38.8	0.0153
	>100–137 cm height	6.5	14.8	0.0147
	0–1.0 cm d.b.h.	8.7	20.2	0.0007
	>1.0–2.5 cm d.b.h.	9.9	31.4	<0.0001
	>2.5–5.0 cm d.b.h.	11.8	23.4	0.0007
	>5.0–10.0 cm d.b.h.	7.4	7.0	0.8659
	Total	62.0	135.6	0.0005
All woody stems (Chinese privet included)	50–100 cm height	122.7	45.2	0.0125
	>100–137 cm height	59.0	15.0	0.0378
	0–1.0 cm d.b.h.	85.9	20.8	0.0001
	>1.0–2.5 cm d.b.h.	37.4	35.0	0.6680
	>2.5–5.0 cm d.b.h.	48.8	28.4	0.0063
	>5.0–10.0 cm d.b.h.	26.1	10.8	0.0004
	Total	380.0	155.2	0.0005

[1] Overstory trees: bitternut hickory, sugarberry, green ash, sweetgum, sycamore, laurel oak, cherrybark oak, elms.
[2] Understory trees: all other species, excluding those identified as overstory species.

Table 2—Percent coverage by plant growth form for Chinese privet invaded and uninvaded plots at Congaree National Park

Plant growth form	Invaded	Uninvaded	*P*
	~ ~ *percent coverage* ~ ~		
Canopy trees	1.78	1.00	0.2565
Understory trees and shrubs	3.35	2.83	0.5783
Herbaceous cover	18.85	42.20	0.0154
Perennial herbs	1.01	0.84	0.6044
Annual herbs	0.25	0.13	0.3559
Vines (<50 cm)	3.20	1.94	0.3067
Vines (≥50 cm)	1.78	0.13	0.0631
Ferns	0.19	0.23	0.7759
Native grasses	0.61	0.73	0.6873
Sedges	6.58	15.60	0.0267
Japanese stiltgrass	3.41	22.20	0.0004

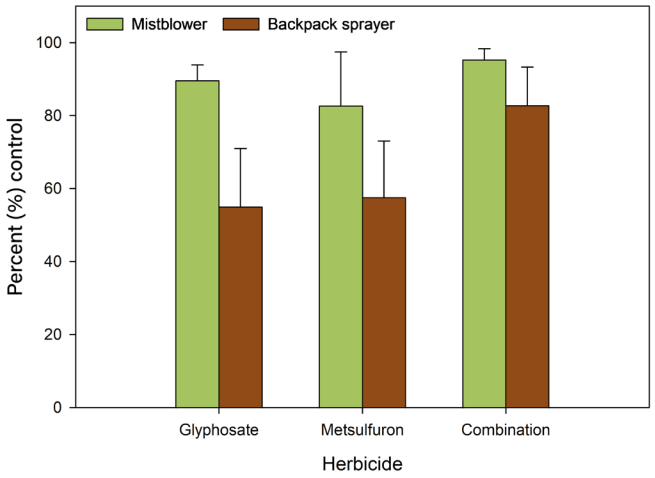

Figure 2—Percent control (+1 standard error) of Chinese privet stems by herbicide and applicator type at Congaree National Park.

Table 3—Model effects for Chinese privet control by herbicide, applicator, and their interaction and average percent control by applicator type for Chinese privet

Variable		*P*			Percent control	
		Applicator	Herbicide	Applicator x herbicide	Backpack	Mistblower
Basal area		0.0108	0.1640	0.5443	65.7	89.2
Stems[1] per 100 m²	**Small**	0.1551	0.8198	0.0254	90.8	94.5
	Medium	0.0213	0.4009	0.7394	84.6	92.5
	Large	0.0105	0.1542	0.4975	63.4	89.1
Canopy coverage		0.0321	0.6329	0.5105	90.3	98.0
Green stems[2]		0.0425	0.2092	0.3381	48.9	32.1

[1] Stem size: Small = 50–137 cm height; Medium = 0–5.0 cm d.b.h.; Large = >5.0–15.0 cm d.b.h.

[2] Values represent percentage of pretreatment basal area that was defoliated but still had live stem tissue following herbicide application. A higher value may indicate a greater chance of privet recovery.

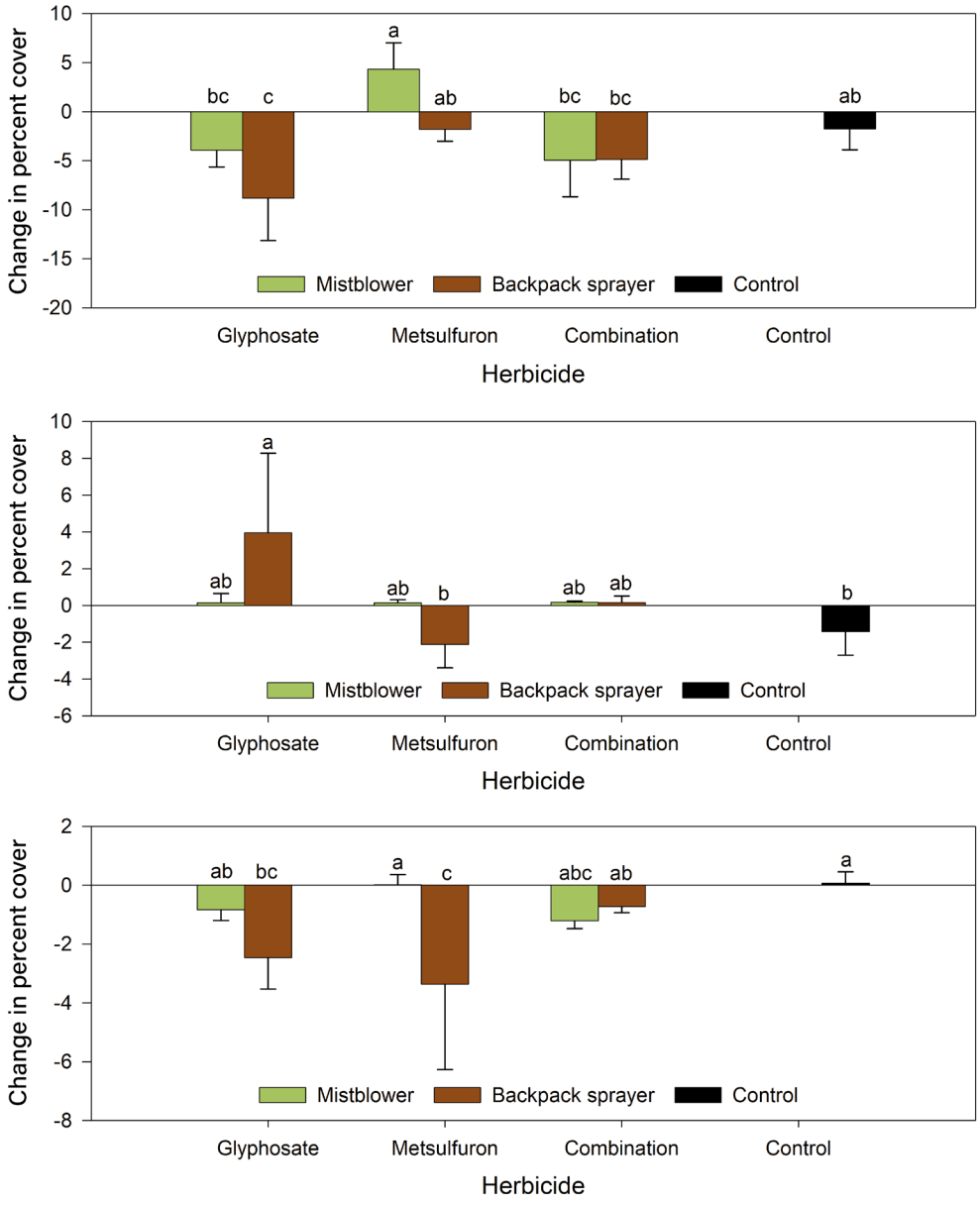

Figure 3—Change in percentage of cover (+ 1 standard error) following treatment for (top) sedges, (middle) Japanese stiltgrass, and (bottom) trees and shrubs. Treatments within graphs that do not share the same letter were significantly different at α = 0.05.

shrub-stratum vine cover, and shrub-stratum density when compared to the control.

DISCUSSION

Dense Chinese privet infestations change forest structure by increasing the net total density of the shrub layer. This increased shrub layer is associated with reduced canopy tree reproduction, potentially having long-term influences on forest structure, supporting findings that Chinese privet can lower the growth and survival of tree seedlings (Greene and Blossey 2014, Turner and others 2022). Chinese privet-invaded areas had lower sedge cover, but dense Chinese privet infestations may also inhibit Japanese stiltgrass. Species richness, diversity, and the cover of most growth forms did not differ between invaded and uninvaded plots. However, many of the invaded plots did not have closed canopies, and correlation analysis showed that richness, herbaceous cover, and canopy tree reproduction decreased as Chinese privet density and cover increased. Observational studies covering large geographic areas and a variety of habitat types have found similar results (Greene and Blossey 2014, Loewenstein and Loewenstein 2005, Merriam 2003). Chinese privet removal experiments have shown increases in cover (Hanula and others 2009) and growth (Osland and others 2009) of understory plants, and increases in density of woody stems (Merriam and Feil 2002). Some removal studies also show an increase in herbaceous diversity (Merriam and Feil 2002), although others do not (Hanula and others 2009, Vidra and others 2007). While covarying factors must be considered, the weight of evidence suggests that direct competition with Chinese privet is a factor in the reduction of native plant abundance in invaded sites. Further, the potential long-term impacts on canopy regeneration and recruitment lend support to the justification of Chinese privet control efforts, although the effects of other sources of disturbance should also be considered.

Herbicide type was not found to be a significant factor in this study. Glyphosate and MSM were both effective, as seen in previous studies (Harrington and Miller 2005, Miller 2005). In the absence of significant differences in effectiveness between herbicides, glyphosate adsorbs tightly to soils, limiting the chance of offsite transport (Vereecken 2005). Metsulfuron, however, is soil-active and while it may reduce impacts on sedges and rivercane, it has an increased risk to canopy trees. While this may not be an issue when trees are dormant, warm winters could cause trees to break dormancy sooner than expected. Based on observation, sections of the Chinese privet canopy were generally either completely defoliated or appeared completely healthy with no signs of herbicide damage. Damage from both glyphosate and MSM can appear as yellow or brown spots on leaves, or leaf or vein discoloration (Obrigawitch and others 1998). The lack of visible damage or deformity suggests that the remaining live stems were not contacted by herbicide spray, and that differences between treatment combinations were primarily a reflection of the spray coverage from the applicator and not the herbicide type.

Mistblowers provided more complete control of Chinese privet than did backpack sprayers. They achieved higher percentage control of shrub-stratum Chinese privet measured as density, basal area, and cover. Mistblower plots also had a lower percentage of stems remaining green after treatment. The height of spray appears to be a major factor in the superior performance of mistblowers for controlling Chinese privet. Generally, mistblowers overall showed fewer nontarget impacts than backpack sprayers. Backpack sprayers had greater reductions in total herbaceous cover, trees and shrubs, vines, and wintergreen species. This may be related to the difference in the size of spray droplets produced by these applicator types. The larger droplets produced by backpack sprayers are more likely to fall through the foliage of the Chinese privet canopy and contact lower layers of sensitive vegetation. Logistical issues, such as weight and maneuverability of equipment, complications related to use of motorized equipment in a wilderness, cost per acre, and public opinion could influence decisions about applicator type. The difference in spray height between mistblowers and backpack sprayers could potentially be overcome by using higher pressures or extension wands with the backpack sprayers, but impacts to sedges and other herbaceous-stratum plants may still be anticipated.

Several factors must be weighed in planning invasive plant treatments, including the size and density of the targeted invader and the potential for nontarget impacts based on the control method used. Many Chinese privet stems are taller than the practical field range of either backpack sprayers or mistblowers. Further, no single combination of applicator and herbicide was clearly superior at minimizing impacts on native plant communities and overall, negative impacts on native plants were limited. However, the backpack-glyphosate treatment reduced the coverage of sedges and increased the coverage of Japanese stiltgrass. The presence of recently killed sedge clumps in treated plots suggested that Japanese stiltgrass readily invaded following the disturbance caused by management. Conversely, tree and shrub cover significantly declined in the backpack-MSM treatment. The change in tree and shrub cover was heavily influenced by two plots with unusually high cover of maple (*Acer* spp.) seedlings. Winter vegetation surveys of treatment areas would help to prepare and mitigate potential nontarget impacts when using broadcast herbicide applications. Canebrakes, dense sedge populations, rare plants, or potentially sensitive or valued wintergreen species could be detected during the preplanning process. In general, impacts to nontarget plants are expected to be relatively small and the long-term benefits to canopy tree regeneration and herbaceous plant cover are expected to outweigh the short-term negative impacts of herbicide application.

CONCLUSIONS

Bottomland forests invaded by Chinese privet have midstory densities that are greater than uninvaded reference conditions. While this increase in the shrub layer may diminish the establishment and growth of Japanese stiltgrass, it also impedes native tree regeneration. Our study suggests that both glyphosate and MSM are effective for controlling Chinese privet in the dormant season. However, glyphosate may also impact native sedge species resulting in opportunities for secondary invasion by Japanese stiltgrass. Mistblowers may be more effective at reaching the tops of tall invasive shrubs and through their smaller droplet size result in greater foliar interception and subsequent mortality of targeted plants and reduced risk to nontarget plants by fewer droplets falling through the invasive canopy.

ACKNOWLEDGMENTS

We thank the field assistants on the project: Elaine Donithan, CB Genrich, Patrick Ma, and Ed Vaughn, who graciously volunteered their time and withstood the many hardships of the Congaree backcountry in summer. We would also like to thank Terri Hogan, Theresa Thom, John Torrence, David Shelley, and the entire staff of Congaree National Park. Additionally, we would like to thank John Hains, William Conner, and Gavin Blosser for the use of their boats and Patrick McMillan and Dixie Damrel for their assistance with plant identification. Funding for this project was provided by the National Park Service. We would also like to sincerely thank Andy Ezell and Mohammad Bataineh for their thoughtful comments on the draft manuscript.

REFERENCES

Ahuja, G. 2003. Chemical control of Chinese privet (*Ligusrtum* [sic] *sinense*). M.S. thesis. Athens, GA: University of Georgia. 47 p.

Devine, M.; Duke, S.O.; Fedtke, C. 1992. Physiology of herbicide action. Englewood Cliffs, NJ: PTR Prentice Hall. 441 p.

Doyle, T.W. 2009. Modeling flood plain hydrology and forest productivity of Congaree Swamp, South Carolina. Scientific Investigations Report 2009-5130. Denver, CO: U.S. Geological Survey. 46 p. https://doi.org/10.3133/sir20095130.

Evans, C.; Bargeron, C.; Moorehead, D. [and others]. 2008. Invasive plants of Georgia's forests: Identification and control. BW-2008-01. Athens, GA: Georgia Forestry Commission, University of Georgia, and U.S. Department of Agriculture Forest Service. 10 p.

Franz, J.E.; Mao, M.K.; Sikorski, J.A. 1997. Glyphosate: a unique global herbicide. Monograph 189. Washington, DC: American Chemical Society. 653 p.

Frey, M.N.; Herms, C.P.; Cardina, J. 2007. Cold weather application of glyphosate for garlic mustard (*Alliaria petiolata*) control. Weed Technology. 21: 656–660. https://doi.org/10.1614/WT-06-153.1.

Fridley, J.D. 2012. Extended leaf phenology and the autumn niche in deciduous forest invasions. Nature. 485: 359–362. https://doi.org/10.1038/nature11056.

Getsinger, K.D.; Netherland, M.D.; Stubbs, D.R. 2011. Aquatic herbicide registration new model. In: Leslie, A.R.; Westbrooks, R.G., eds. Invasive plant management issues and challenges in the United States: 2011 overview. ACS Symposium Series vol. 1073. Washington, DC: American Chemical Society: 45–50. https://doi.org/10.1021/bk-2011-1073.ch005.

Greene, B.; Blossey, B. 2014. Patterns of privet: urbanizing watersheds, invasive *Ligustrum sinense,* and performance of native plant species in piedmont floodplain forests. Ecosystems. 17: 990–1001. https://doi.org/10.1007/s10021-014-9774-4.

Hanula, J.L.; Horn, S.; Taylor, J.W. 2009. Chinese privet (*Ligustrum sinense*) removal and its effect on native plant communities of riparian forests. Invasive Plant Science and Management. 2: 292–300. https://doi.org/10.1614/IPSM-09-028.1.

Harrington, T.B.; Miller, J.H. 2005. Effects of application rate, timing, and formulation of glyphosate and triclopyr on control of Chinese privet (*Ligustrum sinense*). Weed Technology. 19: 47–54. https://doi.org/10.1614/WT-03-220R2.

Jessop, N.; Bateman, R. 2007. Optimising the use of motorised knapsack mistblowers for control of insect pests of cocoa. International Pest Control. 49: 86–88.

Loewenstein, N.J.; Loewenstein, E.F. 2005. Non-native plants in the understory of riparian forests across a land use gradient in the Southeast. Urban Ecosystems. 8: 79–91. https://doi.org/10.1007/s11252-005-1420-7.

Magurran, A.E. 2021. Measuring biological diversity. Current Biology. 31: R1174–R1177. https://doi.org/10.1016/j.cub.2021.07.049.

Merriam, R.W. 2003. The abundance, distribution and edge associations of six non-indigenous, harmful plants across North Carolina. Journal of the Torrey Botanical Society. 130: 283–291. https://doi.org/10.2307/3557546.

Merriam, R.W.; Feil, E. 2002. The potential impact of an introduced shrub on native plant diversity and forest regeneration. Biological Invasions. 4: 369–373. https://doi.org/10.1023/A:1023668101805.

Miller, J.H. 2005. Chinese privet control with herbicide foliar sprays. Wildland Weeds. 8(3): 5–7.

Morris, L.L.; Walck, J.L.; Hidayati, S.N. 2002. Growth and reproduction of the invasive *Ligustrum sinense* and native *Forestiera ligustrina* (Oleaceae): implications for the invasion and persistence of a nonnative shrub. International Journal of Plant Sciences. 163: 1001–1010. https://doi.org/10.1086/342632.

Munger, G.T. 2003. *Ligustrum* spp. In: Fire Effects Information System. Missoula, MT: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences Laboratory.

Nespeca, M.C.; Kemp, C. 2006. Chinese privet control: economic viability and efficacy of dormant season foliar low-volume herbicide treatments. The Nature Conservancy.

Obrigawitch, T.T.; Cook, G.; Wetherington, J. 1998. Assessment of effects on non-target plants from sulfonylurea herbicides using field approaches. Pesticide Science. 52: 199–217. https://doi.org/10.1002/(SICI)1096-9063(199803)52:3<199::AID-PS696>3.0.CO;2-Yopen_in_newISSN0.

Osland, M.J.; Pahl, J.W.; Richardson, C.J. 2009. Native bamboo [*Arundinaria gigantea* (Walter) Muhl., Poaceae] establishment and growth after the removal of an invasive non-native shrub (*Ligustrum sinense* Lour., Oleaceae): implications for restoration. Castanea. 74: 247–258. https://doi.org/10.2179/08-036R2.1.

Patterson, G.G.; Speiran, G.K.; Whetstone, B.H. 1985. Hydrology and its effects on distribution of vegetation in Congaree Swamp National Monument, South Carolina. Water-Resources Investigations Report 85-4256. Columbia, SC: U.S. Department of the Interior, Geological Survey. 31 p. https://doi.org/10.3133/wri854256.

Soil Survey Staff. 2013. Web soil survey. U.S. Department of Agriculture, Natural Resources Conservation Service. http://websoilsurvey.sc.egov.usda.gov/. [Date accessed: January 10, 2013].

Turner, T.N.; Dean, T.J.; Kuehny, J.S. 2022. Long-term suppression of hardwood regeneration by Chinese privet (*Ligustrum sinense*). Frontiers in Forests and Global Change. 4: 725582. 7 p. https://doi.org/10.3389/ffgc.2021.725582.

Vaughn, K.H. 2013. Evaluation of dormant-season herbicide treatment methods for Chinese privet at Congaree National Park. M.S. thesis. Clemson, SC: Clemson University. 101 p.

Vereecken, H. 2005. Mobility and leaching of glyphosate: a review. Pest Management Science. 61: 1139–1151. https://doi.org/10.1002/ps.1122.

Vidra, R.L.; Shear, T.H.; Stucky, J.M. 2007. Effects of vegetation removal on native understory recovery in an exotic-rich urban forest. Journal of the Torrey Botanical Society. 134: 410–419. https://doi.org/10.3159/1095-5674(2007)134[410:EOVRON]2.0.CO;2.

Wilcox, J.; Beck, C.W. 2007. Effects of *Ligustrum sinense* Lour. (Chinese privet) on abundance and diversity of songbirds and native plants in a southeastern nature preserve. Southeastern Naturalist. 6: 535–550.

Management and Silvics

PRIVATE FOREST LANDOWNERS' PERSPECTIVES ON FERAL HOG MANAGEMENT AND CONTROL IN ARKANSAS, LOUISIANA, AND EAST TEXAS

Alyssa Mineau, Nana Tian, Jianbang Gan, and Gordon Holley

ABSTRACT

Feral hogs (*Sus scrofa*) are a rapidly increasingly nonnative invasive species that has spread widely across the Southern United States, impacting water quality and causing severe damage to tree regeneration. As a result, controlling feral hogs is an urgent issue for many forest landowners and silviculturists. To provide land managers in the Western Gulf Region (WGR) with a better sense of the extent of feral hog damage and landowner perspectives on controlling this invasive species, we sent 4,500 mail surveys in the summer of 2021 to randomly selected landowners in Arkansas, Louisiana, and east Texas. A subsample of private forest landowners was selected for this study to explore their perceptions of feral hogs in this region. Results indicated that most private forest landowners are familiar with feral hogs, especially Texas forest landowners (>90 percent reported familiar), and generally have overwhelmingly negative attitudes toward them. Regarding the control and management methods, trapping and shooting are the most acceptable and effective methods among the private forest landowners in the WGR. These findings will help silviculturists better understand private forest landowners' perspectives on feral hogs and the most acceptable control and management methods. For example, exploring suitable silvicultural practices (such as trapping the planting area, substituting appropriate tree species, and more) to avoid feral hog damage is not only a key topic among silviculturists but also among private forest landowners. Understanding forest landowners' concerns and motivations for feral hog control should also lead to the development and deployment of more effective management tools, control policies, and educational programs for feral hogs in the study area.

INTRODUCTION

Feral hogs (*Sus scrofa*) are an invasive species that have spread widely since they were first introduced to North America by European colonists (Conover and Vail 2007). Prolific and quick to mature, the difficult-to-eradicate feral hogs have spread rapidly to 47 States, especially in the last 3–4 decades (Mayer 2014, USDA APHIS 2020). Despite their recreational benefits for some hunters in certain areas (Centner and Shuman 2015), feral hogs are a particularly destructive invasive species in the United States, causing agricultural, forestry, and other environmental losses estimated at around $1.5 billion per year (Mississippi State University 2013).

Feral hogs cause severe and widespread damage to row crops, livestock, water quality, forest regeneration, and infrastructure (Harper and others 2014). Feral hogs feed on various plants and animals across many geographic regions (Barrios-Garcia and Ballari 2012), compete for food with livestock and native wildlife, and carry many infectious diseases that could affect other animals and humans. Though rare, there are even some reports of feral hogs attacking farmers, golfers, hikers, and picnickers (USDA APHIS 2020). Feral hogs can cause severe damage to natural habitats by wallowing, removal of tree bark by "tusking" (Mayer 2009), and foraging.

Alyssa Mineau, Graduate Student, College of Forestry, Agriculture and Natural Resources, University of Arkansas at Monticello, Monticello, AR 71656

Nana Tian, Assistant Professor, Arkansas Forest Resources Center, College of Forestry, Agriculture and Natural Resources, University of Arkansas at Monticello, Monticello, AR 71656

Jianbang Gan, Professor, Department of Ecology and Conservation Biology, Texas A&M University, College Station, TX 77843

Gordon Holley, Professor, Department of Agricultural Sciences and Forestry, Louisiana Tech University, Ruston, LA 71272

Indeed, feral hog foraging can cause widespread tree regeneration failures, especially in recently established pines (*Pinus* spp.) and hardwoods (Campbell and others 2010). As their primary method of food acquisition, rooting is the most widespread and observed type of damage caused by feral hogs (Mayer 2009). Rooting can have disastrous effects on many different landscape types. In the Southern United States, feral hog depredation of newly planted tree seedlings has become a major concern, affecting pines (Fern and others 2020) and hardwood species such as cherrybark oak (*Quercus pagoda*), swamp chestnut oak (*Q. michauxii*), water hickory (*Carya aquatica*), and swamp tupelo (*Nyssa sylvatica* var. *biflora*) (Mayer and others 2000). When available, a staple of feral hog diet is acorns (Elston and Hewitt 2010), and their foraging reduces acorn availability for germination and forest regeneration (Loggins and others 2002).

Given the widespread and costly damages they inflict, controlling feral hogs is an urgent issue for farmers, rural landowners, wildlife managers, and silviculturists. However, feral hogs have a high reproductive capacity and early maturation, which makes their eradication difficult once established. In addition, feral hogs are intelligent and adaptable, which can diminish the efficacy of feral hog control measures. Furthermore, there is no single feral hog control measure that meets the objectives of every stakeholder. Thus, private landowners and natural resources managers use a variety of tools to aid in controlling feral hogs.

Control methods can be divided generally into two categories: nonlethal and lethal. Nonlethal controls are generally centered around reducing the harm feral hogs cause, while lethal controls are generally focused on population reduction or eradication. Nonlethal controls mainly include diversionary feeding, fencing, repellants, and translocation. Lethal controls are used more often than nonlethal controls for reducing feral hog damages. These methods include ground shooting (at daytime or night, with or without hunting dogs), trapping and euthanasia, snares, and aerial gunning.

Understanding the private forest landowners' perspectives on feral hogs and exploring the most acceptable control and management methods is the key to effectively managing the feral hog population. To that end, the specific objective of this study was to survey people in the Western Gulf Region (WGR; this area included Arkansas, Louisiana, and east Texas) to 1) understand private forest landowners' perspectives on feral hogs and 2) examine their support and acceptability for different management and control methods.

METHODS

We developed and conducted a mail-in survey in 2021 across the WGR. The survey questions centered around landowner attitudes towards feral hogs, as well as the control methods they supported or found effective. We generated survey questions based on similar surveys from nearby States (Capelnor and others 2017, Harper and others 2014, Mengak 2012, Tanger and others 2015) and created new questions with the guidance of subject matter experts from the WGR. We followed the Dillman Tailored Design method, including a survey cover page assuring landowners of the confidentiality of their responses and their participation was voluntary (Dillman 2000). The University of Arkansas at Monticello's Institutional Review Board (IRB# FNRf-01) reviewed and approved the survey questionnaire and protocol.

We mailed 4,500 surveys in three States: 2,000 surveys in Arkansas, 1,500 in Louisiana, and 1,000 in east Texas. The surveys were sent to landowners of ≥30 acres of land (including all types of land such as agricultural land, forest land, and more). For this study, we only used surveys from private forest landowners.

Descriptive statistics were used to summarize the responses and analysis of variance (ANOVA) was utilized to test for the differences in private forest landowners' perspectives on feral hogs and their control across the three States. All statistical results are reported at a significance level of 5 percent ($\alpha = 0.05$).

RESULTS

We removed 285 of the questionnaires in Arkansas, 175 in Louisiana, and 86 in east Texas due to ineligibility (e.g., undeliverable addresses, death). The number of usable questionnaires was 361 in Arkansas, 319 in Louisiana, and 226 in east Texas, yielding a response rate of 21 percent (Arkansas), 24 percent (Louisiana), and 25 percent (east Texas), and the overall adjusted response rate was 23 percent for all three States. The response rate for the focal group for this study (those who reported owning forest land) was 27 percent in Arkansas (n = 97), 28 percent in Louisiana (n = 90), and 14 percent in east Texas (n = 31).

When forest landowners were asked about their initial (presurvey) familiarity level with feral hogs, over half (54 percent) of Arkansas respondents indicated that they were very familiar with feral hogs, while it was as high as 70 percent in Louisiana; in east Texas, 94 percent of respondents indicated that they were very familiar with feral hogs.

No significant differences were found in the agreement level in the WGR for most feral hog statements (table 1). Respondents agreed that feral hogs should be eliminated wherever possible, that feral hogs have a negative impact on the natural environment, and that feral hogs harm the economy through damage to crops and livestock. Respondents felt obligated to eradicate feral hogs on their property and were willing to cooperate with neighbors to control feral hogs. Additionally, respondents disagreed with the statement "The problems related to feral hogs are exaggerated."

However, there was a significant difference in agreement level for the statement "Feral hogs are a nuisance and a threat" among the respondents in the three States (P <0.05). Respondents from Louisiana agreed more on this statement than other States, but Arkansas and Texas still "strongly agreed." Significant differences were also found in the responses to the statements "The laws to control feral hogs on private land in Arkansas/Louisiana/Texas are adequate" (P <0.05) as well as "Feral hogs should be managed to a level that does not damage the environment" (P <0.05). East Texas respondents were significantly more likely to disagree with the statement that current laws are adequate to control feral hogs than those in Arkansas and Louisiana. Respondents in Louisiana were more likely to support tighter control regulations than respondents in Arkansas and east Texas.

Support for various control methods varied among the States (table 2). States agreed on their support of capturing and killing feral hogs, as well as wanting technical assistance to control feral hogs. Louisiana respondents showed a statistically significant greater level of support

Table 1—Mean survey scores for respondents' perspectives on different feral hog statements in Arkansas, Louisiana, and east Texas

Survey statements	Arkansas	Louisiana	East Texas
Feral hogs are a nuisance and threat	1.23a	1.05b	1.26a
Feral hogs should be eliminated wherever possible	1.33	1.21	1.33
Feral hogs have a negative impact on the natural environment	1.42	1.25	1.39
Crops and livestock damage caused by feral hogs will harm the economy	1.49	1.26	1.29
The laws to control feral hogs on private land in AR/LA/TX are adequate	2.45a	2.78a	3.06b
I would like to learn more about feral hogs and control methods	2.03	1.99	2.06
The problems related to feral hogs are exaggerated in AR/LA/TX	4.07	4.14	4.07
Feral hogs should be managed to a level that does not damage the environment	1.87a	1.53a	2.24b
I support tighter feral hogs control regulations in AR/LA/TX	2.01a	1.90b	2.25a
I feel obligated to eradicate feral hogs on my property	1.46	1.43	1.69
I am willing to cooperate with my neighbors to control feral hogs together	1.38	1.26	1.38

Survey responses were 1 = strongly agree, 2 = somewhat agree, 3 = neither agree nor disagree, 4 = somewhat disagree, and 5 = strongly disagree.
Means in a row with different letters were statistically different at the 5-percent significance level.

Table 2—Mean survey scores for respondents' support for feral hog control methods

Feral hog control method	Arkansas	Louisiana	East Texas
Do not do anything	4.61	4.80	4.29
Capture and remove using trained dogs	2.26a	2.15a	2.93b
Targeted sharpshooting on the ground over bait sites	1.89a	1.68b	1.97a
Capture and relocate	4.12	4.18	4.24
Capture and kill	1.36	1.25	1.45
Aerial control using helicopters	1.87	1.89	2.28
Use legal toxicants (i.e., warfarin)	2.27a	2.09b	2.28a
Allow sales of feral hogs	3.37a	2.98b	2.31c
Educate people on how to prevent damage	1.70	1.62	1.83
Provide technical assistance for landowners/farmers to control feral hogs	1.47	1.41	1.62
Offer money rewards (bounties) to whoever controlling feral hogs	1.71	1.85	1.93
Provide subsidies (compensations) to landowners/farmers for feral hog damage	2.13	2.03	2.48

Survey responses were 1 = strongly agree, 2 = somewhat agree, 3 = neither agree nor disagree, 4 = somewhat disagree, and 5 = strongly disagree.
Means in a row with different letters were statistically different at the 5-percent significance level.

for targeted sharpshooting and the use of toxicants. All three States' respondents disagreed with the statement "Do not do anything" to manage feral hogs.

In terms of implementing feral hog control measures on private forest land across the WGR, the use and perceived efficacy of the methods varied widely (table 3). The most commonly used methods and the ones with the highest efficacy were trapping hogs and shooting hogs, with 65 percent (for trapping) and 52 percent (for shooting) of respondents reporting those methods to be effective or extremely effective. The method with the lowest perceived efficacy was "Use repellents (e.g., scents, pepper spray)" and "Put up fences"; specifically, over two-thirds of respondents reported those two methods to be either slightly or not at all effective.

Table 3—Mean survey score and percent in each response category for use and perceived efficacy of feral hog control measures

Feral hog control measures	n	Mean survey score	Extremely effective	Effective	Neutral	Slightly effective	Not at all effective
Shoot hogs	153	2.65	24.2	28.1	13.7	26.1	7.8
Trap hogs	127	2.45	23.6	40.9	11.0	15.7	8.7
Hunt with trained dogs	76	2.91	17.1	22.4	26.3	21.1	13.2
Hunt without dogs	79	3.06	13.9	25.3	20.3	21.5	19.0
Put up fences	49	4.06	2.0	8.2	22.4	16.3	51.0
Harass feral hogs (e.g., noise/lights)	53	3.96	7.5	7.5	20.8	9.4	54.7
Use repellents (e.g., scents, pepper spray)	42	4.38	0.0	0.0	26.2	9.5	64.3
Use legal toxicants (e.g., warfarin, sodium nitrite, etc.)	42	3.86	4.8	4.8	38.1	4.8	47.6
Officially lease to someone else to hunt for recreation and enjoyment	59	3.46	8.5	13.6	32.2	15.3	30.5

Survey responses were 1 = strongly agree, 2 = somewhat agree, 3 = neither agree nor disagree, 4 = somewhat disagree, and 5 = strongly disagree.
n = number of landowners who have used the control measure.

DISCUSSION

The results of this study indicated that most private forest landowners across the WGR were familiar with feral hogs, which differed from the familiarity levels reported by Adams and others (2005), where less than half of the respondents reported being knowledgeable about hog biology and its environmental impacts. A possible explanation is that private forest landowners' knowledge of feral hogs has increased with their physical encounters with hogs and better access to hog information through educational and outreach activities in the past 2 decades.

Our study found that across the WGR, forest landowners have overall negative perceptions of feral hogs, consistent with earlier studies (e.g., Adams and others 2005, Caplenor and others 2017, Harper 2014). Specifically, private forest landowners surveyed in this study agreed that feral hogs are a nuisance and threat and should be eliminated, which is in line with the findings from Adams and others (2005), Mengak (2012), and Caplenor and others (2017), who all reported that landowners have negative perceptions on feral hogs and agreed on controlling them.

Additionally, this study revealed the similarities among private forest landowner supportiveness of various feral hog control measures in three different States. For example, the majority of forest landowners in this region strongly support the option of "Capture and kill" and "Targeted sharpshooting on the ground over bait sites" as well as the education and incentive-based statements of "Educate people on how to prevent damage" and "Provide technical assistance for landowners/farmers to control feral hogs." This is comparable to Tennessee landowners in Watkins and others (2019), who reported that most respondents also supported capture and kill, targeted sharpshooting, education, and providing technical assistance. These similar attitudes towards management options can help justify large-scale management plans for feral hogs at the State and Federal levels, and can better guide management protocols to be cost-effective, accessible, and supported by stakeholders.

This study highlights the greatest- and least-utilized methods of feral hog control by WGR forest landowners. The most commonly used control measures were trapping hogs and shooting hogs—which were also reported to be the most effective methods of reducing feral hogs. This aligned with Mayer and others (2009) and Mapston (2007), who reported that shooting was commonly used specifically in regions that had smaller feral hog populations. However, this did not follow the findings of more recent literature that focuses on the efficacy of these control methods. While Mayer (2014) reported that hunting (shooting) only removed about 23 percent of the feral hog population annually, efficacy can vary. As an example, in Texas, aerial shooting is most effective in areas with little tree canopy and few human structures. Bodenchuk (2014) showed that aerial shooting is the lowest cost per animal removed for the Wildlife Services program in Texas.

Ground shooting can be effective when single or small groups can be dispatched but it is not effective for larger groups—sounders—where the survivors learn to avoid humans and become difficult to locate. Campbell and others (2010) reported that flying helicopters low to the ground and shooting a sounder did not disperse other pigs from their established home ranges. Shooting can be opportunistic whenever seen on the property, or more methodical, such as hunting with hog dogs or shooting with thermal scopes at known feral hog feeding sites at night.

Shooting tends to be less efficient than trapping because only a few individuals are removed. When trapping, box traps that catch individuals have proven less effective than a whole-sounder approach to reducing feral hogs and their damages. Trapping protocols or best practices are necessary, as feral hogs can develop trap shyness and learn to avoid traps and people (Massei and others 2011, West and others 2009), making them difficult to find and recapture or shoot.

Harassing feral hogs and putting up fences were the least utilized and the least effective methods at controlling feral hog numbers. Several factors

can drive the efficacy and utilization rates of feral hog traps, from trap design to location, or even bait choice. Trap design can be a potential factor, as traps with smaller gates are less expensive but also less effective than larger/wider gates (Higginbotham 2014). Furthermore, animal-activated gates cannot be selected for target species, but human-activated gates have high initial costs that make them impractical to landowners (Ditchkoff and Bodenchunk 2020). What forest landowners perceive to be effective at reducing feral hog presence may also depend on individual management goals; if a landowner wants feral hogs removed from their land entirely, they may find fences to be ineffective compared to the landowner that seeks crop protection from feral hog foraging.

CONCLUSIONS

This study investigated private forest landowners' perspectives of and experiences with feral hogs across three States in the WGR. Private forest landowners in the WGR are familiar with feral hogs and have overwhelmingly negative attitudes toward feral hogs, which suggested that private forest landowners realized the damages related to feral hogs to forest land and improving the current silvicultural practices might be needed in the future to avoid feral hog damage. Various methods have been implemented by landowners to control feral hogs, but only certain ones were considered to be most effective—specifically trapping and shooting. In addition, private forest landowners most strongly support lethal control measures for managing feral hogs and strongly support receiving education and technical assistance on managing feral hogs. This result indicated that private forest landowners will be interested in the effective practices and management options for controlling feral hogs in this region, which could be a future silvicultural research topic. For example, Fern and others (2020) found that changing tree species or providing logging slash piles as alternative foraging choices outside the tree planting area might be helpful to avoid feral hog damage. However, the practice of debris scattering following a clearcut can help feral hogs discover the seedlings, thus an alternative practice might be needed to protect the seedlings.

LITERATURE CITED

Adams, C.E.; Higginbotham, B.J.; Rollins, D. [and others]. 2005. Regional perspectives and opportunities for feral hog management in Texas. Wildlife Society Bulletin. 33: 1312–1320. https://doi.org/10.2193/0091-7648(2005)33[1312:RPAOFF]2.0.CO;2.

Barrios-Garcia, M.N.; Ballari, S.A. 2012. Impact of wild boar (*Sus scrofa*) in its introduced and native range: a review. Biological Invasions. 14: 2283–2300. https://doi.org/10.1007/s10530-012-0229-6.

Bodenchuk, M.J. 2014. Method-specific costs of feral swine removal in a large metapopulation: the Texas experience. In: Timm, R.M.; O'Brien, J.M., eds. Proceedings of the 26th vertebrate pest conference. Davis, CA: University of California: 269–271. https://doi.org/10.5070/V426110394.

Campbell, T.A.; Long, D.B.; Leland, B.R. 2010. Feral swine behavior relative to aerial gunning in southern Texas. Journal of Wildlife Management. 74: 337–341. https://doi.org/10.2193/2009-131.

Caplenor, C.A.; Poudyal, N.C.; Muller, L.I.; Yoest, C. 2017. Assessing landowners' attitudes toward wild hogs and support for control options. Journal of Environment Management. 201: 45–51. https://doi.org/10.1016/j.jenvman.2017.06.022.

Centner, T.J.; Shuman, R.M. 2015. Governmental provisions to manage and eradicate feral swine in areas of the United States. Ambio. 44: 121–130. https://doi.org/10.1007/s13280-014-0532-9.

Conover, M.R.; Vail, R.M. 2007. In the news. Human-Wildlife Conflicts. 1: 2–3. https://doi.org/10.26077/bhzf-4j22.

Dillman, D.A. 2000. Mail and internet surveys: the tailored design method. 2nd ed. New York, NY: John Wiley & Sons. 464 p.

Ditchkoff, S.S.; Bodenchuk, M.J. 2020. Management of wild pigs. In: VerCauteren, K.C; Beasley, J.C.; Ditchkoff, S.S. [and others], eds. Invasive wild pigs in North America: Ecology, impacts, and management. Boca Raton, FL: CRC Press: 102–110.

Elston, J.J.; Hewitt, D.G. 2010. Intake of mast by wildlife in Texas and the potential for competition with wild boars. Southwestern Naturalist. 55: 57–66. https://doi.org/10.1894/TAL-03.1.

Fern, M.P.; Armstrong, J.B.; Barlow, R.J.; Kush, J.S. 2020. Ecological factors influencing wild pig damage to planted pine and hardwood seedlings. Human-Wildlife Interactions. 14(2): 228–238. https://doi.org/10.26077/e63f-b82a.

Harper, E.E.; Miller, C.A.; Stephenson, A.L. [and others]. 2014. Landowner attitudes and perceived risks toward wild pigs on private lands in Illinois. INHS Technical Report 2014 (16). Human Dimensions Research Program Report HR-14-05. Champaign, IL: Illinois Natural History Survey. 40 p. https://www.ideals.illinois.edu/items/55892.

Higginbotham, B. 2014. The art and (some) science of trapping wild pigs: from traps to gates to triggers and more! In: Timm, R.M.; O'Brien, J.M., eds. Proceedings of the 26th vertebrate pest conference. Davis, CA: University of California: 258–268. https://doi.org/10.5070/V426110558.

Loggins, R.E.; Wilcox, J.T.; van Vuren, D.H., Sweitzer, R.A. 2002. Seasonal diets of wild pigs in oak woodlands of the central coast region of California. California Fish and Game. 88: 28–34.

Mapston, M.E. 2007. Feral hogs in Texas. B-6149. College Station, TX: Texas A&M AgriLife Extension Service. 26 p. https://hdl.handle.net/1969.1/87218.

Massei, G.; Roy, S.; Bunting, R. 2011. Too many hogs? A review of methods to mitigate impact by wild boar and feral hogs. Human-Wildlife Interactions. 5: 77–79. https://doi.org/10.26077/aeda-p853.

Mayer, J.J. 2009. Overview of wild pig damage. In: Mayer, J.J.; Brisbin, I.L., Jr., eds. Wild pigs: biology, damage, control techniques and management. SRNL-RP-2009-00869. Aiken, SC: U.S. Department of Energy, Savannah River National Laboratory: 221–246. https://doi.org/10.2172/975099.

Mayer, J.J. 2014. Estimation of the number of wild pigs found in the United States. SRNL-STI-2014-00292. Aiken, SC: U.S. Department of Energy, Savannah River National Laboratory. 8 p. https://sti.srs.gov/fulltext/SRNL-STI-2014-00292.pdf.

Mayer, J.J.; Brisbin, I.L., Jr. 2009. Wild pigs: biology, damage, control techniques and management. SRNL-RP-2009-00869. Aiken, SC: U.S. Department of Energy, Savannah River National Laboratory. 400 p. https://doi.org/10.2172/975099.

Mayer, J.J.; Nelson, E.A.; Wike, L.D. 2000. Selective depredation of planted hardwood seedlings by wild pigs in a wetland restoration area. Ecological Engineering. 15: S79–S85. https://doi.org/10.1016/S0925-8574(99)00074-9.

Mengak, M.T. 2012. 2012 Georgia wild pig survey. Final report. Athens, GA: Warnell School of Forestry and Natural Resources, University of Georgia. 41 p. https://hdl.handle.net/10724/31083.

Mississippi State University. 2013. Wild pig info. Mississippi State, MS: Mississippi State University Extension. https://www.wildpiginfo.msstate.edu/index.php. [Date accessed: August 27, 2015].

Tanger, S.M.; Guidry, K.M.; Nui, H. 2015. Monetary estimates of feral hog damage to agricultural producers in Louisiana. Journal of the NACAA. 8(2): 8 p. https://www.nacaa.com/journal/c24d174c-19d3-478f-ac32-9d1b869d75c7.

U.S. Department of Agriculture, Animal and Plant Health Inspection Service [USDA APHIS]. 2020. Feral swine-managing an invasive species. www.aphis.usda.gov/aphis/ourfocus/wildlifedamage/operational-activities/feral-swine. [Date accessed: August 20, 2019].

Watkins, C.; Caplenor, C.A.; Poudyal, N.C. [and others]. 2019. Comparing landowner support for wild hog management options in Tennessee. Journal of Environment Management. 232: 722–728. https://doi.org/10.1016/j.jenvman.2018.11.075.

West, B.C.; Cooper, A.L.; Armstrong, J.B. 2009. Managing wild pigs: a technical guide. Human-Wildlife Interactions Monographs 1. Starkville, MS: Berryman Institute. 55 p. https://doi.org/10.26079/600a-ed8d.

PLANT SUCCESSION FOLLOWING DOZER LINE INSTALLATION THROUGH A SHALE WOODLAND GLADE ON THE OUACHITA NATIONAL FOREST IN ARKANSAS

Virginia L. McDaniel, Gabriel L. De Jong, and Christopher M. Schalk

ABSTRACT

Thousands of kilometers of dozer lines are plowed each year to suppress fires on national forest lands. The recolonization and recovery process of these disturbed landscapes is not well studied and may differ across regions. In spring 2021, a dozer line was constructed through a botanically diverse, high-quality shale woodland-glade plant community on the Ouachita National Forest, AR, to suppress a 10-ha spot-over from a prescribed burn. We installed two 100-m transects in the dozer line and identified all plant species in quadrats along each transect. We compared these transects to undisturbed plant community monitoring transects that were collected from the same communities near the dozer line. While some of the area revegetated within the first growing season, the quality of plant community (determined by their coefficients of conservatism, or C-values) was lower in the dozer line than surrounding woodland-glade and several nonnative species increased in their frequency of occurrence within the dozer line. These results highlight the importance of monitoring these disturbed areas because it enables early detection and control of nonnative invasive species and helps to better understand ecological effects of management activities.

INTRODUCTION

Fire suppression and its associated activities can have many unintended consequences, including undesired landscape impacts (e.g., increased erosion) and loss of biodiversity (Backer and others 2004). Ultimately, the irony of fire suppression is that with increased success in suppressing fires, the harder and more costly they become to suppress (Calkin and others 2005, Collins and others 2019). Wildfires have increased in frequency and size over the last decade (Pausas and Keeley 2021, Weber and Yadav 2020), which subsequently increases costs and coverage of control lines on the landscape.

Prescribed burns can mitigate the unintended consequences of fire suppression as they restore the natural fire regime, help prevent catastrophic wildfires, lower the cost of fire suppression, and restore biodiversity and ecosystem structure and function (Jenkins and others 2011, Knapp and others 2009, Mitchell and others 2006). However, prescribed fires can have spot-overs (when fire moves outside control lines but is contained quickly), and on rare occasions escape, which can increase the cost and necessitate suppression tactics.

One of the most common forms of fire control is disruption of fuel continuity through the installation of firelines (Pyne and others 1996). Firelines remove fuel, which alters fire behavior and also increases access for fire personnel (Pyne and others 1996, Wilson 1988). Firelines can be installed by hand, or through tracked vehicles, such as plows or dozers (Pyne and others 1996). Dozers create firelines more efficiently than those created by hand (Pyne and others 1996). However, dozers also create large, localized

Virginia L. McDaniel, Forestry Technician, U.S. Department of Agriculture, Forest Service, Southern Research Station, Hot Springs, AR 71902
Gabriel L. De Jong, Terrestrial Ecologist, The Nature Conservancy in Arkansas, Little Rock, AR 72205
Christopher M. Schalk, Research Ecologist, U.S. Department of Agriculture, Forest Service, Southern Research Station, Nacogdoches, TX 75962

impacts through displacement of large volumes of topsoil, soil compaction, increased erosion potential through removal of the vegetation layer and physical destruction of plants (Ingalsbee and others 2019). Removal of the vegetation also resets the successional trajectory of the fireline by altering the conditions in favor of disturbance-adapted species (e.g., weedy, nonnative) to colonize and establish in these areas (Weinberger and Kaczynski 2022).

Thousands of kilometers of dozer lines are plowed each year to suppress wildfires (Gannon and others 2020, Lemons and others 2023, Steele and Barney 1983). Despite this large area of disturbance, few studies have looked at the trajectory of succession in these disturbed areas, with most studies having been conducted in the Western United States. In Glacier National Park, MT, Benson and Kurth (1995) found dozed sites were more representative of early successional plant communities with a higher number of nonnative species compared to undisturbed and burned sites. Similarly, in the southwestern Cascade Mountains, CA, dozer lines contained greater cover of nonnative species, when compared to burned and undisturbed sites, with dozed sites converting from native shrublands to grasslands dominated by nonnative species. (Weinberger and Kaczynski 2022). Dozed lines were slow to recover in the Klamath Mountains in northwestern California as, even 2.5 years postfire, the majority of the firelines were still bare ground with little organic material present (Mayer 2022). On the Ouachita National Forest, an average of 8 km of dozer line are plowed each year to contain wildfires (Strothers and Wilson 2023).

Glades are shallow-bedrock natural communities that support unique assemblages of drought-adapted forbs, warm-season grasses, and specialized fauna. In the Ouachita Mountains ecoregion, glades support many endemic plants that are found in that ecoregion and nowhere else in the world. Historically, many glades were associated with open woodland communities, forming woodland-glade mosaics in the landscape. The open vegetation structure and herbaceous-dominated ground layer of glades and woodlands are maintained by both abiotic and biotic factors, such as fire, drought, frost heave, and grazing. The most critical of these in the deeper-soiled components of these mosaic ecosystems is fire (Nelson 2005). In a suppressed fire regime, these glades and woodland mosaics are invaded by woody species, especially eastern red cedar (*Juniperus virginiana*) (Miller and others 2017). This leads to declines in herbaceous species diversity and coverage (Vander Yacht and others 2020) and decreases the amount and quality of habitat for wildlife that frequently use or are wholly dependent on open vegetation structure (Comer and others 2011). The U.S. Department of Agriculture, Forest Service uses thinning and prescribed burning to restore the structure and diverse plant community on glades.

Coefficients of conservatism (C-values) are values assigned to plants within a given region to rank their fidelity to habitat quality or stability (Wilhelm 1977). Plants that are tolerant of highly altered, degraded, or destabilized natural communities are given low values (0 to 3), whereas those tolerant of moderately altered, degraded, or destabilized communities are assigned mid-range values (4 to 6), and those that are only found in the most intact, stable natural communities of high ecological integrity are assigned the highest values (7 to 10) (Swink and Wilhelm 1994; Wilhelm and Ladd 1988; Witsell and others, 2024). C-values have been developed for many States and ecoregions and can be used in various analyses, including floristic quality assessments (FQAs), to assess the quality or health of natural communities, track the progress of ecosystem restoration efforts, or show how an action has degraded a plant community (Freyman and others 2016). C-values have recently been developed for all plant species in Arkansas (Witsell and others, 2024). The utility of these C-values is that they can provide insight into the response of plant communities to a variety of natural (e.g., droughts) and anthropogenic (e.g., dozer lines) disturbances (Witsell and others, 2024).

On April 5, 2021, the Jessieville-Winona-Fourche Ranger District of the Ouachita National Forest conducted a prescribed burn in the Redbank Restoration Area (centrum latitude and longitude coordinates = 34.7208, -93.3773). Prescribed burns conducted for restoration in glades are often done on the high end of the prescription (i.e., low end of the prescribed burn range of relative humidity) to

increase eastern red cedar mortality and return glades to their desired open structure. During this prescribed burn, the fire jumped the line and began moving towards a timber sale where the trees were already marked with paint. Paint runs the risk of being burned off on marked trees, resulting in the trees needing to be remarked. To stop advancement of the fire and protect the timber sale, the fire management officer put a dozer line around the spot-over in the middle of a shale woodland-glade complex.

Given the paucity of monitoring results or scientific studies assessing the impacts of dozer lines on natural communities in this region of the United States, we monitored the response of the woodland-glade plant community in the newly created dozer line. We compared this baseline data to another existing plant community monitoring dataset from a nearby, intact glade (see methods below). Specifically, we sought to address the following questions:

1) What is the timeframe for the native plant community to (re)colonize a dozer line through a diverse woodland-glade in Arkansas? 2) Are plants in the dozer line weedier (i.e., have a lower C-value) than plants in the high-quality woodland-glade? 3) Does the installation of the dozer line facilitate the colonization of new nonnative, invasive species that were not present in the woodland-glade prior to the disturbance?

STUDY SITE

The Ouachita National Forest is located in the Ouachita Mountains (Level III) ecoregion of western Arkansas and eastern Oklahoma (Woods and others 2004). The ridges of this mostly east-west trending mountain range are covered with pine (*Pinus* spp.), pine-oak (*Quercus* spp.), and oak woodlands and forests (USDA Forest Service 1999). We conducted our study in the Redbank Restoration Area (hereafter: Redbank) on the Jessieville-Winona-Fourche Ranger District of the Ouachita National Forest. It is located in the Central Hills, Ridges, and Valleys and Fourche Mountains (Level IV) ecoregions of the Ouachita Mountains ecoregion (fig. 1), in Montgomery and Garland Counties, AR. The Redbank unit consists of 988 ha of land owned and managed by the Forest Service. Underlying bedrock is the Stanley Shale formation, a significant geologic formation in this ecoregion, known to produce many glades in valleys and along headwater streams. These shale glades and woodlands are considered a rare plant community on the Ouachita National Forest

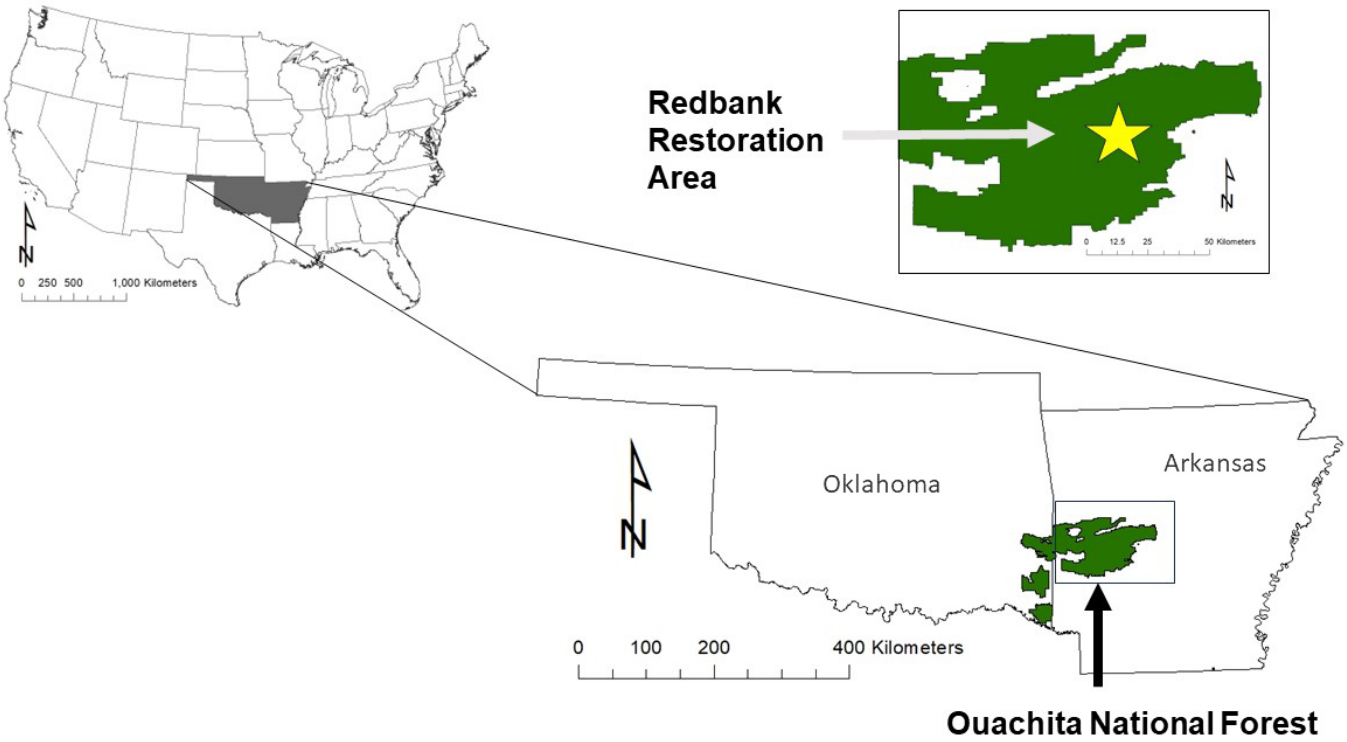

Figure 1—Location of Redbank Restoration Area denoted with yellow star in the Ouachita National Forest.

(USDA Forest Service 2005). Redbank is located 4 km north of Lake Ouachita, 12 km east of the township of Story, and 30 km northeast of Mount Ida. Average high and low temperatures in Mount Ida are 22 °C and 8 °C, respectively. Average total annual precipitation is 1,453 mm (https://www.usclimatedata.com/climate/mount-ida/arkansas/united-states/usar0959). Redbank ranges in elevation from 189 to 283 m.

METHODS

In May 2018, prior to the 2021 prescribed burn, The Nature Conservancy and the Forest Service installed three transects to monitor the vegetation in a glade restoration project at Redbank. Transects were 100 m long and species were identified in 0.25-m^2 quadrats, which were placed at 2-m intervals, starting at 0 m (51 quadrats per transect, hereafter: control transects). Each species was assigned a Daubenmire cover class as follows: 1 = 0.1 to 1 percent; 2 = 1.1 to 5 percent 3 = 5.1 to 25 percent; 4 = 25.1 to 50 percent; 5 = 50.1 to 75 percent; 6 = 75.1 to 95 percent and 7 = 95.1 to 100 percent. Data on control transects were only collected in the spring (May 2018).

The dozer line was created during the prescribed burn on April 5, 2021, clearing the vegetation in a 5- to 30-m-wide swath over a distance of 300 m. The dozer line began at the edge of a road and ran north for 100 m up a gentle, south-facing slope through the shale woodland-glade. It then cut across a forested ridgetop for 100 m before turning back down the slope of the shale woodland-glade, and tying back in with the road for the final 100 m. The plowed area was not reseeded. We installed forty 1-m^2 quadrats at 5-m intervals along the center of the dozer line on the two 100-m sections that bisected the shale woodland-glade. We identified plants to species and gave each a Daubenmire cover class estimate as follows: 1 = 0.1 to 1 percent; 2 = 1.1 to 2 percent; 3 = 2.1 to 5 percent; 4 = 5.1 to 10 percent; 5 = 10.1 to 25 percent; 6 = 25.1 to 50 percent; 7 = 50.1 to 75 percent; 8 = 75.1 to 95 percent; and 9 = 95.1 to 100 percent. Daubenmire classes were, unfortunately, slightly different than the control transect. Data were collected in late May 2021 (2 months postfire) and early September 2021 (5 months postfire).

Climatic conditions were similar during the spring of the two collection years with average rainfall between January and May was 147 mm in 2018 and 155 mm in 2021 (Jessieville RAWS, https://wrcc.dri.edu/cgi-bin/rawMAIN.pl?sdAJES). Nomenclature of species lists presented here follows Gentry and others (2013).

Analyses

We performed Wilcoxon two-sample tests to compare total cover class values between the control and spring dozer line quadrats using midpoint values of each percent cover range. For these analyses, we only used the control quadrats that were located along the transect nearest the dozer line and we dropped two quadrats from the analysis because they were in the dozer line (n = 49). To compare species C-values between the control area and dozer line, we used Wilcoxon two-sample tests using all three transects in the control areas (n = 151) and the two spring dozer line transects (n = 40). The total area sampled was comparable across the two transects (37.75 m^2 in control quadrats; 40.00 m^2 in dozer line quadrats). To understand how the dozer line was recovering between spring and fall, we only analyzed the two dozer line transects and performed a Wilcoxon signed-rank test. All analyses were performed in SAS (version 9.4, SAS Institute Inc., Cary, NC).

We calculated 10 indices for assessing floristic quality, including mean C-value, mean C-value for native species, cover-weighted mean C-value, cover-weighted mean C-value for native species, floristic quality index (FQI), FQI of native species, cover-weighted FQI, cover-weighted FQI of native species, adjusted FQI, and adjusted cover-weighted FQI (see https://cnhp.colostate.edu/download/documents/cwic_docs/FQA_calculations.pdf for terminology, description, and calculations).

Because control transect data were only collected in the spring, all analyses that compare control and dozer line transects used only the spring dozer line transect data. Additionally, our analysis considered the quadrat size difference. For FQA analyses we ensured that the total sampling area compared was similar.

RESULTS

Species Richness and Cover

Overall, the control transects had higher total species richness compared to the dozer line transects (161 species in a 37.75 m^2 total area, versus 86 species in a 40 m^2 total area, app.). Despite being only one-quarter the size of the dozer line quadrats, the control quadrats had an average of 8.3 species versus 9.6 species per quadrat in the dozer line quadrats.

Median total cover in the spring was significantly higher in the control transect (median = 62.5 percent) than the dozer line (median = 15.0 percent) (Z = -5.43, P <0.001) (fig. 2). By late summer, however, cover on the dozer line had increased significantly (Wilcoxon signed-rank test, S = -168.5, P <0.001) (figs. 3 and 4), but was still not as high as the spring cover in the control transects (Z = -3.79, P <0.001). Ocular estimation from photos indicated that some areas of the dozer line revegetated relatively quickly, while other areas remained sparsely vegetated (fig. 4).

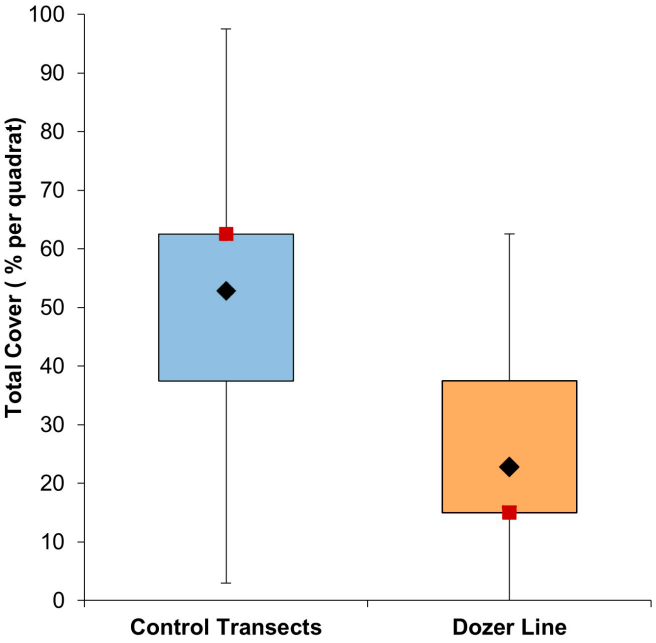

Figure 2—Total plant cover of control (forty-nine 0.25-m^2 plots collected in spring 2018) and dozer line transects (forty 1-m^2 plots collected in spring 2021). Box-and-whisker plots display maximum value, upper quartile, median (red squares), mean (black diamond), lower quartile, and minimum values.

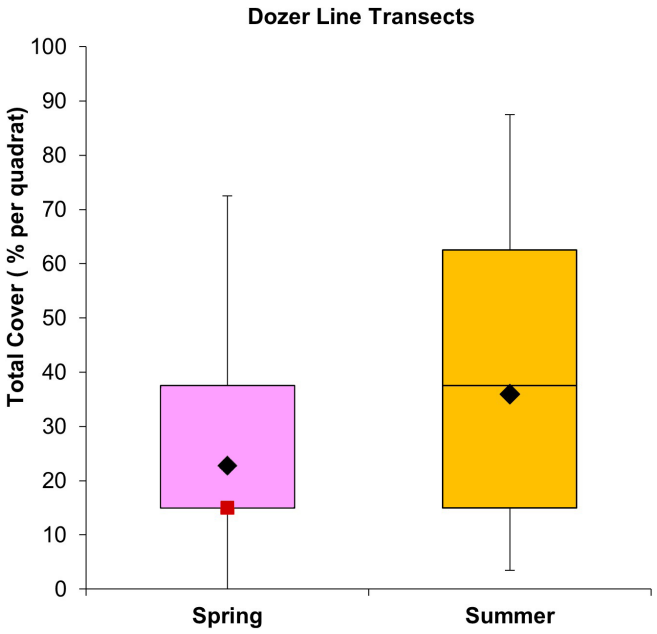

Figure 3—Total plant cover of the dozer line across seasons (forty 1-m^2 plots, spring versus summer 2021). Box-and-whisker plots display maximum value, upper quartile, median (red square), mean (black diamond), lower quartile, and minimum values.

Figure 4—Dozer line in a natural shale glade-woodland community at Redbank Restoration Area. (left) early April (directly after dozer line installation), (middle) May (2 months later, i.e., spring) and (right) September (5 months later, i.e., summer).

Floristic Quality of Dozer Line Transect

The dozer line had a greater number of species with low C-values (1 to 3) than the control transects, indicating that it was dominated by weedier, more disturbance-adapted species (Z = -2.39, *P* = 0.017) (figs. 5 and 6). There were fewer highly conservative species (i.e., C-values 7 to 10) in the dozer line, proportionally, than in the control transects. The dozer line had more mid-range valued species (C-values 5 to 6), proportionally, than the control transects (fig. 6). The control transects scored higher (indicating higher quality) in 9 out of the 10 metrics calculated for floristic quality, including mean C-value, mean native C-value, cover-weighted C-value, cover-weighted native mean C-value, FQI, native FQI, cover-weighed FQI, cover-weighted native FQI, adjusted FQI, and adjusted cover-weighted FQI (table 1). Cover-weighted mean C-value was slightly higher in the dozer line than the control transects (table 1).

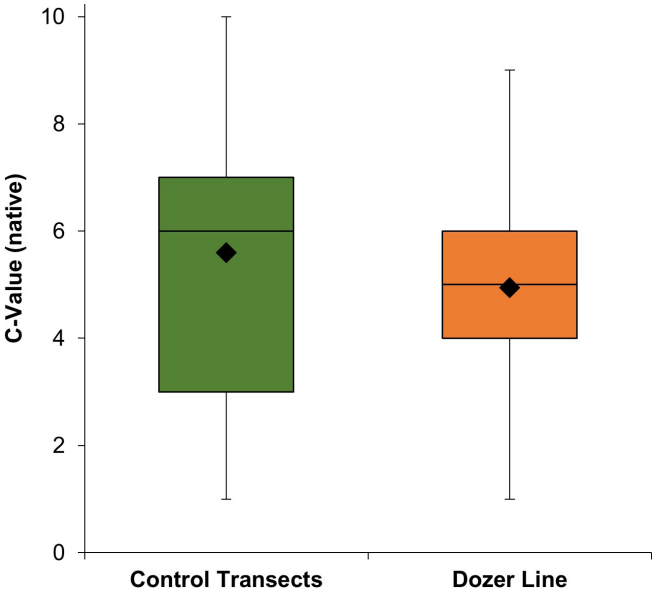

Figure 5—Distribution of coefficient of conservatism (C-values) for native species in the control transects versus the dozer line. Box-and-whisker plots display maximum value, upper quartile, median, mean (black diamond), lower quartile, and minimum values.

Table 1—Floristic quality assessment indices (FQI) for control and dozer line quadrats

Index	Control	Dozer line
Mean C-value	5.19	4.69
Mean C-value, native species	5.60	4.95
Cover-weighted mean C-value	5.33	5.35
Cover-weighted mean C-value, native species	5.69	5.40
FQI	63.33	41.14
FQI, native species	65.80	42.25
Cover-weighted FQI	65.09	46.96
Cover-weighted FQI, native species	66.81	46.14
Adjusted FQI	49.93	45.65
Adjusted cover-weighted FQI	54.73	52.58

See https://cnhp.colostate.edu/download/documents/cwic_docs/FQA_calculations.pdf for terminology, description, and calculations.
C-value = coefficient of conservatism.

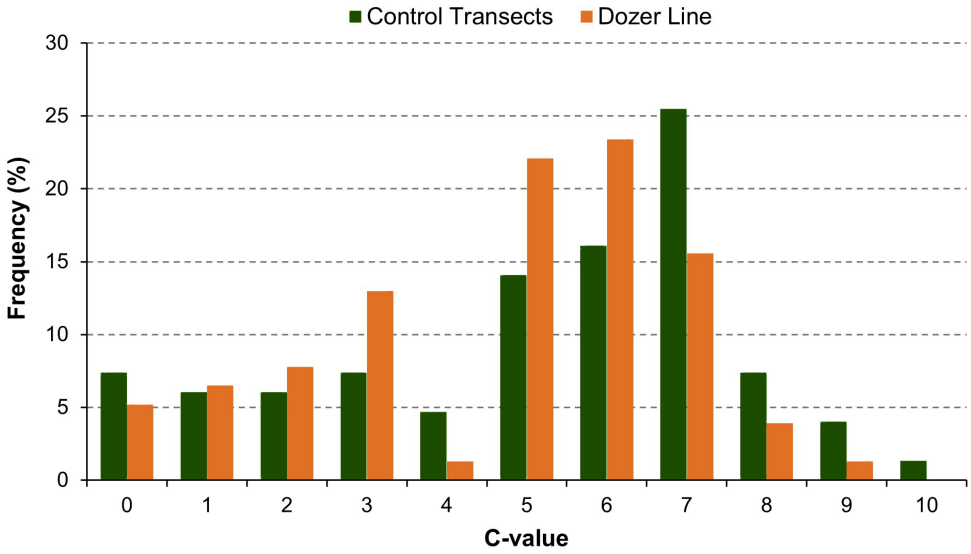

Figure 6—Frequency distribution of coefficient of conservatism (C-values) for plant species in the control transects versus the dozer line transects.

Nonnative Species

There were more nonnative species detected in the control transects (11) than the dozer line transects (4) (table 2). Two species were found in both transects, nine were found only in the control transects and two were found only in the dozer line transects. Average relative coverage of nonnative species was higher in the dozer line than the control plots (1.68 versus 0.45, table 2). Nonnative species in control transects had low relative cover values (>1.5), but the relative cover values of two species in the dozer line increased [hairy bittercress (*Cardamine hirsute*): 0.12 to 3.95 and smooth crab grass (*Digitaria ischaemum*): 0 to 2.94]. Smooth crab grass was not found in the glade prior to the dozer line, but now was found well into the interior of the woodland-glade along the dozer line corridor.

DISCUSSION

The installation of the dozer line reduced plant species richness while also increasing the coverage and occurrence of weedier, less conservative species relative to the undisturbed transect. Specifically, the dozer line had fewer highly conservative species (C-values 7 to 10), but more mid-range matrix species (C-values 4 to 6) and weedy species (C-values 1 to 3). These results are consistent with prior studies that monitor similar types of disturbance on plant communities and show that highly conservative species do not persist in areas of high disturbance, like a dozer line (Swink and Wilhelm 1994, Wilhelm 1977, Wilhelm and Ladd 1988).

Nonetheless, the shale woodland-glade plant community did exhibit some resilience to the dozer line disturbance. More than 80 plant species (76 native, 4 nonnative) were found growing in the dozer line just 5 months postdisturbance (combined total from spring and summer surveys). While total plant cover had not fully returned to predozer line levels, it had increased significantly between the spring and summer surveys. In the Ouachita ecoregion, the plant community exhibits much higher resiliency to dozer line disturbance compared to the Western United States in terms of total coverage and recolonization of native species. In the Western United States, dozer lines were still uncolonized by plants 2 years postdisturbance (Mayer 2022) and native shrubland communities were converted to nonnative grasslands (Weinberger and Kaczynski 2022). This is most

Table 2—Coverage and frequency of occurrence of nonnative species detected in control transects and dozer line transects

		Relative cover		Relative frequency	
Scientific name	Common name	Control	Dozer	Control	Dozer
Lonicera japonica	**Japanese honeysuckle**	**1.25**	**0.00**	**0.79**	**0.00**
Vulpia bromoides	Brome six-weeks grass	1.00	0.00	0.79	0.00
Veronica arvensis	Corn speedwell	0.50	0.00	0.71	0.00
Aira caryophyllea	Annual hair grass	0.62	0.00	0.55	0.00
Bromus spp.	Brome	0.54	0.00	0.47	0.00
Bromus racemosus	Bald brome	0.33	0.00	0.39	0.00
Lespedeza cuneata	**Sericea lespedeza**	**0.37**	**0.00**	**0.32**	**0.00**
Cardamine hirsuta	Hairy bittercress	0.12	3.95	0.24	4.43
Agrostemma githago	Corncockle	0.17	0.00	0.16	0.00
Kummerowia striata	Japanese bush-clover	0.04	0.29	0.08	0.52
Ligustrum sinense	**Chinese privet**	**0.04**	**0.00**	**0.08**	**0.00**
Digitaria ischaemum	Smooth crab grass	0.00	2.34	0.00	2.60
Schedonorus arundinaceus	**Tall fescue**	**0.00**	**0.15**	**0.00**	**0.26**

Bold species are considered invasive (Gentry and others 2013).

likely due to greater rainfall in the Eastern United States and perhaps greater availability of soil nutrients.

Shale woodland-glades are diverse plant communities with robust seedbanks, which may explain the recovery of the dozer line. Grodzińska and others (2000) found that the re-establishment of vegetation in mined areas was determined by the availability of seeds in the seedbank for establishment. The recent restoration work at Redbank has promoted flowering and seed production from the diverse assemblage of plants and thus improved the seedbank. Germination rates of weedy, low C-value plants were likely higher in the dozer line as seeds were exposed to more light, little competition, and access to bare soil. Additionally, perennial plants that were disturbed may have been able to resprout from intact roots. The restoration work conducted at Redbank may have made the area more resilient to this disturbance.

Studies have shown that roadways and dozer lines alter local environments and create conditions that are conducive to establishment of nonnative species (e.g., newly disturbed areas adapted to disturbance, early successional habitats) (Benson and Kurth 1995, Weinberger and Kaczynski 2022). While we did not see an increase in number of nonnative species, we did see the increase in frequency and cover of a particular nonnative species. The introduction of smooth crab grass to the woodland-glade system could have long-reaching effects. Crab grass is difficult to remove and may require herbicide to control it (Kim and others 2002). Continued monitoring is needed to determine if this species persists and expands further into the woodland-glade.

Management Implications

The Ouachita Forest Plan requires managers to rehabilitate dozer lines used to control fires, which involves seeding the disturbed area with annual fast-growing, nonpersistent, nonnative species [e.g., browntop millet (Urochloa ramose) or perennial ryegrass (Lolium perenne)]. The purpose is to quickly stabilize the soil to prevent erosion. This study suggests that if the seed bank and rootstock bank in an area are robust, seeding may not be necessary, thus reducing energy and monetary expenditures while also preventing the introduction of nonnative species, even nonpersistent ones. Shale woodland-glades tend to have robust seedbanks which may underlie their resiliency to disturbance. However, it should be noted that this was the first time this area was subjected to the installation of a dozer line. Increases in the frequency or severity of disturbances may alter the response of the plant community and require different management actions (e.g., reseeding).

An increase in nonnative species after the first disturbance may increase the seed bank of nonnative species following an additional disturbance. While the vector and pathway of smooth crab grass into the dozer line is unknown (e.g., brought in from the roadside or carried in by the equipment used to plow the dozer line), the equipment should be checked and cleaned after being in an area with invasives to further curb the spread of nonnative invasive species. Also, dozer operators may want to be mindful when plowing off-road and avoid plowing roadside weeds. Monitoring these disturbed areas is important because early detection and treatment is key for success and will reduce costs associated with their control long term. It is important to note that the roadside areas connected to the dozer line were previously treated with herbicide to control nonnative species, especially sericea lespedeza (*Lespedeza cuneata*). This factor may have reduced the potential for invasion, thus reducing overall invasion levels after dozer line installation.

Finally, while bulldozers are important fire suppression tools, thoughtful installation of dozer lines is essential to sustainably managing national forest lands. Capitalizing on natural boundaries (e.g., streams or drains) or utilizing alternative techniques (e.g., leaf blowers) in areas that support rare or endemic plants would be useful measures to mitigate the impact that fire control efforts may have on ecosystems.

ACKNOWLEDGMENTS

We would like to thank the USDA Forest Service's Southern Research Station and Jessieville-Winona-Fourche Ranger District, and The Nature Conservancy of Arkansas, for supporting this research. We appreciate the information provided

by Ouachita National Forest Fire Management Officers Wyatt Palin, Derrick Orrell, Adam Strothers, Kris Wilson, Justus Beggs, and R.L. Self as well as the support of Wildlife Biologists Mary Lynn Mentz and Rhonda Huston. Theo Witsell, Ecologist and Chief of Research for the Arkansas Natural Heritage Commission and Kristen Kaczynski, Associate Professor in Applied Ecology at the Department of Earth & Environmental Sciences, California State University at Chico, provided valuable critique that greatly improved this manuscript. This research was supported in part by the USDA Forest Service.

REFERENCES

Backer, D.M.; Jensen, S.E.; McPherson, G.R. 2004. Impacts of fire-suppression activities on natural communities. Conservation Biology. 18(4): 937–946. https://doi.org/10.1111/j.1523-1739.2004.494_1.x.

Benson, N.C.; Kurth L.L. 1995. Vegetation establishment on rehabilitated bulldozer lines after the 1988 Red Bench Fire in Glacier National Park. In: Brown, J.K.; Mutch, R.W.; Spoon, C.W.; Wakimoto, R.H., tech. coords. Proceedings: symposium on fire in wilderness and park management. Gen. Tech. Rep. INT-320. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station: 164–167.

Calkin, D.E.; Gebert, K.M.; Jones, J.G.; Neilson, R.P. 2005. Forest Service large fire area burned and suppression expenditure trends, 1970–2002. Journal of Forestry. 103(4): 179–183. https://doi.org/10.1093/jof/103.4.179.

Collins, B.M.; Miller, J.D.; Knapp, E.E.; Sapsis, D.B. 2019. A quantitative comparison of forest fires in central and northern California under early (1911–1924) and contemporary (2002–2015) fire suppression. International Journal of Wildland Fire. 28(2): 138–148. https://doi.org/10.1071/WF18137.

Comer, C.E.; Bell, A.L.; Oswald, B.P. [and others]. 2011. Vegetation and avian response to prescribed fire on glade habitats in the Missouri Ozarks. American Midland Naturalist. 165(1): 91–104. https://doi.org/10.1674/0003-0031-165.1.91.

Freyman, W.A., Masters, L.A.; Packard, S. 2016. The Universal Floristic Quality Assessment (FQA) Calculator: an online tool for ecological assessment and monitoring. Methods in Ecology and Evolution. 7(3): 380–383. https://doi.org/10.1111/2041-210X.12491.

Gannon, B.M.; Thompson, M.P.; Deming, K.Z. [and others]. 2020. A geospatial framework to assess fireline effectiveness for large wildfires in the western USA. Fire. 3(3): 43. 19 p. https://doi.org/10.3390/fire3030043.

Gentry, J.L; Johnson, G.P.; Baker, B.T. [and others], eds. 2013. Atlas of the vascular plants of Arkansas. Fayetteville, AR: University of Arkansas Herbarium. 709 p.

Grodzińska, K.; Korzeniak, U.; Szarek-Lukaszewska, G.; Godzik, B. 2000. Colonization of zinc mine spoils in southern Poland – preliminary studies on vegetation, seed rain and seed bank. Fragmenta Floristica et Geobotanica. 45(1-2): 123–145.

Ingalsbee, T.; Beasley, M.; Plummer, D.; Cowen, M. 2019. Carr Fire CATlines: the environmental impacts of bulldozers in wildfire suppression. Eugene, OR: Firefighters United for Safety, Ethics, and Ecology. 13 p. https://static1.squarespace.com/static/5e2c7d5a807d5d13389c0db6/T/5e5429427bcf434aa9c601c7/1582573910305/Carr-Fire_final.pdf. [Date accessed: October 10, 2023]

Jenkins, M.A.; Klein, R.N.; McDaniel, V.L. 2011. Yellow pine regeneration as a function of fire severity and post-burn stand structure in the southern Appalachian Mountains. Forest Ecology and Management. 262: 681–691. https://doi.org/10.1016/j.foreco.2011.05.001.

Kim, T.-J.; Neal, J.C.; Ditomaso, J.M.; Rossi, F.S. 2002. A survey of weed scientists' perceptions on the significance of crabgrasses (*Digitaria* spp.) in the United States. Weed Technology. 16(1): 239–242. https://doi.org/10.1614/0890-037X(2002)016[0239:ASOWSP]2.0.CO;2.

Knapp, E.E.; Estes, B.L.; Skinner, C.N. 2009. Ecological effects of prescribed fire season: a literature review and synthesis for managers. Gen. Tech. Rep. PSW-224. Albany, CA: U.S. Department of Agriculture Forest Service, Pacific Southwest Research Station. 80 p. https://doi.org/10.2737/PSW-GTR-224.

Lemons, R.E.; Prichard, S.J.; Kerns, B.K. 2023. Evaluating fireline effectiveness across large wildfire events in north-central Washington State. Fire Ecology. 19: 8. 16 p. https://doi.org/10.1186/s42408-023-00167-6

Mayer, M.J. 2022. An analysis of dozer line impact on local vegetation in the Klamath Mountains Bioregion. M.S. thesis. Chico, CA: California State University, Chico. 67 p.

Miller, J.E.D.; Damschen, E.I.; Ratajczak, Z.; Özdoğan, M. 2017. Holding the line: three decades of prescribed fires halt but do not reverse woody encroachment in grasslands. Landscape Ecology. 32: 2297–2310. https://doi.org/10.1007/s10980-017-0569-9.

Mitchell, R.J.; Hiers, J.K.; O'Brien, J.J. [and others]. 2006. Silviculture that sustains: the nexus between silviculture, frequent prescribed fire, and conservation of biodiversity in longleaf pine forests of the southeastern United States. Canadian Journal of Forest Research. 36(11): 2724–2736. https://doi.org/10.1139/x06-100.

Nelson, P.W. 2005. The terrestrial natural communities of Missouri. Jefferson City, MO: Missouri Department of Natural Resources. 550 p.

Pausas, J.G.; Keeley, J.E. 2021. Wildfires and global change. Frontiers in Ecology and the Environment. 19(7): 387–395. https://doi.org/10.1002/fee.2359.

Pyne, S.J.; Andrews, P.L.; Laven, R.D. 1996. Introduction to wildland fire. New York, NY: John Wiley and Sons. 769 p.

Steele, R.W.; Barney, R.J. 1983. Bulldozers in fire management: current designs and uses. Res. Note INT-328. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station. 4 p.

Strothers, A.; Wilson, J.K. 2023. Personal communication. District Zone Fire Management Officer, Mena-Oden Ranger District, Ouachita National Forest, 1603 Hwy 71, Mena, AR 71953 and District Zone Fire Management Officer, Oklahoma Ranger District, Ouachita National Forest, 111 S State Hwy 259A, Broken Bow, OK 74728.

Swink, F.; Wilhelm, G. 1994. Plants of the Chicago region. 4th ed. Indianapolis, IN: Indiana Academy of Science. 921 p.

U.S. Department of Agriculture, Forest Service [USDA Forest Service]. 1999. Ozark-Ouachita Highlands Assessment: terrestrial vegetation and wildlife. Gen. Tech. Rep. SRS-35. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 201 p. https://doi.org/10.2737/SRS-GTR-35.

U.S. Department of Agriculture, Forest Service [USDA Forest Service]. 2005. Revised Land and Resource Management Plan: Ouachita National Forest. Manage. Bull. R8-124 A. Hot Springs, AR: U.S. Department of Agriculture Forest Service, Southern Region. 175 p. https://www.fs.usda.gov/main/ouachita/landmanagement/planning. [Date accessed: October 3, 2023].

Vander Yacht, A.L.; Keyser, P.D.; Barrioz, S.A. [and others]. 2020. Litter to glitter: promoting herbaceous groundcover and diversity in mid-southern USA oak forest using canopy disturbance and fire. Fire Ecology. 16: 17. 19 p. https://doi.org/10.1186/s42408-020-00072-2.

Weber, K.T.; Yadav, R. 2020. Spatiotemporal trends in wildfires across the western United States (1950–2019). Remote Sensing. 12: 2959. 11 p. https://doi.org/10.3390/rs12182959.

Weinberger, H.S.; Kaczynski, K.M. 2022. Dozer lines influence post-fire plant community recovery. Madroño. 69(1): 74–87. https://doi.org/10.3120/0024-9637-69.1.74.

Wilhelm, G.S. 1977. Ecological assessment of open land areas in Kane County, Illinois: a checklist of the Kane County flora with numerical evaluations: its basis, rationale and application. Geneva, IL: Kane County Urban Development Commission. 32 p.

Wilhelm. G.; Ladd, D. 1988. Natural area assessment in the Chicago region. In: McCabe, R.E., ed. Transactions of the 53rd North American wildlife and natural resources conference. Washington, DC: Wildlife Management Institute: 361–375.

Wilson, A.A.G. 1988. Width of firebreak that is necessary to stop grass fires: some field experiments. Canadian Journal of Forest Research. 18(6): 682–687. https://doi.org/10.1139/x88-104.

Witsell, C.T.; McDaniel, V.L.; Baker, B.T. [and others] [2024]. Coefficients of conservatism for the vascular flora of Arkansas. Phytoneuron. 46: 1–118.

Woods, A.J.; Foti, T.L.; Chapman, S.S. [and others]. 2004. Ecoregions of Arkansas (color poster with map, descriptive text, summary tables and photographs). Reston, VA: U.S. Geological Survey.

APPENDIX

Table A.1—List of species found in the control transects in the spring and in dozer line transects in the spring and summer with C-value, relative cover, and relative frequency for each species

		Control		Dozer	
Scientific name	C-value	Relative cover	Relative frequency	Relative cover	Relative frequency
Acalypha gracilens[1]	5	N/A	N/A	4.33	6.67
Acalypha virginica	1	0.04	0.08	7.46	8.07
Acalypha virginica[1]	1	N/A	N/A	0.71	0.95
Agrostemma githago[2]	0	0.17	0.16	—	—
Aira caryophyllea[2]	0	0.62	0.55	—	—
Ambrosia artemisiifolia	1	0.08	0.16	0.15	0.26
Ambrosia artemisiifolia[1]	1	N/A	N/A	0.20	0.24
Amsonia ciliata	9	0.21	0.16	—	—
Andropogon gerardi[1]	7	N/A	N/A	0.50	0.24
Andropogon virginicus var. *virginicus*[1]	2	N/A	N/A	1.11	1.43
Antennaria parlinii	6	1.08	0.95	0.29	0.26
Antennaria parlinii[1]	6	N/A	N/A	0.30	0.48
Aristolochia serpentaria	6	0.04	0.08	0.29	0.26
Asclepias hirtella	7	0.12	0.08	—	—
Asplenium platyneuron	4	0.62	0.87	—	—
Astragalus distortus var. *engelmannii*	7	0.17	0.16	—	—
Bromus racemosus[2]	0	0.33	0.39	—	—
Bromus sp.	?	0.54	0.47	—	—
Callicarpa americana[1]	5	N/A	N/A	0.10	0.24
Callirhoe pedata	9	0.42	0.55	—	—
Cardamine hirsuta[2]	0	0.12	0.24	3.95	4.43
Carduus nutans[1, 2]	0	N/A	N/A	0.40	0.24
Carex albicans var. *albicans*	5	0.12	0.16	—	—
Carex communis[2]	0	0.08	0.08	—	—
Carex complanata	7	0.33	0.32	0.58	0.52
Carex latebracteata	8	2.00	1.66	0.29	0.26
Carex muehlenbergii	6	0.17	0.16	—	—
Carex spp.	?	0.67	0.55	—	—
Carex spp. 1	?	0.17	0.16	—	—
Carex spp. 2	?	0.12	0.16	—	—
Carya spp.	?	—	—	0.29	0.26
Carya texana	6	0.87	0.55	3.07	2.34
Carya texana[1]	6	N/A	N/A	5.44	4.05

Scientific name	C-value	Control		Dozer	
		Relative cover	Relative frequency	Relative cover	Relative frequency
Celtis tenuifolia	7	0.17	0.16	0.58	0.52
Cerastium brachypodum	7	0.46	0.79	—	—
Cercis canadensis	3	0.04	0.08	—	—
Chaerophyllum tainturieri	2	0.04	0.08	—	—
Chamaecrista nictitans var. *nictitans*	3	—	—	0.29	0.26
Chamaecrista nictitans var. *nictitans*[1]	3	N/A	N/A	0.91	1.19
Chasmanthium sessiliflorum	6	1.21	0.95	0.58	0.52
Chasmanthium sessiliflorum[1]	6	N/A	N/A	1.41	1.43
Cirsium carolinianum	7	0.08	0.08	—	—
Cirsium spp.	?	—	—	1.02	1.04
Cirsium spp.[1]	?	N/A	N/A	0.20	0.24
Clinopodium arkansanum	7	0.25	0.24	—	—
Clitoria mariana	6	0.46	0.63	0.29	0.26
Clitoria mariana[1]	6	N/A	N/A	0.91	1.19
Conyza canadensis var. *canadensis*	1	1.58	1.34	—	—
Conyza canadensis var. *canadensis*[1]	1	N/A	N/A	0.10	0.24
Coreopsis grandiflora var. *grandiflora*	7	0.87	0.79	2.49	2.34
Coreopsis grandiflora var. *grandiflora*[1]	7	N/A	N/A	1.81	2.62
Crataegous spp.	?	0.08	0.08	0.15	0.26
Crataegus uniflora	7	0.17	0.16	—	—
Croton monanthogynus	3	—	—	2.05	2.08
Croton monanthogynus[1]	3	N/A	N/A	3.23	2.14
Croton willdenowii	3	0.54	0.79	0.58	0.78
Croton willdenowii[1]	3	N/A	N/A	1.11	1.19
Cyperus echinatus[1]	5	N/A	N/A	0.10	0.24
Cyperus strigosus[1]	3	N/A	N/A	0.10	0.24
Danthonia spicata	5	2.37	2.13	0.58	0.52
Danthonia spicata[1]	5	N/A	N/A	0.20	0.24
Delphinium carolinianum subsp. *carolinianum*	7	0.08	0.08	—	—
Desmodium perplexum	5	0.33	0.24	0.44	0.26
Desmodium perplexum[1]	5	N/A	N/A	0.30	0.24
Dichanthelium acuminatum	7	1.08	1.26	0.29	0.26
Dichanthelium acuminatum[1]	7	N/A	N/A	0.10	0.24
Dichanthelium boscii	5	0.08	0.08	0.29	0.26
Dichanthelium boscii[1]	5	N/A	N/A	0.20	0.24
Dichanthelium commutatum	6	0.12	0.08	0.44	0.52

Scientific name	C-value	Control		Dozer	
		Relative cover	Relative frequency	Relative cover	Relative frequency
Dichanthelium commutatum[1]	6	N/A	N/A	1.71	2.38
Dichanthelium dichotomum	6	0.33	0.39	—	—
Dichanthelium laxiflorum	4	4.20	3.15	0.29	0.26
Dichanthelium laxiflorum[1]	4	N/A	N/A	10.79	7.38
Dichanthelium linearifolium	5	2.45	2.13	—	—
Dichanthelium linearifolium[1]	5	N/A	N/A	2.52	2.86
Dichanthelium microcarpon	5	0.12	0.16	—	—
Dichanthelium polyanthes	4	0.21	0.24	—	—
Dichanthelium ravenelii[1]	7	N/A	N/A	0.20	0.24
Dichanthelium spp.	?	—	—	5.7	6.51
Dichanthelium spp. 1[1]	?	N/A	N/A	1.11	1.19
Dichanthelium sphaerocarpon[1]	5	N/A	N/A	0.20	0.24
Digitaria ciliaris[1, 2]	0	N/A	N/A	0.40	0.24
Digitaria ischaemum[2]	0	—	—	2.34	2.60
Digitaria ischaemum[1, 2]	0	N/A	N/A	5.14	4.05
Diodia teres	2	—	—	0.88	1.04
Diodia teres[1]	2	N/A	N/A	1.61	1.19
Draba aprica	9	0.21	0.39	—	—
Echinacea pallida	8	0.42	0.39	—	—
Elymus glabriflorus	5	0.04	0.08	—	—
Elymus glabriflorus[1]	5	N/A	N/A	0.20	0.24
Erechtites hieraciifolius var. *hieraciifolius*	1	0.58	1.03	3.51	3.65
Erechtites hieraciifolius var. *hieraciifolius*[1]	1	N/A	N/A	1.31	0.95
Erigeron annuus	2	0.67	0.63	—	—
Erigeron strigosus	5	1.33	1.26	0.44	0.78
Erigeron strigosus[1]	5	N/A	N/A	0.20	0.48
Eriogonum longifolium var. *longifolium*	8	1.08	0.79	—	—
Eupatorium serotinum[1]	2	N/A	N/A	2.22	2.62
Euphorbia corollata	6	0.25	0.32	—	—
Euphorbia cyathophora	5	0.08	0.16	—	—
Euphorbia maculata	1	—	—	0.73	0.78
Euphorbia maculata[1]	1	N/A	N/A	0.20	0.48
Euphorbia ouachitana	8	1.04	1.42	—	—
Fimbristylis puberula var. *puberula*	7	0.04	0.08	—	—
Frangula caroliniana	4	0.04	0.08	—	—
Fraxinus americana	5	0.42	0.47	0.73	0.52

Scientific name	C-value	Control		Dozer	
		Relative cover	Relative frequency	Relative cover	Relative frequency
Fraxinus americana[1]	5	N/A	N/A	0.30	0.24
Galactia regularis	6	0.21	0.32	0.88	0.78
Galactia regularis[1]	6	N/A	N/A	1.81	2.38
Galium circaezans	5	—	—	0.29	0.26
Gamochaeta purpurea	3	2.45	3.15	0.15	0.26
Gamochaeta purpurea[1]	3	N/A	N/A	0.20	0.48
Geranium carolinianum var. *carolinianum*	1	0.71	1.18	—	—
Glandularia canadensis	6	0.17	0.16	—	—
Glyceria striata	5	1.62	1.18	—	—
Grindelia lanceolata	7	0.54	0.39	1.32	1.30
Helianthus hirsutus	7	3.20	2.76	7.60	5.21
Helianthus hirsutus[1]	7	N/A	N/A	7.86	4.76
Hieracium gronovii	5	0.12	0.08	—	—
Houstonia longifolia	7	0.08	0.16	—	—
Houstonia nigricans var. *nigricans*	8	0.33	0.24	—	—
Hypericum drummondii	3	—	—	0.15	0.26
Hypericum drummondii[1]	3	N/A	N/A	0.20	0.24
Hypericum hypericoides var. *hypericoides*	5	0.08	0.16	—	—
Hypericum punctatum	4	0.04	0.08	—	—
Hypoxis hirsuta	7	0.12	0.24	—	—
Juniperus virginiana var. *virginiana*	2	0.50	0.71	—	—
Krigia cespitosa var. *cespitosa*	2	0.79	1.18	—	—
Krigia virginica	5	0.58	0.87	—	—
Kummerowia striata[2]	0	0.04	0.08	0.29	0.52
Kummerowia striata[1, 2]	0	N/A	N/A	0.71	0.71
Lactuca canadensis	3	0.25	0.24	0.15	0.26
Lathyrus pusillus	7	0.42	0.71	0.44	0.52
Lathyrus venosus	7	0.12	0.24	—	—
Lepidium virginicum	1	0.37	0.47	—	—
Lespedeza cuneata[2]	0	0.37	0.32	—	—
Lespedeza cuneata[1, 2]	0	N/A	N/A	0.81	0.95
Lespedeza procumbens	5	1.12	1.34	1.75	1.82
Lespedeza repens	5	0.67	0.71	1.02	1.30
Lespedeza repens[1]	5	N/A	N/A	3.02	2.86
Lespedeza violacea[1]	7	N/A	N/A	0.30	0.24
Lespedeza virginica	7	0.04	0.08	—	—

Scientific name	C-value	Control		Dozer	
		Relative cover	Relative frequency	Relative cover	Relative frequency
Liatris squarrulosa	8	0.08	0.08	—	—
Ligustrum sinense[2]	0	0.04	0.08	—	—
Lobelia spicata	6	0.29	0.39	—	—
Lonicera japonica[2]	0	1.25	0.79	—	—
Minuartia patula	6	0.08	0.16	—	—
Mollugo verticillata[1, 2]	0	N/A	N/A	0.50	0.48
Monarda fistulosa var. *mollis*	7	0.12	0.08	0.29	0.26
Monarda russeliana	7	0.17	0.16	—	—
Muhlenbergia sobolifera[1]	7	N/A	N/A	0.60	0.71
Muhlenbergia spp.	?	0.33	0.24	—	—
Myosotis verna	6	0.21	0.39	0.29	0.26
Nemastylis nuttallii	10	0.17	0.16	—	—
Nuttallanthus texanus	2	0.12	0.08	0.29	0.26
Nyssa sylvatica	5	—	—	0.29	0.26
Oenothera linifolia	6	0.12	0.16	—	—
Opuntia nemoralis	9	0.04	0.08	—	—
Ostrya virginiana	4	0.42	0.39	—	—
Ostrya virginiana[1]	4	N/A	N/A	0.10	0.24
Oxalis dillenii	2	0.21	0.24	5.99	5.99
Oxalis dillenii[1]	2	N/A	N/A	2.62	4.29
Oxalis texana	6	3.16	3.86	—	—
Oxalis violacea	6	0.46	0.63	0.58	0.78
Packera glabella	2	0.04	0.08	—	—
Panicum anceps subsp. *anceps*	4	0.08	0.08	—	—
Panicum dichotomiflorum subsp. *dichotomiflorum*[1]	1	N/A	N/A	0.20	0.24
Panicum virgatum	6	—	—	0.29	0.26
Parthenocissus quinquefolia	3	0.17	0.16	2.63	2.34
Parthenocissus quinquefolia[1]	3	N/A	N/A	1.11	1.90
Passiflora lutea[1]	5	N/A	N/A	0.10	0.24
Penstemon arkansanus	7	0.21	0.24	0.29	0.26
Penstemon spp.[1]	?	N/A	N/A	0.20	0.24
Phlox pilosa subsp. *pilosa*	6	0.46	0.47	0.15	0.26
Physalis virginiana	6	0.17	0.24	1.02	1.30
Physalis virginiana[1]	6	N/A	N/A	0.20	0.48
Phytolacca americana var. *americana*	2	—	—	1.02	1.30
Phytolacca americana var. *americana*[1]	2	N/A	N/A	0.10	0.24

Scientific name	C-value	Control		Dozer	
		Relative cover	Relative frequency	Relative cover	Relative frequency
Pinus echinata	6	—	—	0.15	0.26
Pinus echinata[1]	6	N/A	N/A	0.20	0.48
Piptochaetium avenaceum	6	5.40	4.02	3.80	3.91
Piptochaetium avenaceum[1]	6	N/A	N/A	2.12	2.38
Plantago virginica	2	1.25	1.66	0.44	0.52
Polygala verticillata[1]	6	N/A	N/A	0.30	0.71
Prunella vulgaris subsp. *lanceolata*	3	0.37	0.32	0.15	0.26
Pycnanthemum albescens	7	—	—	0.15	0.26
Pycnanthemum albescens[1]	7	N/A	N/A	0.10	0.24
Quercus marilandica var. *marilandica*	8	0.33	0.24	0.58	0.26
Quercus rubra	6	0.33	0.24	0.29	0.26
Quercus rubra[1]	6	N/A	N/A	0.30	0.24
Quercus spp.	?	—	—	0.29	0.26
Quercus stellata	6	0.33	0.16	0.88	0.52
Quercus stellata[1]	6	N/A	N/A	1.51	0.71
Quercus velutina	5	0.12	0.08	—	—
Rhus aromatica var. *aromatica*	6	1.79	0.95	1.32	0.78
Rhus aromatica var. *aromatica*[1]	6	N/A	N/A	1.81	1.19
Rhus copallinum	3	0.21	0.16	1.02	1.56
Rhus copallinum[1]	3	N/A	N/A	0.50	0.71
Rhus glabra	3	—	—	0.73	1.04
Rhynchosia latifolia[1]	8	N/A	N/A	0.20	0.24
Rosa carolina	7	0.21	0.16	0.73	0.52
Rosa carolina[1]	7	N/A	N/A	0.10	0.24
Rubus spp.	?	0.17	0.16	1.46	1.30
Rubus spp.[1]	?	N/A	N/A	1.21	0.95
Rudbeckia hirta var. *hirta*	3	—	—	0.15	0.26
Rudbeckia hirta var. *hirta*[1]	3	N/A	N/A	0.71	0.48
Ruellia humilis	6	0.91	0.95	0.15	0.26
Ruellia pedunculata subsp. *pedunculata*	6	0.12	0.24	0.58	0.52
Salvia lyrata	2	0.46	0.47	—	—
Schedonorus arundinaceus[2]	0	—	—	0.15	0.26
Schizachyrium scoparium	6	6.53	3.63	0.73	0.52
Schizachyrium scoparium[1]	6	N/A	N/A	0.71	0.71
Scleria oligantha	6	4.90	4.81	4.97	4.69
Scleria oligantha[1]	6	N/A	N/A	8.06	7.38

Scientific name	C-value	Control		Dozer	
		Relative cover	Relative frequency	Relative cover	Relative frequency
Scutellaria ovata	7	0.62	1.10	2.63	2.60
Scutellaria ovata	7	N/A	N/A	0.20	0.24
Scutellaria parvula var. *australis*	6	0.04	0.08	—	—
Sedum pulchellum	5	0.04	0.08	—	—
Smilax bona-nox	2	1.12	1.03	3.36	3.13
Smilax bona-nox[1]	2	N/A	N/A	3.33	2.38
Smilax rotundifolia	3	0.12	0.08	—	—
Solanum carolinense[1]	1	N/A	N/A	0.60	0.48
Solidago altissima subsp. *altissima*	1	0.12	0.08	—	—
Solidago odora subsp. *odora*	7	0.04	0.08	—	—
Solidago radula	8	0.24	0.24	—	—
Solidago spp.	?	1.66	1.34	—	—
Solidago ulmifolia	7	0.83	0.71	1.61	1.82
Solidago ulmifolia[1]	7	N/A	N/A	1.01	1.67
Sporobolus clandestinus	7	0.37	0.24	—	—
Sporobolus clandestinus[1]	7	N/A	N/A	0.30	0.48
Streptanthus maculatus* subsp. *obtusifolius	8	0.17	0.24	—	—
Strophostyles spp.[1]	?	N/A	N/A	0.10	0.24
Stylosanthes biflora	7	0.42	0.55	—	—
Stylosanthes biflora[1]	7	N/A	N/A	0.20	0.48
Symphyotrichum anomalum	7	0.21	0.24	0.58	0.52
Symphyotrichum anomalum[1]	7	N/A	N/A	0.40	0.95
Symphyotrichum lanceolatum	4	0.04	0.08	—	—
Symphyotrichum patens	7	0.42	0.55	—	—
Symphyotrichum spp.[1]	?	N/A	N/A	0.30	0.48
Toxicodendron radicans	1	0.08	0.08	0.29	0.52
Tradescantia longipes	8	1.00	1.10	1.32	1.30
Tridens flavus var. *flavus*[1]	3	N/A	N/A	1.21	1.19
Trifolium dubium[2]	0	0.04	0.08	—	—
Triodanis perfoliata	2	4.53	5.99	—	—
Ulmus alata	2	1.00	1.18	1.32	1.30
Ulmus alata[1]	2	N/A	N/A	1.51	1.90
unknown forb 2	?	0.37	0.39	—	—
unknown forb 3	?	0.08	0.08	—	—
unknown graminoid 1	?	0.04	0.08	0.58	0.52
unknown graminoid 2	?	0.04	0.08	—	—

Scientific name	C-value	Control		Dozer	
		Relative cover	Relative frequency	Relative cover	Relative frequency
Vaccinium arboreum	6	0.21	0.16	—	—
Valerianella longiflora	7	0.83	1.03	—	—
Valerianella nuttallii	8	0.67	0.63	—	—
Valerianella palmeri	8	0.96	1.10	—	—
Valerianella radiata	2	0.12	0.16	—	—
Vernonia baldwinii	6	—	—	0.44	0.26
Vernonia baldwinii[1]	6	N/A	N/A	0.40	0.24
Veronica arvensis[2]	0	0.50	0.71	—	—
Vicia minutiflora	6	0.21	0.32	—	—
Viola bicolor	1	1.33	1.81	—	—
Viola palmata	5	—	—	0.44	0.26
Viola pedata	7	0.12	0.16	0.15	0.26
Viola sororia var. *missouriensis*	5	—	—	0.29	0.52
Vulpia bromoides[2]	0	1.00	0.79	—	—
Vulpia octoflora	3	0.46	0.71	—	—
Woodsia obtusa subsp. *occidentalis*	6	0.50	0.32	—	—

Species with more than one C-value indicates that the variety was unknown and thus the C-value could be either value.
Species in bold are species of conservation concern in Arkansas.
N/A = control transects that were not measured in the summer.
[1] Species found in the summer dozer line transects.
[2] Nonnative species.

EASTERN AND WESTERN LOBLOLLY PINE SOURCES IN THE WESTERN GULF COASTAL PLAIN REGION—ROOT SYSTEM OBSERVATIONS

Mary Anne Sayer, Michael C. Tyree, Michael A. Blazier, Joshua P. Adams, and Brett T. Wolfe

ABSTRACT

In the Western Gulf region, eastern sources of loblolly pine (*Pinus taeda*) are often chosen for planting rather than western sources. Eastern sources are considered less drought resistant than western sources, but to date, they have performed well in the western range. Climate shifts are expected to increase drought severity across the Western Gulf region. To facilitate planting decisions, we need knowledge of the ability of eastern and western loblolly pine sources to tolerate severe drought. Three genotypes of 15-year-old loblolly pine, two from the eastern and one from the western range, were treated with two levels each of stand density and drought. The 3-year study concluded in 2022 and carbon and water status parameters, and biomass allocation are being analyzed. Genotypes differed in their responses to stand density and drought, and it appears that carbon allocation within the root system may be related to stand production under severe drought.

INTRODUCTION

Loblolly pine (*Pinus taeda*) production is vital to wood supply in the United States and as such, is important to local and State economies in the Southeast (Abt and others 2002, Fox and others 2007). Among the loblolly pine Coastal Plain breeding population, four generations of genetic improvement over 65 years have produced a 48-percent increase in stemwood growth over unimproved sources (McKeand and others 2021). These gains continue to bolster the establishment of genetically improved loblolly pine (Fox and others 2007).

Elite eastern genotypes, many originating in the Atlantic Coastal Plain (ACP), are superior in growth compared to western sources (Lambeth and others 2005). Western sources, on the other hand, are more drought tolerant than eastern sources (Schmidtling 2001, 2003; South and others 2023). Despite this generalization, research has shown that some eastern sources tolerate water deficit. For example, Blazier and others (2004) reported that an ACP source outperformed a genetically improved local loblolly pine source on a droughty site in Oklahoma. Similarly, in a comparison of loblolly pine performance between ACP and Texas Lost Pines (TLP) families on a xeric site in North Carolina, growth of the ACP provenance was superior to that of the TLP provenance (Smith and others 2014).

Tree improvement programs have narrowed the productivity gap between eastern and western loblolly pine genotypes. Chmura and others (2007) found comparable second-year growth between a superior ACP loblolly pine genotype and a subset of western loblolly pine genotypes at two of three western sites. The ACP sources, however, remain highly desirable in the West Gulf Coastal Plain due to their record of growth west of the Mississippi River (Lambeth and others 2005, Matallana-Ramirez and others 2021).

Historically, the Western Gulf Coastal Plain experiences prolonged drought, and the severity

Mary Anne Sayer, Research Plant Physiologist, U.S. Department of Agriculture, Forest Service, Southern Research Station, Pineville, LA 71360
Michael C. Tyree, Associate Professor, Department of Biology, Indiana University of Pennsylvania, Indiana, PA 15705
Michael A. Blazier, Dean, College of Forestry, Agriculture and Natural Resources, University of Arkansas, Monticello, AR 71656
Joshua P. Adams, Associate Professor, Department of Agricultural Science and Forestry, Louisiana Tech University, Ruston, LA 71272
Brett T. Wolfe, Assistant Professor, School of Renewable Natural Resources, Louisiana State University, Baton Rouge, LA 70803

and duration of these events are expected to increase (IPCC 2021). As such, drought resistance traits may become as important as growth traits in loblolly pine genotype selection for this region. While eastern sources generally perform well under water deficit (Lambeth and others 2005), their drought resistance and growth response to anticipated changes in climate are unknown.

Physiological attributes important to loblolly pine drought resistance are affected by genotype and its interaction with the environment (Aspinwall and others 2011a, Gonzalez-Benecke and Martin 2010, Ingwers and others 2017, Stovall and others 2013). In support, Li and others (2021) reported that gene expression is different between drought resistant and sensitive loblolly pine genotypes. Based on their analysis of loblolly pine biomass accumulation and stomatal conductance using needle carbon isotope composition, Ingwers and others (2017) indicated genetic traits that maximize water uptake may control loblolly pine production under water stress. Globally, root-to-shoot ratio is negatively correlated with precipitation (Mokany and others 2006), suggesting that carbon allocation patterns favoring root system development benefit water uptake. Loblolly pine root-to-shoot ratio, however, is not consistently correlated with genotype performance (Aspinwall and other 2011b, Stovall and others 2013). Rather, root-to-shoot ratio may come into play only for a subset of genotypes under conditions of water deficit (Ingwers and others 2017).

Within the root system, biomass partitioning between fine and woody roots may also affect water uptake. Despite the importance of fine roots to soil resource acquisition (McCormack and others 2015), Aspinwall and others (2011b) found biomass allocation to the taproot rather than fine roots was correlated with genotype performance in loblolly pine. Rooting depth, which may affect production where water limits growth, is also of interest (Hacke and others 2000, Qi and others 2020). These observations indicate that the taproot and other deep woody roots are important to water uptake during drought.

In addition to carbon allocation pattern, xylem cavitation resistance of woody roots promotes water uptake as soil dries. Aspinwall and others (2011a) observed a genotype effect on loblolly pine terminal shoot hydraulic conductivity. They indicated that cavitation resistance may be under genetic control and tied to crown function during drought. The heritability of xylem function was also observed in loblolly pine seedling root systems with higher hydraulic conductivities in a Texas seed source compared to one from Mississippi (Sword Sayer and others 2005). Because woody roots are more vulnerable to cavitation than branches, woody roots assume an outsized role in whole-tree hydraulic function during drought (Domec and others 2010).

Where the risk of growth limitation by drought is high, stand density management provides an opportunity to sustain stemwood growth on an individual tree basis (Bottero and others 2017, McNulty and others 2014). Thinning at an early initial age and frequently thereafter may lessen water deficit effects on tree growth. Managing stands to minimize intraspecific competition also conserves crown mass by reducing self-pruning. In turn, carbon supplied belowground supports drought resilience by root system expansion and greater access to deep water (Mackay and others 2020).

In 2018–2019, a study was established to evaluate drought resistance of 15-year-old loblolly pine under two stand densities and in response to artificial drought. The present objective was to combine study observations of tree growth and fine root biomass with woody root system information from a subsample of trees to identify genetically controlled root system variables that may have a role in drought resistance. We hypothesized that there were eastern versus western genetic differences in loblolly pine root system attributes that affected growth responses to stand density and drought.

METHODS

Study Site

The study was located at the Louisiana State University Agricultural Center Hill Farm Research Station near Homer, LA (32°75’ N 93°04’ W) in a loblolly pine progeny trial. The soil was a complex of two U.S. Department of Agriculture, Natural Resources Conservation

Service series: Sacul very fine sandy loam (fine, mixed, active, thermic Aquic Hapludults) that is slowly permeable with a shallow argillic horizon (23 cm), and Wolfpen loamy fine sand (loamy, siliceous, semiactive, thermic Arenic Paleudalfs) that is moderately permeable with a deep argillic horizon (69 cm). The climate is humid subtropical. Using 1992 to 2022 values from three National Oceanic and Atmospheric Administration (NOAA) weather stations near the study site (USR0000LCAN, US1LACB0007, USC00164355), 30-year mean annual precipitation was estimated to be 141 cm and 30-year mean air temperatures in October through March and April through September, were estimated to be 11.4 °C and 23.9 °C, respectively (NOAA 2023).

Using the NOAA data, monthly average air temperature and sum of precipitation were used to estimate six monthly water balance parameters by the Thornthwaite water balance model (McCabe and Markstrom 2007). Soil moisture storage capacity of the 1-m depth was estimated as 108 mm using plant-available water-holding capacity data above and below the argillic horizon (B_t) (Sayer unpublished data). Water balances in January through March, April through June, July through September, and October through December were determined for 2007 through 2022 and monthly water balances were determined for the ambient rainfall and drought conditions in 2020, 2021, and 2022. Water balance parameters estimated were soil storage, runoff, surplus, potential evapotranspiration, actual evapotranspiration, and deficit.

In fall 2004, after harvesting a 50-year-old loblolly pine stand, the study area was prepared by prescribed fire and subsoiling. Five genotypes were planted in 2005 as container-grown seedlings in 0.06-ha plots at a 1.8 m by 4.9 m spacing. For the present study, three of these genotypes were chosen: LA, originating in the western range, and 7-56 and C93, originating in the eastern range. The LA seedlings were grown by open-pollinated seed with an origin near the study site. The 7-56 seedlings were also grown by open-pollinated seed but with an ACP origin. The C93 seedlings were rooted cuttings from 7-56 and, therefore, were of ACP origin.

Basal Area, Tree Height, and Fine Root Biomass

We delineated three blocks in the original study based on soil drainage. Within each block, three main plots were identified, each containing one of the genotypes (7-56, C93, LA). Four subplots within each main plot were randomly chosen from groups of four consecutive trees in planting rows that were of the dominant crown class and free of abnormalities. Each subplot was assigned one of two stand densities and one of two drought levels. Two alternate trees in each subplot were chosen for nondestructive measurement of basal area and tree height or destructive measurement of fine root biomass (FRB).

Stand density treatment levels were the original 1,122 trees/ha (UT) or thinning with removal of half of the trees in and adjacent to the subplot. Removals included the two trees on either side of measurement trees and alternate trees in tree rows adjacent to subplots for a stand density of 561 trees/ha (T). Thinning was completed by late 2018.

Drought treatment levels were ambient throughfall and stemflow (A) and imposed drought (D) that removed approximately 80 percent of throughfall and stemflow. Our intention was to simulate severe drought by excluding as much throughfall and stemflow as possible. The trees were first prepared for the D treatment by excavating a rectangular trench approximately 10 cm wide and 1.5 m deep around each of the 36 subplots, such that the distances between tree stems or stumps and the short and long trenched sides of the subplots were 0.9 m and 2.4 m, respectively. Trenches around subplots were lined with 0.1-mm thick, transparent, plastic sheeting and backfilled. Trenches minimized lateral flow of subsurface water and root migration across subplot boundaries. Trenches were completed by mid-2019.

Shelters of trees receiving the D treatment consisted of wooden frames supporting 0.1-mm thick, transparent, plastic sheeting wrapped around tree boles within frames to restrict stemflow. Frames peaked in the planting row at approximately 1 m above the forest floor and sloped to approximately 10 cm above the forest floor over the two long trenches of the D

trees. Shelters were completed by June 2020 and remained functional until July 2022 with two exceptions. Hurricanes increased September and October 2020 precipitation by 60 and 30 percent, respectively. These rainfall events caused overland flow into the D subplots. Second, a winter storm in February 2021 caused some collapse and breakage of the shelters until repairs were completed in April 2021. Shelters were continuously effective for 16 months between April 2021 and July 2022.

Total tree height (TH) was measured annually with a clinometer. Diameter at breast height (d.b.h.) was quantified every 6 to 12 months with a diameter tape. Tree basal area (BA) was determined by the equation, $\pi(\text{d.b.h.}/2)^2$ and expressed in square centimeters. Using BA values at age 15 years (BA_{15}), which was immediately before the stand density treatment was completed, relative basal area growth (RG) was calculated by equation (1):

$$RG = \left[\frac{BA_{15+x} - BA_{15}}{BA_{15}}\right] \quad (1)$$

where x = the measurement interval in years after the stand density treatment was complete.

To measure FRB, a tractor-mounted hydraulic probe was used in January 2021 to extract five soil cores (5.1 cm diameter by 50 to 60 cm long), 0.5 to 1.0 m from the base of each destructive measurement tree. The B_t of each soil core was identified and soil above and below the top of the B_t was pooled by subplot. Pine fine roots with diameters of at least 2 mm and ironstone (plus plinthite) were isolated by sample. Volumes of ironstone in samples were determined by volume displacement and expressed as cubic centimeters. Fine root samples were oven-dried to equilibrium at 70 °C and weighed. The subplot sample volumes were adjusted by ironstone volume and FRB was expressed as milligrams per cubic centimeter of mineral soil.

Root System Architecture

Three trees per genotype outside the three blocks were chosen for root system excavation in July 2021. Trees were chosen from among those that were not thinned, did not have adjacent missing trees, and were within 5 percent of mean genotype d.b.h. with no abnormalities. Each tree was felled at a 50-cm stump height and removed from the area. A 2.0-m square border equidistant from the stump center was painted on the forest floor. Using a track hoe, the outer edge of the painted square was excavated by first digging a 1-m trench and cutting wood roots at the edge of the soil block with pruning shears. Using the track hoe, soil was dug from the 4.0-m^3 block with most woody roots in the soil block remaining intact. When the top 0.5 m of the root system was exposed, the stump was struck by the track hoe bucket to free the root system from soil. A chain was attached between the stump and track hoe bucket, and the root system was pulled out of the soil. Root systems were trimmed to their 4.0-m^3 volume using pruning shears. The track hoe was used to excavate woody roots >0.5-cm diameter below 1.0 m to determine maximum rooting depth (MRD). Severed woody roots ≥1.0 cm in diameter within the 4.0-m^3 block were collected and paired with the appropriate root system.

Root systems were shaken and washed free of soil, elevated by a chain, and evaluated for root architecture. Numbers of small (1.0–5.0 cm diameter) (SFOLR) and large (>5.0 cm diameter) (LFOLR) first-order lateral roots, and number of sinker roots ≥1.0 cm in diameter at the 1.0 m depth of the taproot (SR) were counted. Using a chainsaw and pruning shears, SFOLR and LFOLR were removed from the taproot (TR) and partitioned into three woody root diameter categories: >0.5 cm and ≤1.0 cm (small woody roots, SWR), >1.0 cm and ≤5.0 cm (medium woody roots, MWR), and >5.0 cm (large woody roots, LWR). The SWR, MWR, and LWR were pooled by tree. After oven-drying to equilibrium at 70 °C, TR, SWR, MWR, and LWR dry weights by tree were quantified. Total number of first-order lateral roots (TFOLR) and total woody root system dry weight (TWR) were calculated by addition.

Statistical Analyses

Effects of genotype, stand density, and drought on RG, TH, and FRB were evaluated by analyses of variance using a split plot, randomized compete block experimental design with three blocks (SAS Institute, Inc., Version 9.1, Cary, NC). Genotype (LA, 7-56, or C93) was the whole plot effect and the factorial combination of stand density (UT, T) and drought (A, D) was the subplot effect.

The effect of genotype on root system variables (MRD, SR, SFOLR, LFOLR, TFOLR, TR, SWR, MWR, LWR, TWR) among nine trees in 2021 was evaluated by analyses of covariance using a completely randomized design with three replications and d.b.h. as the covariate.

Before both analyses, data were transformed as needed to natural logarithm values to meet the assumption of normally distributed experimental errors. Main and interaction effects and means assessed by the Tukey-Kramer test were considered significantly different at an α-level of 0.05 and marginally significantly different at an α-level of 0.10.

RESULTS AND DISCUSSION

Tree height among genotypes was measured annually as available light transitioned from an open condition to light limitation by early crown closure. Under open conditions, Blazier and others (2018) reported that by age 5 years, the open-pollinated 7-56 genotype and two varietal genotypes, C93 (which shares maternal genetics with 7-56) and C9, exhibited superior tree height compared to a fourth genotype, 8-103. The superior growth of ACP genotypes such as 7-56, C93, and C9 is well documented (Aspinwall and others 2011a, Chmura and others 2007, Stovall and others 2013). At age 14 years, lower crowns became light limited as evidenced by self-pruning, and our stand density treatment was implemented.

Once light limitations were remedied by a reduction in stand density from 1,122 to 561 trees/ha, all subplots regardless of genotype responded 6 months afterward with a significant increase in RG (table 1). At ages 15.5 and 16.5 years, mean RG increased 38 and 59 percent, respectively, in response to thinning. By age 18.0 years, trees in UT subplots were 4 percent taller than trees in T subplots [UT: 21.5 ± 0.5 m (mean ± standard error); T: 20.7 ± 0.5 m]. Similar tradeoffs between loblolly pine stem diameter and height growth in the absence of thinning have been observed (Ginn and others 1991, Sword Sayer and others 2004). By age 18.0 years, a significant interaction between stand density and genotype indicated that 7-56 and LA RG maintained positive responses to thinning (fig. 1). However, C93 RG was similar between UT and T subplots suggesting the magnitude of the C93 response to release by thinning was less than those for 7-56 and LA.

Physiological and developmental differences between C93 and 7-56 observed in 2009–2010 provide insight about the poor response of C93 to thinning relative to that of 7-56 and LA. Observations of 7-56 and C93 crown physiology at age 5 years indicated that these genotypes have a high potential for carbon fixation under favorable conditions (Osbon and others 2012). However, water deficit began in April 2010 and continued well into 2011 (fig. 2). By late summer 2010, C93 whole-crown carbon fixation suffered with lower gas exchange rates and greater premature foliage

Table 1—Significance (probability of a greater F-value) of genotype, stand density, drought, and their interactions on tree relative basal area growth and total tree height

		Relative basal area growth				Total tree height		
Source of variation	df	15.5 yr[1]	16.5 yr	18.0 yr[2]	18.5 yr	15.5 yr[1]	16.5 yr	18.0 yr[2]
Block	2	0.6192	0.2982	0.0701	0.2123	0.0005	0.0010	0.0230
Genotype (G)	2	0.3331	0.6194	0.1939	0.1632	0.0001	0.0001	0.0001
Stand density (T)	1	0.0005	0.0001	0.0001	0.0001	0.1542	0.0985	0.0236
Drought (D)	1	0.4627	0.2559	0.5139	0.5782	0.9490	0.4974	0.2268
G x T	2	0.1609	0.0574	0.0464	0.0250	0.8643	0.7311	0.6977
G x D	2	0.0354	0.1887	0.0166	0.0486	0.1413	0.0816	0.0343
T x D	1	0.6846	0.5478	0.4197	0.2212	0.2747	0.4675	0.1027
G x T x D	2	0.7425	0.6514	0.3697	0.5253	0.9620	0.6133	0.9242

Analysis of relative basal area growth at 15.5 years was conducted with natural logarithm-transformed values to establish normally distributed experimental errors.
df = degrees of freedom.
[1] 6 months after thinning was completed.
[2] After 18 months of intact shelter function.

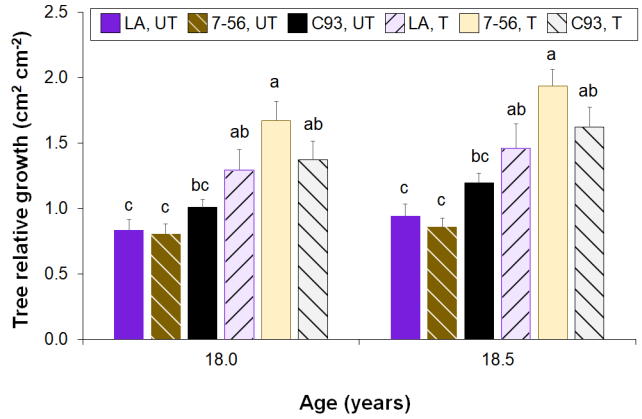

Figure 1—Tree relative basal area growth of one western source (LA) and two eastern sources (7-56, C93) of loblolly pine at ages 18.0 and 18.5 years in response to two levels of stand density [1,122 trees per hectare (UT); 561 trees per hectare (T)]. Means at the same age associated with a different lowercase letter are significantly different at an α-level of 0.05 by the Tukey-Kramer test. Error bars represent one standard error of the mean.

senescence compared to 7-56 (Blazier and others 2018, Hood and others 2012).

FRB also differed between C93 and 7-56 at age 5 years with a larger proportion of root system biomass allocated to fine roots in C93 compared to 7-56 (Blazier and others 2018). The opposite was true of woody roots, with a smaller proportion of root system biomass allocated to woody roots in C93 compared to 7-56 (Blazier and others 2018). The severe drought also caused a 90-percent decrease in FRB among all genotypes in 2010 (Sayer unpublished data). As such, a larger amount of carbon invested in C93 root systems was lost by drought compared to 7-56 root systems. In contrast to 7-56, C93 had a smaller investment in woody roots that may have reduced access to deep water and in turn, reduced carbon fixation rates and accelerated drought-induced foliage senescence.

By 2019, an increase in available light after thinning would likely have led to an increase in whole-crown gas exchange and thus, demand for water (Ginn and others 1991, Tang and others 2003). Because water deficit is common at this study site (fig. 2), root access to deep water may have been needed to meet this demand. The poor C93 response to release may be attributed to C93 root system architecture if it failed to support the access and transport of adequate water for sustained carbon fixation. Root system

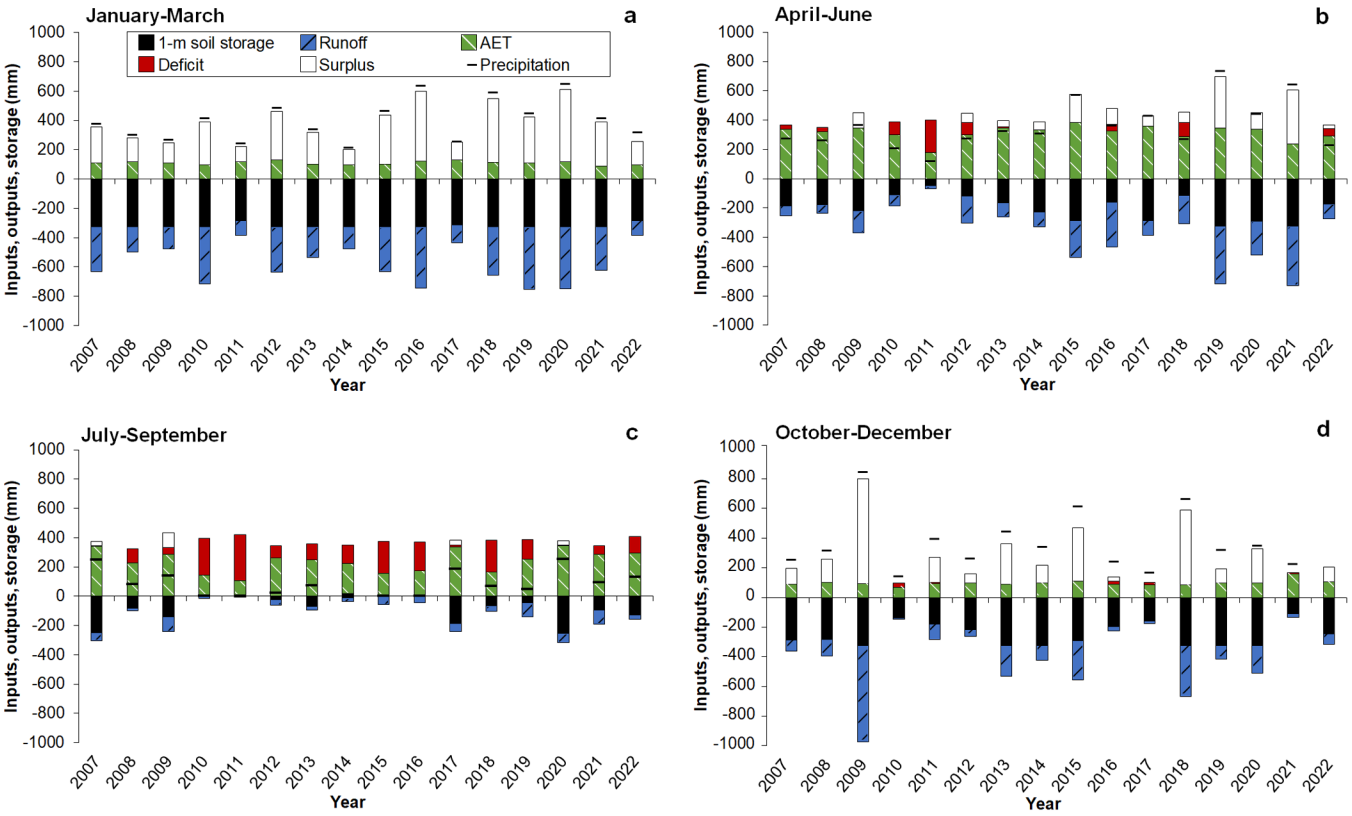

Figure 2—Water balance using the Thornthwaite model (McCabe and Markstrom 2007) and a 1-m soil water storage depth, summed annually by 3-month quarter between 2007 and 2022. AET = actual evapotranspiration.

Table 2—Mean and standard error (SE) of root system architecture variables for nine excavated trees in July 2021 and the significances (probability of a greater F-value in analyses of covariance) of the covariate, diameter at breast height (d.b.h.), and the main effect of genotype

	C93		7-56		LA		*P*	
Variable	Mean	SE	Mean	SE	Mean	SE	d.b.h.	Genotype
MRD	1.6	0.2	1.4	0.1	1.3	0.3	0.5918	0.4713
SR	6.0	2.0	4.0	2.0	4.0	1.0	0.3373	0.7122
SFOLR	6.0	1.0	7.0	2.0	6.0	2.0	0.6143	0.1497
LFOLR	9.0	1.0	13.0	1.0	10.0	1.0	0.8769	0.8561
TFOLR	15.0	1.0	20.0	2.0	16.0	1.0	0.8649	0.2796
TPR	19.7	2.7	18.8	3.3	19.8	4.3	0.0325	0.8230
SWR	0.36	0.05	0.35	0.04	0.23	0.09	0.0187	0.0657
MWR	5.6	1.2	5.0	0.7	5.5	0.5	0.5772	0.9035
LWR	7.9	0.3	7.7	1.2	7.5	1.2	0.0246	0.9955
TWR	33.5	1.6	31.9	4.8	33.0	5.6	0.0103	0.7635

MRD = maximum rooting depth (m); SR = number of sinker roots; SFOLR = number of small first order lateral roots; LFOLR = number of large first order lateral roots; TFOLR = total number of first order lateral roots; TPR = taproot dry mass (kg); SWR = small woody root dry mass (kg); MWR = medium woody root dry mass (kg); LWR = large woody root dry mass (kg); TWR = total woody root system dry mass (kg).

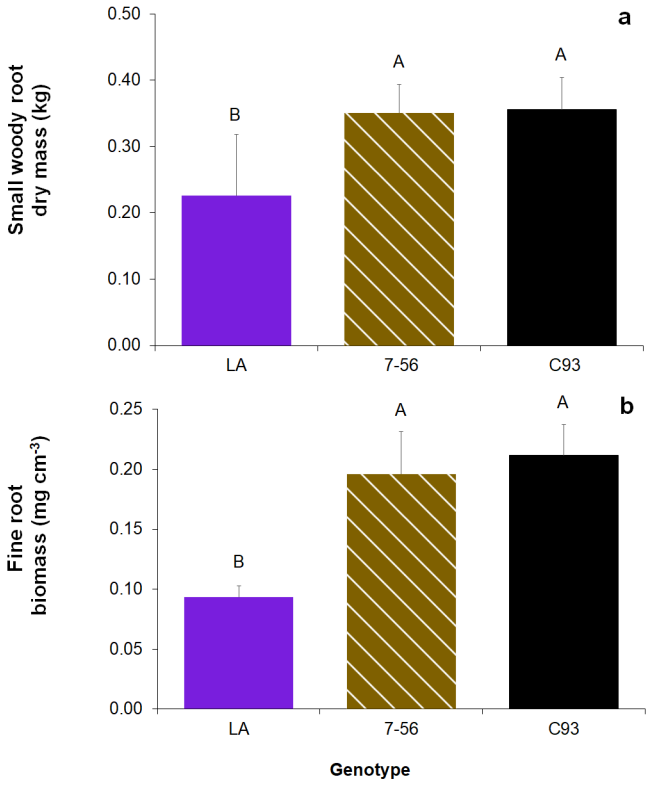

Figure 3—(a) Small woody root (>0.5 and ≤1.0 cm diameter) dry mass in July 2021 of nine trees outside experimental units, three each of the C93, 7-56, and LA genotypes, and (b) fine root biomass in January 2021 of 36 subplots by genotype. For each variable, means associated with a different uppercase letter are significantly different at an α-level of 0.10 by the Tukey–Kramer test. Error bars represent one standard error of the mean.

excavations in 2021 indicated that depth of rooting, numbers of first-order lateral roots, and dry masses of the taproot, and medium and large woody roots were not significantly affected by genotype (table 2). However, with marginal significance (P >0.0657), dry mass of small woody roots was affected by genotype, with greater values for C93 and 7-56 compared to the LA genotype (fig. 3a). With marginal significance, FRB also was affected by genotype (P >0.0536) but not stand density, drought, or their interactions, with greater FRB for C93 and 7-56 compared to the LA genotype (fig. 3b). Perhaps C93's high investment in fine roots and small woody roots was problematic to growth after thinning if this sacrificed large woody root growth. While this was not supported by our current evaluation of three trees per genotype, an expanded set of woody root variables will be assessed using nine trees per genotype excavated in 2022. A larger sample size and evaluation of additional variables important to water access and transport will improve resolution of genotype effects on root system function.

Root system architectural differences between the two eastern genotypes, C93 and 7-56, and the western genotype, LA, also may have had a role in growth responses to drought. At ages 15.5 and 16.5

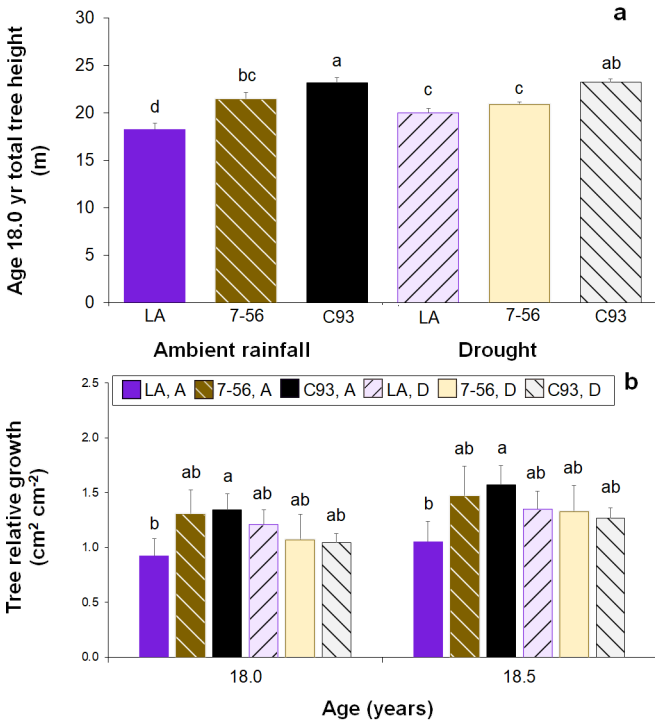

Figure 4—(a) Total tree height and (b) tree relative basal area growth of one western source (LA) and two eastern sources (7-56, C93) of loblolly pine in response to two levels of drought (ambient precipitation = A; artificial drought = D). Means at the same age associated with a different lowercase letter are significantly different at an α-level of 0.05 by the Tukey-Kramer test. Error bars represent one standard error of the mean.

years, TH differed significantly by genotype with mean TH of C93 being 13 and 24 percent greater than that of 7-56 and LA, respectively, and mean TH of 7-56 being 10 percent greater than that of LA (table 1). However, by age 18 years, which was 12 months after the drought treatment started, interaction between drought and genotype had a significant effect on TH and RG. At this time, C93 and 7-56 maintained greater TH (fig. 4a) and RG (fig. 4b) than LA under ambient conditions. Under drought conditions, however; the TH of C93 maintained its superior rank but that of 7-56 decreased and became equivalent to the TH of LA. A similar response to drought was observed in RG at ages 18.0 and 18.5 years with no differences among the three genotypes.

While it appears under ambient conditions, the RG of C93 and 7-56 were superior to that of LA, this warrants further investigation. Surprisingly, a significant interaction between genotype and drought affected RG at age 15.5 years, which was before drought treatment installation. With marginal significance, RG was similar among all but two comparisons. Specifically, the A treatment for 7-56 and the D treatment for LA had 45 and 48 percent larger RG, respectively, than the A treatment for LA. This finding requires an understanding of what caused RG to differ between the A and D treatments for LA before drought shelters were installed.

Under ambient conditions, water was generally available in the upper 1 m of soil at our study site (fig. 5a). However, artificial drought reduced throughfall and stemflow to approximately 20 percent of ambient conditions and nearly depleted water storage to a 1 m depth (fig. 5b). Our early observations of genotype effects on FRB and small woody root dry mass indicate that genetic control of carbon allocation within the root system may have a role in the performance of eastern and western sources of loblolly during severe drought. Woody root access to water deeper than 1 m and acclimation of woody roots to extreme xylem tensions represent two means of sustained water

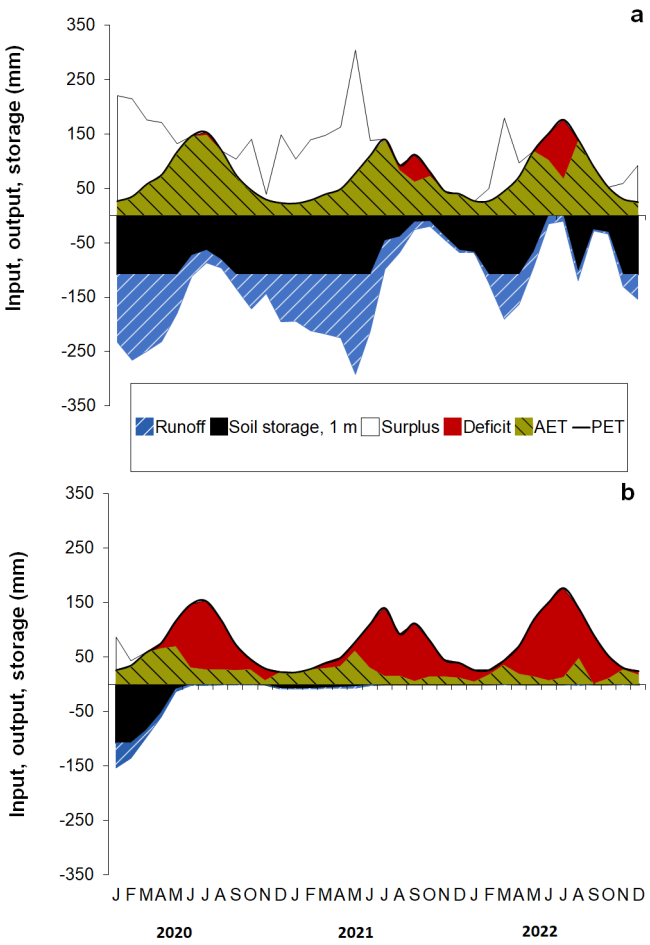

Figure 5—Water balance using the Thornthwaite model (McCabe and Markstrom 2007) and a 1-m soil water storage depth, by month between 2020 and 2022 under (a) ambient precipitation and (b) drought. AET = actual evapotranspiration; PET = potential evapotranspiration.

supply during severe drought. Future analyses will provide an understanding of root access to deep water and investigate relationships between whole-crown physiology and both the distribution of woody root sizes and their hydraulic function among genotypes.

ACKNOWLEDGMENTS

This work was supported by the USDA National Institute of Food and Agriculture, USDA AFRI/EPSCOR grant 2018-67014-27507. The authors gratefully acknowledge several individuals for their dedication during the installation and conduct of this study: Jacob Floyd, Ryan Holbrook, Zach Kennedy, Dan Leduc, Alan Springer, Kristi Wharton, Trent Verdin, and Valerie West.

REFERENCES

Abt, K.L.; Winter, S.A.; Huggett, R.J., Jr. 2002. Local economic impacts of forests. In: Wear, D.N.; Greis, J.G. eds. Southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 239–267. https://doi.org/10.2737/SRS-GTR-53.

Aspinwall, M.J.; King, J.S.; Domec, J.-C. [and others]. 2011a. Genetic effects on transpiration, canopy conductance, stomatal sensitivity to vapour pressure deficit, and cavitation resistance in loblolly pine. Ecohydrology. 4(2): 168–182. https://doi.org/10.1002/eco.197.

Aspinwall, M.J.; King, J.S.; McKeand, S.E.; Bullock, B.P. 2011b. Genetic effects on stand-level uniformity and above- and belowground dry mass production in juvenile loblolly pine. Forest Ecology and Management. 262(4): 609–619. https://doi.org/10.1016/j.foreco.2011.04.029.

Blazier, M.A.; Hennessey, T.C.; Lynch, T.B. [and others]. 2004. Productivity, crown architecture, and gas exchange of North Carolina and Oklahoma/Arkansas loblolly pine families growing on a droughty site in southeastern Oklahoma. Forest Ecology and Management. 194(1-3): 83–94. https://doi.org/10.1016/j.foreco.2004.02.014.

Blazier, M.A.; Tyree, M.C.; Sword Sayer, M.A. [and others]. 2018. Gas exchange and productivity in temperate and droughty years of four eastern, elite loblolly pine genotypes grown in the Western Gulf region. International Journal of Agronomy. 2018: 3942602. https://doi.org/10.1155/2018/3942602.

Bottero, A.; D'Amato, A.W.; Palik, B.J. [and others]. 2017. Density-dependent vulnerability of forest ecosystems to drought. Journal of Applied Ecology. 54(6): 1605–1614. https://doi.org/10.1111/1365-2664.12847.

Chmura, D.J.; Rahman, M.S.; Tjoelker, M.G. 2007. Crown structure and biomass allocation patterns modulate aboveground productivity in young loblolly pine and slash pine. Forest Ecology and Management. 243(2-3): 219–230. https://doi.org/10.1016/j.foreco.2007.02.027.

Domec, J.-C.; Schäfer, K.; Oren, R. [and others]. 2010. Variable conductivity and embolism in roots and branches of four contrasting tree species and their impacts on whole-plant hydraulic performance under future atmospheric CO_2 concentration. Tree Physiology. 30(8): 1001–1015. https://doi.org/10.1093/treephys/tpq054.

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

Ginn, S.E.; Seiler, J.R.; Cazell, B.H.; Kreh, R.E. 1991. Physiological and growth responses of eight-year-old loblolly pine stands to thinning. Forest Science. 37(4): 1030–1040.

Gonzalez-Benecke, C.A.; Martin, T.A. 2010. Water availability and genetic effects on water relations of loblolly pine (*Pinus taeda*) stands. Tree Physiology. 30(3): 376–392. https://doi.org/10.1093/treephys/tpp118.

Hacke, U.G.; Sperry, J.S.; Ewers, B.E. [and others]. 2000. Influence of soil porosity on water use in *Pinus taeda*. Oecologia. 124: 495–505. https://doi.org/10.1007/PL00008875.

Hood, W.G.; Tyree, M.C.; Dillaway, D.N. [and others]. 2012. Whole canopy gas exchange among elite loblolly pine families subjected to drought stress. In: Butnor, John R., ed. 2012. Proceedings of the 16th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 157–158.

Ingwers, M.W.; McGuire, M.A.; Aubrey, D.P. [and others]. 2017. Half-sibling loblolly pine clones exhibited intraspecific variation, a G x E interaction, and differences in stable isotope composition in response to soil moisture availability. Environmental and Experimental Botany. 138: 88–98. https://doi.org/10.1016/j.envexpbot.2017.03.010.

Intergovernmental Panel on Climate Change (IPCC). 2021. Summary for policymakers. In: Masson-Delmotte, V.; Zhai, P.; Pirani A. [and others] eds. Climate change 2021: the physical science basis. Cambridge, UK: Cambridge University Press: 3–32. https://doi.org/10.1017/9781009157896.001.

Lambeth, C.; McKeand, S.; Rousseau, R.; Schmidtling, R. 2005. Planting nonlocal seed sources of loblolly pine—managing benefits and risks. Southern Journal of Applied Forestry. 29(2): 96–104.

Li, J.; West, J.B.; Hart, A. [and others]. 2021. Extensive variation in drought-induced gene expression changes between loblolly pine genotypes. Frontiers in Genetics. 12: 661440. 15 p. https://doi.org/10.3389/fgene.2021.661440.

Mackay, D.S.; Savot, P.R.; Grossiord, C. [and others]. 2020. Conifers depend on established roots during drought: results from a coupled model of carbon allocation and hydraulics. New Phytologist. 225(2): 679–692. https://doi.org/10.1111/nph.16043.

Matallana-Ramirez, L.P.; Whetten, R.W.; Sanchez, G.M.; Payn, K.G. 2021. Breeding for climate change resilience: a case study of loblolly pine (*Pinus taeda* L.) in North America. Frontiers in Plant Science. 12: 606908. 22 p. https://doi.org/10.3389/fpls.2021.606908.

McCabe, G.J.; Markstrom, S.L. 2007. A monthly water-balance model driven by a graphical user interface. Open-File report 2007-1088. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. 6 p.

McCormack, M.L.; Dickie, I.A.; Eissenstat, D.M. [and others]. 2015. Redefining fine roots improves understanding of below-ground contributions to terrestrial biosphere processes. New Phytologist. 207(3): 505–518. https://doi.org/10.1111/nph.13363.

McKeand, S.E.; Payn, K.G.; Heine, A.J.; Abt, R.C. 2021. Economic significance of continued improvement of loblolly pine genetics and its efficient deployment to landowners in the southern United States. Journal of Forestry. 119(1): 62–72. https://doi.org/10.1093/jofore/fvaa044.

McNulty, S.G.; Boggs, J.L.; Sun, G. 2014. The rise of the mediocre forest: why chronically stressed trees may better survive extreme episodic climate variability. New Forests. 45: 403–415. https://doi.org/10.1007/s11056-014-9410-3.

Mokany, K.; Raison, R.J.; Prokushkin, A.S. 2006. Critical analysis of root:shoot ratios in terrestrial biomes. Global Change Biology. 12(1): 84–96. https://doi.org/10.1111/j.1365-2486.2005.001043.x.

National Oceanic and Atmospheric Administration [NOAA]. 2023. Climate data online. https://www.ncei.noaa.gov/cdo-web/ [Date accessed: August 21, 2023].

Osbon, B.S.; Blazier, M.A.; Tyree, M.C.; Sword-Sayer, M.A. 2012. Whole-canopy gas exchange among four elite loblolly pine seed sources planted in the Western Gulf Region. In: Butnor, John R., ed. 2012. Proceedings of the 16th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-156. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 235–236.

Qi, J.; Markewitz, D.; McGuire, M.A. [and others]. 2020. Throughfall reduction × fertilization: deep soil water usage in a clay rich ultisol under loblolly pine in the southeast USA. Frontiers in Forests and Global Change. 2: 93. 13 p. https://doi.org/10.3389/ffgc.2019.00093.

Sayer, M.A. 2023. Unpublished data. On file with: U.S. Department of Agriculture, Forest Service, Southern Research Station, 2500 Shreveport Highway, Pineville, LA 71360.

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

Schmidtling, R.C. 2003. The southern pines during the Pleistocene. Acta Horticulturae. 615: 203–209. https://doi.org/10.17660/ActaHortic.2003.615.19.

Smith, B.C.; Bullock, B.P.; Isik, F.; McKeand, SE. 2014. Modeling genetic effects on growth of diverse provenances and families of loblolly pine across optimum and deficient nutrient regimes. Canadian Journal of Forest Research. 44(11): 1453–1461. http://dx.doi.org/10.1139/cjfr-2013-0379.

South, D.B.; Starkey, T.E.; Lyons, A. 2023. Why healthy pine seedlings die after they leave the nursery. Forests. 14(3): 645. https://doi.org/10.3390/f14030645.

Stovall, J.P.; Fox, T.R.; Seiler, J.R. 2013. Allometry varies among 6-year-old *Pinus taeda* (L.) clones in the Virginia Piedmont. Forest Science. 59(1): 50–62. https://doi.org/10.5849/forsci.10-095.

Sword Sayer, M.A., Brissette, J.C.; Barnett, J.P. 2005. Root growth and hydraulic conductivity of southern pine seedlings in response to soil temperature and water availability after planting. New Forests. 30: 253–272. https://doi.org/10.1007/s11056-005-7481-x.

Sword Sayer, M.A.; Goelz, J.C.G; Chambers, J.L. [and others]. 2004. Long-term trends in loblolly pine productivity and stand characteristics in response to thinning and fertilization in the West Gulf region. Forest Ecology and Management. 192(1): 71–96. https://doi.org/10.1016/j.foreco.2004.01.006.

Tang, Z.; Chambers, J.L.; Sword, M.A.; Barnett, J.P. 2003. Seasonal photosynthesis and water relations of juvenile loblolly pine relative to stand density and canopy position. Trees. 17: 424–430. https://doi.org/10.1007/s00468-003-0256-0.

Ecosystem Services

DOES RESTORING WOODLAND STRUCTURE WITH LOBLOLLY OR SLASH PINE PRODUCE DESIRED ECOSYSTEM FUNCTIONS?

John L. Willis, Don C. Bragg, Jeffery B. Cannon, Kamal J.K. Gandhi, Kathryn R. Kidd, Adam D. Polinko, Joshua J. Puhlick, Dan Saenz, Mary Anne Sayer, Christopher M. Schalk, Andrew B. Self, Courtney M. Siegert, and J. Morgan Varner

EXTENDED ABSTRACT

Restoring fire-maintained woodlands (forests with a noncontiguous canopy and herbaceous understory) has been a priority in the Southeastern United States for 4 decades. One of the major perceived challenges to restoration is the increased presence of loblolly (*Pinus taeda*) and slash (*P. elliottii*) pines, which have been widely planted throughout the region. Land managers have traditionally addressed this problem by clearcutting loblolly and slash pines and planting longleaf (*P. palustris*) or shortleaf (*P. echinata*) pines that had been historically prevalent on their respective sites prior to Euro-American settlement (Frost 1993, Mohr and Roth 1897). While effective at quickly converting overstory species composition, this approach requires many decades to produce a mature, pine-dominated woodland; meanwhile, many of the desired structural and functional attributes of older, open forests remain absent from stands across the landscape.

John L. Willis, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Auburn, AL 36849

Don C. Bragg (deceased), Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Monticello, AR 71656

Jeffery B. Cannon, Landscape Ecologist, The Jones Center at Ichauway, Newton, GA 39870

Kamal J.K. Gandhi, Professor, D.B. Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA 30602

Kathryn R. Kidd, Associate Professor, Arthur Temple College of Forestry and Agriculture, Stephen F. Austin State University, Nacogdoches, TX 75962

Adam D. Polinko, Assistant Professor, Department of Forestry, Mississippi State University, Starkville, MS 39762

Joshua J. Puhlick, Silviculturist, The Jones Center at Ichauway, Newton, GA 39870

Dan Saenz, Research Wildlife Biologist, U.S. Department of Agriculture, Forest Service, Southern Research Station, Nacogdoches, TX 75965

Mary Anne Sayer, Research Plant Physiologist, U.S. Department of Agriculture, Forest Service, Southern Research Station, Pineville, LA 71360

Christopher M. Schalk, Research Ecologist, U.S. Department of Agriculture, Forest Service, Southern Research Station, Nacogdoches, TX 75965

Andrew B. Self, Associate Extension Professor, Department of Forestry, Mississippi State University, Starkville, MS 39762

Courtney M. Siegert, Associate Professor, Department of Forestry, Mississippi State University, Starkville, MS 39762

J. Morgan Varner, Director of Fire Research, Tall Timbers Research Station, Tallahassee, FL 32312

An alternative restoration approach is to cut loblolly and slash pine monocultures to an open forest density (about 40–60 square feet per acre of basal area) and manage the stand as a woodland featuring the existing species, or slowly convert the stand to longleaf or shortleaf pine through underplanting after cutting (Kirkman and others 2007). This approach could accelerate the restoration process by retaining residual overstory structure, which is important for the ecological function of woodlands (fig. 1). However, frequent fire will be necessary to maintain open stand conditions and promote shortleaf and longleaf pine.

Unfortunately, little information exists describing how woodlands featuring loblolly and slash pine functionally compare to longleaf or shortleaf pine woodlands. To address this knowledge gap, we reviewed the literature to evaluate the influence of southern pine species identity on several important ecosystem attributes including forest health, fire dynamics, resistance to weather-related disturbances, hydrological integrity, carbon sequestration, wildlife habitat, and wood quantity and quality.

Our review suggests that restoring woodland structure with loblolly or slash pine will have limited impacts on most evaluated attributes. Notable exceptions include wood fiber productivity, because slash and loblolly pines possess a production advantage on most sites, and wind-induced mortality, due to the fact that longleaf pine is the most wind-resistant southern pine. Too little information exists to confidently evaluate the influence of southern pine species identity on hydrological integrity and boring insect resistance. Hence, evidence suggests that cutting loblolly and slash pine monocultures to a woodland density is likely to accelerate the development of many desirable ecosystem attributes with minimal consequences for overall woodland system integrity.

Figure 1—Example of a mature stand of planted slash pine with an underplanting of longleaf pine on The Jones Center at Ichauway. This stand is still frequently burned and will be maintained at a lower basal area to allow for a gradual transition towards longleaf pine. (USDA Forest Service photo by Don C. Bragg)

LITERATURE CITED

Frost, C.C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. In: Hermann, S.M., ed. Proceedings of the Tall Timbers fire ecology conference. No. 18. Tallahassee, FL: Tall Timbers Research Station: 17–43.

Kirkman, L.K.; Mitchell, R.J.; Kaeser, M.J. [and others]. 2007. The perpetual forest: using undesirable species to bridge restoration. Journal of Applied Ecology. 44(3): 604–614. https://doi.org/10.1111/j.1365-2664.2007.01310.x.

Mohr, C.; Roth, F. 1897. The timber pines of the southern United States. Bull. 13. Washington, DC: U.S. Department of Agriculture, Division of Forestry. 176 p.

ASSESSMENT OF TRADEOFFS AMONG ECOSYSTEM SERVICES OF WETLAND RESERVE EASEMENTS

Tyler W. Corbin, Mohammad M. Bataineh, Benjamin Babst, and I-Kuai Hung

ABSTRACT

Carbon storage, attainment of desired wildlife conditions, and potential timber yields for 30 of the oldest Wetland Reserve Easements in Arkansas were quantified. Observed and projected values were used to evaluate tradeoffs among carbon storage, wildlife habitat desired conditions, and merchantable yield. Plantations with relatively higher stocking had higher carbon storage and merchantable yields and were projected to continue to grow over the next 35 years. Low stocking, as a result of lower planting survival or volunteer recruitment, resulted in plantations with about half of the total carbon storage and merchantable yield. Only 18 percent of plantations were within stocking targets and only a handful of plantations were or were projected to be within desired conditions when various habitat metrics were considered. Enhancing tree survival through improved establishment methods as well as use of tending operations may be required to accelerate and enhance achievement of desired conditions while optimizing carbon and yield.

INTRODUCTION

The Wetland Reserve Program (WRP) was established under the Food, Agriculture, Conservation, and Trade Act of 1990 (Farm Bill) (Public Law 101-624. 101st Congress 28 November 1990) and continued as the Wetland Reserve Easements component of the Agricultural Conservation Easement Program in the 2014 Farm Bill reauthorization (NRCS 2023). This voluntary program aims to restore wetlands through financial incentives to retire marginal cropland using mainly permanent easements. Afforestation and hydrological modifications are conducted to establish processes that would ideally allow recovery of wetland functions. Since the program's inception, over 440,000 ha have been enrolled in the Lower Mississippi Alluvial Valley (LMAV) (USDA 2023).

In general, little is known about what ecosystem services and functions we can expect from restored wetlands, the timeframe in which services and functions may materialize, and how they compare to those of historical wetlands (Mitsch and others 1998). A general absence of effective long-term monitoring using consistent evaluation criteria has further complicated evaluating the efficacy of easements in restoring wetland functions and services (Berkowitz 2013, King and others 2006, Rewa 2005, Ruiz-Jaen and Aide 2005, Suding 2011). Along with a lack of long-term monitoring, forest successional patterns in which initial hydrologic and floristic conditions influence outcomes, and lack of coherent performance standards are persistent obstacles to evaluating wetland restoration and afforestation in the LMAV (Berkowitz 2013). Generally, the timescale required for afforested easements to reach maturity extends much longer than that of post-restoration monitoring requirements. Because of the variability among afforested bottomland hardwood successional patterns, evaluating restoration performance within short monitoring timeframes is ineffective.

Quantifying ecosystem services and evaluating their tradeoffs provides a standardized

Tyler W. Corbin, Graduate Student, University of Arkansas at Monticello, College of Forestry, Agriculture, and Natural Resources, Monticello, AR 71656

Mohammad M. Bataineh, Associate Professor, University of Arkansas at Monticello, College of Forestry, Agriculture, and Natural Resources, Monticello, AR 71656

Benjamin Babst, Associate Professor, University of Arkansas at Monticello, College of Forestry, Agriculture, and Natural Resources, Monticello, AR 71656

I-Kuai Hung, Professor, Stephen F. Austin State University, Arthur Temple College of Forestry and Agriculture, Nacogdoches, TX 75961

restoration assessment and allows linking of ecological outcomes and societal values (Suding 2011). As an increasing number of WRP plantations are reaching canopy closure, assessing potential tradeoffs among the services provided by bottomland afforestation may provide essential information for landowners and decisionmakers. The Lower Mississippi Valley Joint Venture, Forest Resource Conservation Working Group emphasized managing forests within the LMAV to enhance structural diversity and meet the habitat requirements of forest-dependent wildlife (Wilson and others 2007). Enhancement of habitat quality for priority wildlife, however, may come at a cost of lower potential timber production or carbon storage or both in the form of retention of less merchantable trees and reduced stand stocking. Ultimately, by evaluating restoration through an ecosystem services framework, we may be able to objectively assess restoration efforts and better understand the outcomes of management alternatives. Evaluating restoration through an ecosystem services framework may inform silvicultural strategies implemented within WRP plantations that meet program objectives of maximizing wildlife habitat while also optimizing carbon storage and providing merchantable timber to mitigate management costs. Furthermore, use of simulation models in such evaluations may mitigate the need for long-term monitoring data.

The goal of this study was to use an ecosystem services approach to evaluate restoration efforts on the oldest enrolled Arkansas WRP easements, ranging from 15 to 23 years since planting. Specifically, we used field inventory data to 1) quantify carbon storage, attainment of desired wildlife conditions, and merchantable yield; and 2) evaluate potential tradeoffs among wildlife desired conditions, carbon storage, and timber yield.

METHODS

Study Area

Thirty WRP easements across nine Arkansas counties (Ashely, Chicot, Desha, Drew, Lee, Phillips, Prairie, White, and Woodruff) were selected. All easements were within the LMAV, which is characterized by a history of flooding and significant human alteration of the landscape. Within the LMAV, total forest area is estimated at 3.07 million hectares (7.6 million acres) encompassing approximately 28 percent of the total land area. The LMAV is located within the subtropical region of the Northern Temperate Zone between 29° and 37°N latitude and ranges from 89° to 92°W longitude. Across the Northern Temperate Zone, annual precipitation ranges from 100 to 160 cm. The northern portion of the LMAV has a growing season of 185 days, while the central and southern portions have a growing season of 229 and 245 days, respectively. Daily average January temperatures across the LMAV range from 3 °C to 11 °C. In July, temperatures average to 27 °C and can regularly exceed 38 °C. The dominant soil orders within the LMAV are Alfisols, Inceptisols, Entisols, and Vertisols, with elevation ranging from 30 to 91 m above sea level (Gardiner and Oliver 2005).

Sampling Design and Measurements

The 30 easements represented a subset of the oldest easements in Arkansas, ranging from 15 to 23 years since planting. This subset was selected using a stratified random sampling approach in which all enrolled easements with a closing date before 2003 were grouped by age, area, county, and watershed membership. For each county, easements were placed into one of four bins constructed by cross-tabulating easement area and closing date. Easement age and area were divided into two classes based on available county data. Easement membership to watershed subbasin (8-digit hydrologic unit) was recorded using the U.S. Geological Survey Watershed Boundary Dataset (USGS 2018). Subbasin membership was used to ensure a minimum of one easement within each subbasin was included in the sample. Easements were randomly selected from each stratum (combination of easement age, easement area, and watershed subbasin). The selected easements encompassed the range of available conditions, spanning a wide range of planted tree species [e.g., cherrybark oak (*Quercus pagoda*), Nuttall oak (*Q. texana*), water oak (*Q. nigra*), overcup oak (*Q. lyrata*), swamp chestnut oak (*Q. michauxii*), baldcypress (*Taxodium*

distichum), sugarberry (*Celtis* spp.), and green ash (*Fraxinus pennsylvanica*)], planting spacing (e.g., 3.7 m versus 4.6 m), site preparation (e.g., ripping, disking, none), ownership (e.g., private, State, Federal), and early survival and stocking conditions.

Within each easement, 1 to 8 hardwood plantations were sampled in 2018 and 2019 for a total of 105 plantations. The number of plantations selected per easement was dependent upon easement area and heterogeneity of plantations; larger, heterogeneous easements warranted a higher number of sampled plantations to represent the heterogeneity in current conditions. Within each plantation, 2–21 sampling plots were established along linear transects across compositional and structural heterogeneity, for a total of 804 plots. Linear transects were randomly selected from a grid overlay and plots were located at fixed 50- to 200-m intervals along transects based on field conditions with shorter spacing used for heterogeneous conditions (i.e., more plots) and larger spacing for homogenous conditions (i.e., fewer plots).

A series of concentric, nested, fixed-area, circular plots were established for vegetation measurements. Plots were 0.04 ha for overstory [live trees with diameter at breast height (d.b.h.) ≥7.6 cm], while saplings (height ≥1.3 m and d.b.h. <7.6 cm) were tallied in 0.01-ha subplots. Seedlings (0.3 to <1.3 m in height) and nontree vegetation were measured within three quadrats (1 m^2 each) placed at the end of one of three radial transects (5.7 m each) at azimuths of 0, 120, and 240 degrees from magnetic north. For overstory, status, species identity, d.b.h., origin (i.e., volunteer versus planted), tree height, crown attributes, vigor class, and vine and cavity presence were recorded. When present, the number and size class of each cavity was recorded. Cavity size was categorized as either small (3–25 cm in diameter) or large (≥25 cm). Total tree height was measured to the tallest live branch excluding dieback. Crown diameter was measured as the average horizontal distance for the widest axis and its perpendicular axis. Saplings were identified to species, and d.b.h., total height, and height to live crown were measured.

Analytical Approach

To quantify carbon storage and merchantable yield, total dry biomass was estimated using individual tree d.b.h. measurements and species-specific allometric equations (Jenkins and others 2003) as made available in the Southern Variant of the Forest Vegetation Simulator (FVS) (FVS Staff 2008). A carbon fraction of 0.5 was used to convert dry biomass to carbon content using the carbon submodel of the Fire and Fuels Extension of FVS (Rebain 2010). Timber yield was calculated as the sum of merchantable biomass of individual trees adjusting for tree quality as indicated by field tree vigor classification. Tree estimates were pooled at plot and plantation levels for summarization.

Wildlife habitat suitability was assessed using Mahalanobis distance (D^2) as a metric of a plantation's deviation from desired reference conditions (Etherington 2019, Mahalanobis 1936). D^2 represents the distance of each plantation from the center of a multivariate space reflecting desired condition metrics defined by the Wilson and others (2007) to meet habitat requirements of forest-dependent wildlife (table 1). To quantify desired reference conditions, target stocking levels of 60–80 percent were used to identify within-target sampled plantations (Goelz 1995). A subset of 19 sampled plantations meeting stocking requirements were used. Using this subset of plantations allowed for defining the multivariate reference space using 12 comparable metrics, circumventing the need to permute plausible habitat metrics for hypothetical reference plantations. The 12 metrics of desired habitat included canopy cover, basal area, stocking, overstory vine presence, dominant trees per unit area, midstory canopy cover, ground cover, hard mast regeneration presence, quadratic mean diameter (QMD), trees per unit area, snags (d.b.h ≥10 cm) per unit area, and cavity trees per unit area (table 1). To convert D^2 values into probabilities with a 0–1 scale, with one reflecting desired reference conditions, the inverse of Chi-squared cumulative distribution function, with n = 12, was used (Etherington 2019). Thus, plantations with a D^2 value less than that expected by chance would have probabilities closer to one and were more likely within desired reference conditions (Etherington 2019).

Table 1—Desired forest conditions to meet habitat requirements of forest-dependent wildlife species

Forest condition	Target	Conditions that may warrant management
Canopy cover (percent)	60–80	>90
Basal area (m^2/ha)	16–20	>28
Stocking (percent)	60–80	>100
Vines in overstory (percentage of inventory plots)	40–60	<30
Super-emergent trees (percentage of inventory plots)	10–20	<5
Super-emergent trees (per acre)	4–6	<1
Midstory canopy (percentage of stand)	30–60	<20
Vines in midstory (percentage of inventory plots)	50–70	<30
Understory canopy cover (percentage of stand)	40–50	<30
Ground cover (percent)	20–50	<20
Cane present (percentage of inventory plots)	20–40	<20
Regeneration of hard mast tree species (percentage of inventory plots)	30–50	<20
Logs that provide coarse woody debris (per acre)	2–4	<2
Cavity trees (snags) >4 inches diameter at breast height (per acre)	4–6	<4
Large "den" trees or "unsound cull" trees (per 10 acres)	1–4	<1

Wilson and others (2007)

Hardwood plantations developmental trajectories were projected forward 35 years (2019–2054) using FVS. Individual tree model inputs included standard site attributes, inventory design information, and individual tree metrics (Dixon 2002). FVS projections were incorporated to evaluate potential tradeoffs among wildlife habitat suitability, potential timber yield, and carbon storage by comparing metrics for each service over the 35-year projection period. Because FVS projects individual tree growth reliably, only four primary habitat metrics (i.e., basal area, stocking, QMD, and trees per unit area) were used to evaluate desired conditions of future plantations. The multivariate reference space defined by the four primary habitat metrics for a set of 22 hypothetical plantations was used to calculate D^2 and its probabilities for future plantations.

RESULTS

Carbon

Total (aboveground and belowground) carbon storage ranged from 4.8 to 81.9 tonnes/ha (fig. 1). Plantations in Desha and Chicot Counties had the lowest total carbon, while Monroe, White, Prairie, and Drew plantations had the highest stored carbon estimates. Aboveground carbon storage ranged from 3.8 to 66.9 tonnes/ha and reflected similar across-county patterns as those shown for total carbon storage (fig. 1). Sampled plantations were projected to continue accumulating total carbon within the next 30 to 40 years, with the rate of increase declining near 2054 (fig. 2). Plantations in Monroe County were projected to continue to contain the highest total carbon. Desha and Chicot plantations were projected to continue to contain the lowest total carbon. Ashley and Woodruff plantations were projected to have similar stored carbon trajectories, considering the similarity in their initial carbon storage conditions at time of sampling. Differences between counties in aboveground carbon storage trajectories were the same as those for total carbon.

Yield

Observed merchantable yield ranged from 1.7 to 84.8 tonnes/ha (fig. 3). Desha and Chicot Counties had the lowest merchantable yields, whereas Monroe, Prairie, Drew, and White

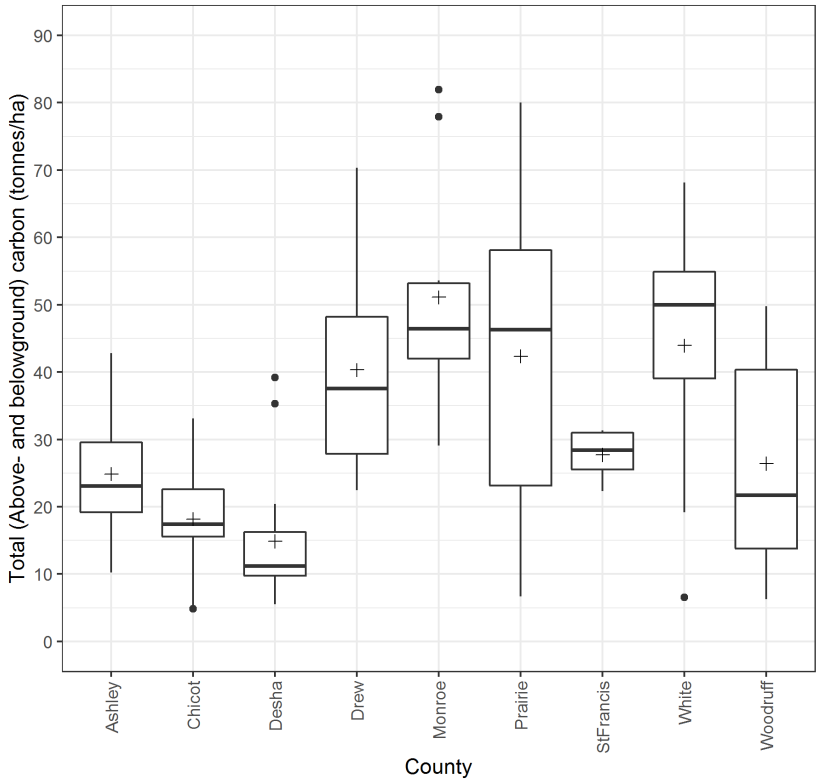

Figure 1—Total carbon, aboveground and belowground, for 105 hardwood plantations within 30 of the early enrolled Wetland Reserve Program easements across nine Arkansas counties. Mean and median shown as a plus and horizontal line, respectively. Upper and lower box bounds indicate third and first quartiles, respectively. Vertical whisker represents minimum and maximum values; dots represent outliers.

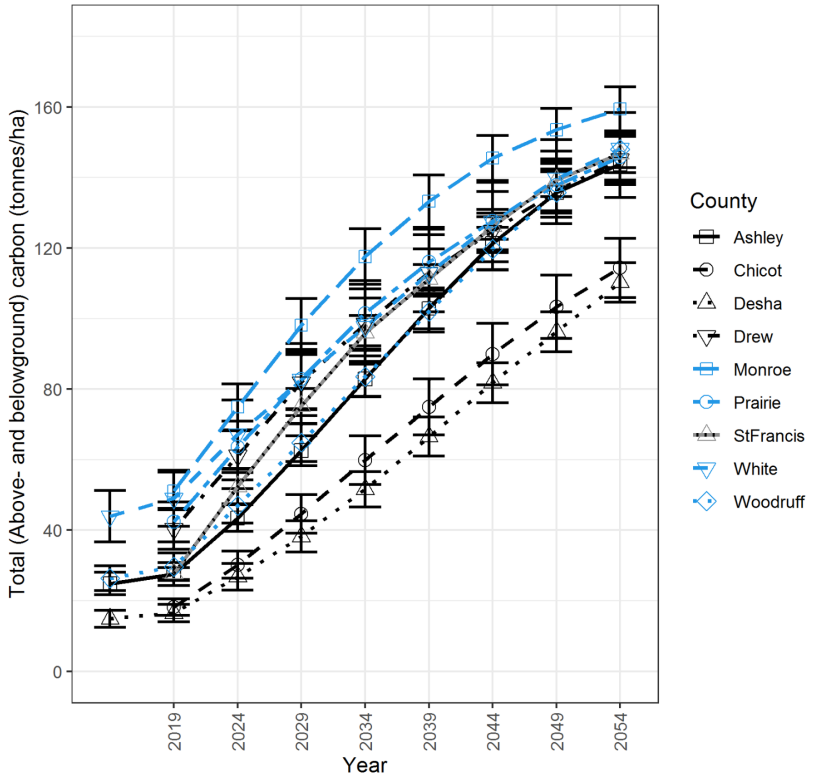

Figure 2—Projected mean estimates and standard error bars of total carbon, aboveground and belowground, for 105 hardwood plantations within 30 of the early enrolled Wetland Reserve Program easements across nine Arkansas counties.

Counties had the highest yields. Within county variability in merchantable yield was highest for Prairie County, whereas lowest variability was for St. Francis County. Overall, plantation level QMD ranged from 10.4 to 21.6 cm, indicating marginal timber merchantability of the sampled counties at current conditions. Simulated yield over a 35-year period showed a continual increase in merchantable yield over time (fig. 4). Merchantable yield highest estimates were for Monroe County throughout the simulation period. Lowest yield estimates were projected for Desha and Chicot Counties. Yield estimates for Ashley, White, Woodruff, Prairie, and St. Francis Counties converged over the simulation period (fig. 4).

Desired Conditions

Target stocking levels of 60–80 percent were used to identify within-target sampled plantations (Goelz 1995). A subset of 19 sampled plantations meeting stocking requirements were used to define the multivariate reference space, reflecting desired conditions across 12 metrics. The 19 reference plantations belonged to 13 easements across 6 counties. For the remaining 86 plantations, the majority (88 percent) represented lower than target desired stocking, while 10 plantations had higher than target stocking. In general, Prairie, Monroe, White, and Drew plantations had higher stocking values, exceeding 80 percent stocking. Desha and Chicot plantations were consistently low, with none exceeding 60-percent stocking. A total of 49 plantations (47 percent of those sampled) were below 44-percent stocking and represented understocked conditions at an average establishment age of 20 years.

Mahalanobis distance, D^2, representing the proximity of plantations current conditions to desired reference conditions, varied greatly among and within counties (fig. 5). Values of D^2 ranged from 14.6 to 1,952.8 for the 86 nonreference plantations. Inverse Chi-square probabilities showed that only four to six plantations had D^2 small enough to be within the desired reference conditions defined by the 12 observed habitat metrics (fig. 5). The four plantations most likely within desired reference conditions had 0.07 to 0.26 probabilities of D^2 less than that expected by chance, and the additional two plantations had probabilities of 0.04 each. Although reference plantations were selected based on attainment of desired stocking, stocking for the six plantations closest to desired reference conditions ranged from 55 to 88 percent, indicating that stocking alone was not the primary determinant of D^2. For the remainder of the plantations, D^2 values were from 27.9 to 1,952.8 with associated probabilities of 0–0.01 (fig. 5). When habitat metrics were truncated to four using hypothetical plantations, only five plantations across five counites had probabilities of 0.5 or higher, indicating a higher likelihood of meeting desired conditions.

Projecting developmental trajectories over the next 35 years, from 2019 to 2054, showed a decline in the number of plantations approaching desired conditions for wildlife habitat for most counties (fig. 6). In 2024, four plantations (4 percent of all plantations) were projected to have probabilities of 0.5 or higher, while four, five, three, and two plantations were projected ($P \geq 0.5$) in 2029, 2039, 2049, and 2054, respectively. Plantations with low current stocking (i.e., Chicot and Desha; below 30-percent stocking) transitioned toward desired conditions within the next 10 years or more and remained within targets over a period of 5–10 years. Without management, higher current stocking plantations deviated more from desired conditions over the projection period. Across all projection periods, plantations with higher likelihood of attaining desired conditions were consistently among the lowest in merchantable yield and total carbon storage.

DISCUSSION

Carbon, merchantable yield, and attainment of desired wildlife conditions were quantified for a representative sample of the oldest enrolled WRP easements in Arkansas to provide a standardized restoration assessment linking ecological outcomes and societal values (Suding 2011). At plantation age of 15–23 years, low stocking levels (i.e., Ashley, Desha, and Chicot easements) resulted in about half of the total carbon storage, aboveground plus belowground, expected for bottomland hardwood afforestation in the LMAV (Shoch and others 2009). Plantations with higher stocking at similar age, due to higher planting survival or volunteer recruitment, had twice the amount of total carbon storage and

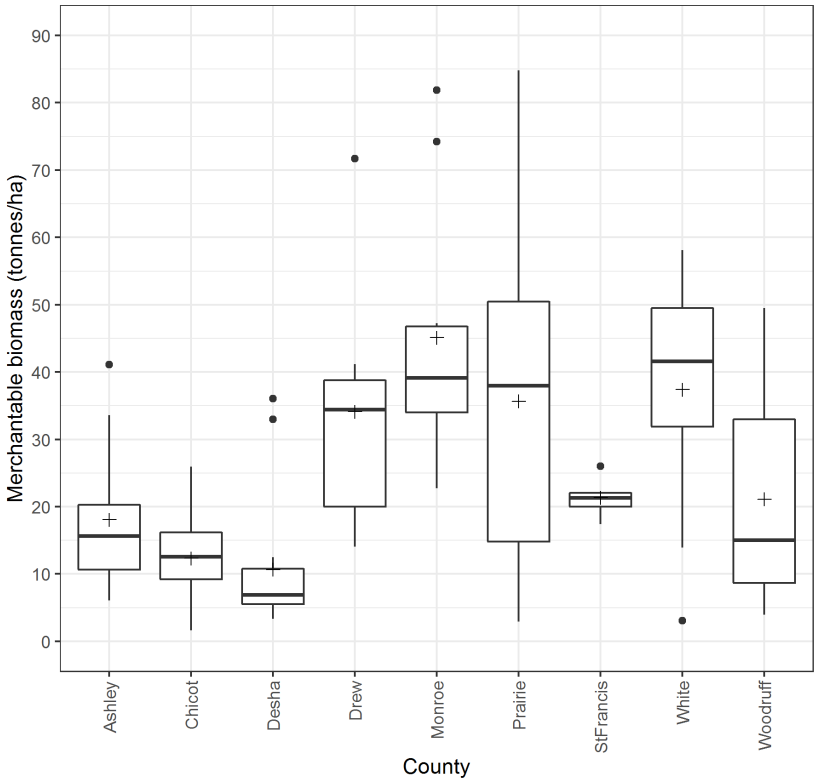

Figure 3—Merchantable biomass for 105 hardwood plantations within 30 of the early enrolled Wetland Reserve Program easements across nine Arkansas counties. Mean and median shown as a plus and horizontal line, respectively. Upper and lower box bounds indicate third and first quartiles, respectively. Vertical whisker represents minimum and maximum values; dots represent outliers.

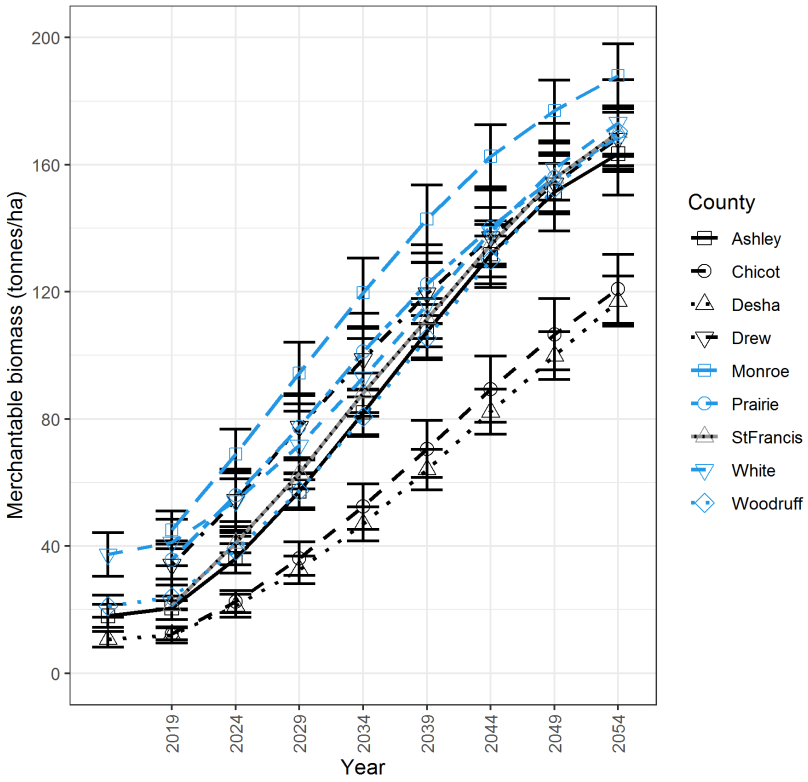

Figure 4—Projected mean estimates and standard error bars of merchantable biomass for 105 hardwood plantations within 30 of the early enrolled Wetland Reserve Program easements across nine Arkansas counties.

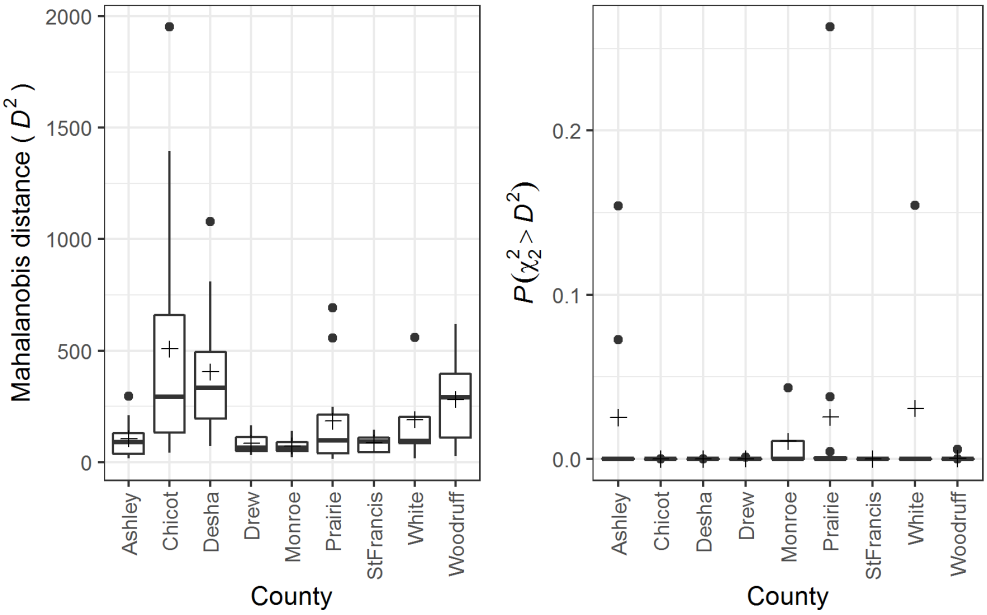

Figure 5— (left) Observed Mahalanobis distances (D^2) and (right) inverse Chi-squared probabilities, indicating proximity to desired wildlife habitat reference conditions of 86 hardwood plantations, within 30 of the early enrolled Wetland Reserve Program easements, across 9 Arkansas counties. Mean and median shown as a plus and horizontal line, respectively. Upper and lower box bounds indicate third and first quartiles, respectively. Vertical whisker represents minimum and maximum values; dots represent outliers.

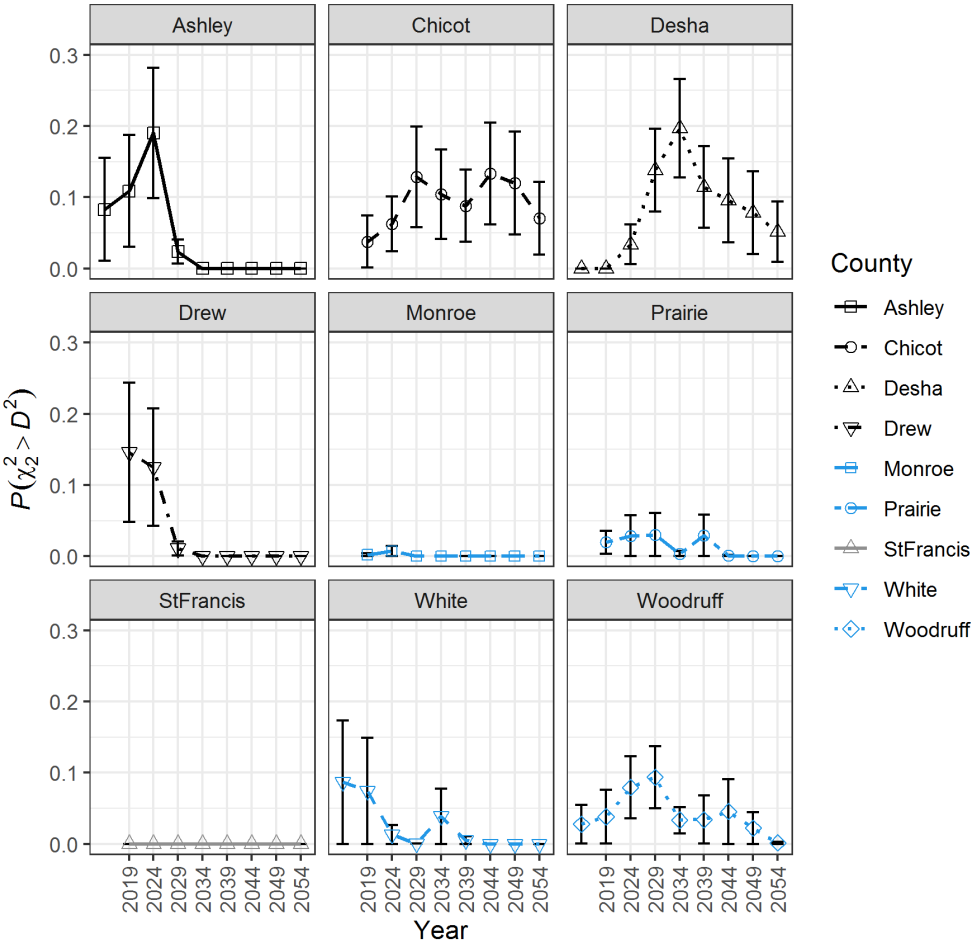

Figure 6—Mahalanobis distances (D^2) and inverse Chi-squared probabilities indicating projected proximity to desired conditions of 105 hardwood plantations, within 30 of the early enrolled Wetland Reserve Program easements, across nine Arkansas counties.

were comparable to those presented elsewhere (Shoch and others 2009). Inflection points in total carbon trajectories were projected around age 55 years since planting, with estimates of 110 to 160 tonnes/ha, indicating plantations transition into understory reinitiation stage when enhanced structural diversity would be expected (Oliver and Larson 1996). Higher potential merchantable yields were associated with higher stocking. Despite marginal merchantability as indicated by QMD, plantations in Prairie, Monroe, White, and Drew had >30 tonnes/ha on average (fig. 3). Regardless, proximity to processing facilities as well as timber quality, product class, and markets would largely influence timber provision potential from those plantations. While plantations with higher stocking were associated with higher carbon storage and potential timber yields, only 19 plantations were within desired stocking targets for wildlife habitat. When 12 habitat metrics were used to determine desired conditions, only 4–6 plantations had $P \geq 0.04$ with a maximum likelihood of 0.26 of attaining desired conditions. Truncating reference habitat conditions to four density-centric metrics resulted in the identification of five plantations with $P \geq 0.5$, indicating a higher likelihood of meeting desired conditions. Only a handful of plantations were projected to have high likelihoods of attaining desired conditions, with attainment of desired conditions lasting from 5 to 10 years.

With only 18 percent of plantations within stocking targets, a majority of plantations do not currently meet desired conditions for forest-dependent wildlife. Although plantations with low initial stocking were projected to transition toward desired conditions, their low carbon and potential yield coupled with their transitory period within future desired conditions highlight the need for alteration of plantation establishment practices. Enhancing planting survival through use of narrower spacing, higher planting density, improved planting stock, or better matching of species and sites would aid in accelerating the achievement of desired stocking (Stanturf and others 2004), with associated higher carbon storage and earlier timing of tending treatments albeit at potentially higher cost and effort (Stanturf and others 2001). Active tending and more intensive afforestation strategies may also be required to achieve and maintain desired conditions. Most plantations in this study would likely require thinning in the 2029–2034 timeframe (fig. 6), at ages of 25 to 38 years, with mean merchantable yields from 41.6 to 60.2 tonnes/ha of lower value pulpwood at an average QMD of about 11.7 to 14.2 cm (fig. 4). Such low commercial pulpwood thinning potential reflects an afforestation strategy that seeks to reduce establishment costs per unit area, thus limiting the options to regulate a plantation's future structure and mitigate management costs via timber production (Stanturf and others 2004). Enhancing structural diversity via thinning would result in forgone plantation level carbon storage compared to no management scenario (D'Amato and others 2011, Krcmar and others 2005). However, thinning to release residual trees from competition would improve residual tree vigor and future plantations merchantability (Goelz and Meadows 1997).

Although this study considered carbon storage, potential timber yield, and wildlife habitat suitability, additional ecosystem services provided by the WRP were not assessed. Although plantation deviation from desired structural reference conditions was assessed, landscape level metrics of habitat connectivity, edge structure, or species diversity was not considered. It was also beyond the scope of this study to evaluate tradeoffs using ecosystem service valuation. Future research evaluating optimization models of ecosystem service interactions and drivers of the WRP at specific spatial scales could inform future management practices and guide restoration efforts.

ACKNOWLEDGMENTS

This research was funded by the Natural Resources Conservation Service-Arkansas under funding opportunity number USDA-NRCS-AR-17-01 and grant 68-7103-17-114 with partial funding provided by the McIntire-Stennis Cooperative Forestry Research Program (project no. 1023834- ARK02681). Thanks to all students who participated in field data collection and entry. Special thanks to Amanda Bataineh and Hans Williams for reviews of an earlier version of this paper.

LITERATURE CITED

Berkowitz, J.F. 2013. Development of restoration trajectory metrics in reforested bottomland hardwood forests applying a rapid assessment approach. Ecological Indicators. 34: 600–606. https://doi.org/10.1016/j.ecolind.2013.06.025.

D'Amato, A.W.; Bradford, J.B.; Fraver, S.; Palik, B.J. 2011. Forest management for mitigation and adaptation to climate change: insights from long-term silviculture experiments. Forest Ecology and Management. 262(5): 803–816. https://dx.doi.org/10.1016/j.foreco.2011.05.014.

Dixon, G.E., comp. 2002. Essential FVS: a user's guide to the Forest Vegetation Simulator. Revised June 4, 2021. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Forest Management Service Center. 226 p.

Etherington, T.R. 2019. Mahalanobis distances and ecological niche modelling: correcting a chi–squared probability error. PeerJ. 7: e6678. 8 p. https://doi.org/10.7717/peerj.6678.

FVS Staff. 2008. Southern (SN) variant overview–Forest Vegetation Simulator. Revised October 5, 2021. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Forest Management Service Center. 83 p.

Gardiner, E.S.; Oliver, J.M. 2005. Restoration of bottomland hardwood forests in Lower Mississippi Alluvial Valley, U.S.A. In: Stanturf, J.A.; Madsen, P., eds. Restoration of boreal and temperate forests. Boca Raton, FL: CRC Press: 235–251. https://doi.org/10.1201/9780203497784.ch15.

Goelz, J.C.G. 1995. A stocking guide for southern bottomland hardwoods. Southern Journal of Applied Forestry 19(3): 103–104. https://doi.org/10.1093/sjaf/19.3.103.

Goelz, J.C.G.; Meadows, J.S. 1997. Stand density management of southern bottomland hardwoods. In: Meyer, D.A., ed. Proceedings of the 25th annual hardwood symposium: 25 years of hardwood silviculture: a look back and a look ahead. Memphis, TN: National Hardwood Lumber Association: 73–82.

Jenkins, J.C.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A. 2003. National-scale biomass estimators for United States tree species. Forest Science. 49(1): 12–35.

King, S.L.; Twedt, D.J.; Wilson, R.R. 2006. The role of the Wetland Reserve Program in conservation efforts in the Mississippi River Alluvial Valley. Wildlife Society Bulletin. 34(4): 914–920. https://doi.org/10.2193/0091-7648(2006)34[914:TROTWR]2.0.CO;2.

Krcmar, E.; van Kooten, G.C.; Vertinsky, I. 2005. Managing forest and marginal agricultural land for multiple tradeoffs: compromising on economic, carbon and structural diversity objectives. Ecological Modelling. 185: 451–468. https://doi.org/10.1016/j.ecolmodel.2004.12.014.

Mahalanobis, P.C. 1936. On the generalized distance in statistics. Proceedings of the National Institute of Science of India. 2(1): 49–55.

Mitsch, W.J.; Wu, X.; Nairn, R.W. [and others]. 1998. Creating and restoring wetlands. BioScience. 48(12): 1019–1030. https://doi.org/10.2307/1313458.

Natural Resources Conservation Service [NRCS]. 2023. Agricultural Conservation Easement Program, Wetland Reserve Easements: Restoring America's wetlands [Leaflet]. Natural Resources Conservation Service. 8 p.

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

Rebain, S.A., comp. 2010. The fire and fuels extension to the Forest Vegetation Simulator: updated model documentation. Revised December 18, 2012. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Forest Management Service Center. 398 p.

Rewa, C.A. 2005. Wildlife benefits of the Wetlands Reserve Program. In: Haufler, J.B., ed. Fish and wildlife benefits of Farm Bill conservation programs: 2000-2005 update. Tech. Rev. 05-2. Bethesda, MD: The Wildlife Society: 133–145.

Ruiz-Jaen, M.C.; Aide, T.M. 2005. Restoration success: how is it being measured? Restoration Ecology. 13(3): 569–577. https://doi.org/10.1111/j.1526-100X.2005.00072.x.

Shoch, D.T.; Kaster, G.; Hohl, A.; Souter, R. 2009. Carbon storage of bottomland hardwood afforestation in the Lower Mississippi Valley, USA. Wetlands. 29(2): 535–542. https://doi.org/10.1672/08-110.1.

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. https://doi.org/10.3368/er.22.3.183.

Stanturf, J.A.; Schoenholtz, S.H.; Schweitzer, C.J.; Shepard, J.P. 2001. Achieving restoration success: myths in bottomland hardwood forests. Restoration Ecology. 9(2): 189–200. https://doi.org/10.1046/j.1526-100x.2001.009002189.x.

Suding, K.N. 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. Annual Review of Ecology, Evolution, and Systematics. 42: 465–487. https://doi.org/10.1146/annurev-ecolsys-102710-145115.

U.S. Department of Agriculture [USDA]. 2023. Easement program data. U.S. Department of Agriculture. https://www.farmers.gov/data/easements-overview. [Date accessed: May 5, 2023].

U.S. Geological Survey [USGS]. 2018. Watershed boundary dataset. U.S. Department of the Interior, Geological Survey, National Geospatial Technical Operations Center. https://www.usgs.gov/national-hydrography/watershed-boundary-dataset. [Date accessed: January 1, 2018].

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 Resources Conservation Working Group. 88 p.

EVALUATING STEM AND BRANCH CHARACTERISTICS OF 5-YEAR-OLD SONDEREGGER PINE SAPLINGS

Kelsey D. Shoemaker, D. Paul Jackson, and Joshua P. Adams

ABSTRACT

Sonderegger pine (*Pinus* x *sondereggeri*), the natural hybrid of longleaf (*P. palustris*) and loblolly pine (*P. taeda*), is often described as having stem and branch deformities at maturity that may lead to poor wood quality. For this reason, Sonderegger pine seedlings are traditionally culled from southern forest tree nurseries to ensure only pure longleaf pine seedlings are outplanted to meet landowner objectives. The general acceptance of Sonderegger pine as an inferior tree has led to an absence of research to determine its morphological development from the seedling to mature stand management phases. In 2018, 400 Sonderegger pine seedlings were operationally detected in the nursery and outplanted on the Catahoula Ranger District of the Kisatchie National Forest (Pollock, LA) with the main objective of evaluating and describing their morphological development over time. Before planting, seedling height to the terminal bud and root collar diameter were measured. In the field, tree height and diameter at breast height measurements were recorded, and evaluations for stem straightness, forking defects, ramicorn branches, and disease occurrence were made. These observations along with ongoing assessments of branch angle, diameter, and height along the bole may provide insight into Sonderegger pine morphology at age five as it relates to long-term wood quality and utilization.

INTRODUCTION

In the Southeastern United States, longleaf (*Pinus palustris*) and loblolly (*P. taeda*) pines are most widely known for providing highly diverse forest ecosystems and a fast-growing, reliable supply of timber for wood products, respectively. Of the major southern pines, longleaf pine is considered the most valuable in terms of wood-product quality, aesthetics, and tolerance to fire, insects, and diseases (Barnett 2002). Longleaf pine is also a good choice for sawtimber and pole production, and its needles are ideal for pine straw raking, which offers an alternative source of income for landowners (Susaeta and Gong 2019). Loblolly pine is the leading commercial timber species in the United States (Edwards 1987, Hepting 1971, Schultz 1997, Xiong and others 2010) and has even been referred to as one of the most important timber species in the world (Wahlenberg 1960). For more than 60 years, loblolly pine has been the primary focus of many southern forest tree improvement programs (Cumbie and others 2012, Xiong and others 2010), and significant improvements in growth rate, wood volume, stem straightness, and disease tolerance have been made.

During pollen flight in early spring, longleaf and loblolly pine can cross-pollinate and produce a natural hybrid known as Sonderegger pine (*Pinus* x *sondereggeri*). Sonderegger pine usually occurs in nature as single specimens or in small groups where both parent species co-exist (Saucier and others 1979). Cones collected for longleaf pine seedling propagation in the nursery usually contain some hybrid seeds. Longleaf and Sonderegger pine seeds cannot be distinguished by visual inspection alone (Bolner and others 2019), causing many hybrids to be sown in the nursery. According to Wahlenberg (1946), Sonderegger pine seedlings can make up

Kelsey D. Shoemaker, Graduate Student, Louisiana Tech University, School of Agricultural Sciences and Forestry, Ruston, LA 71272
D. Paul Jackson, Professor, Louisiana Tech University, School of Agricultural Sciences and Forestry, Ruston, LA 71272
Joshua P. Adams, Associate Professor, Louisiana Tech University, School of Agricultural Sciences and Forestry, Ruston, LA 71272

0.5–6.0 percent of a longleaf pine seedlot, and this range may vary substantially across temporal and spatial scales (Dorman 1976). However, for decades Sonderegger pine has been considered less desirable than longleaf and loblolly pine due to reports of poor stem form, irregular branching, and susceptibility to brown-spot needle blight (*Lecanostica acicula*) and fusiform rust (*Cronartium quercuum* f. sp. *fusiforme*) (Saucier and others 1979, Schoenike and others 1975). Because of this, seedlings that exhibit hybrid characteristics in the nursery are usually culled before being packaged and shipped to landowners for outplanting. In the nursery, stem elongation of longleaf pine seedlings is the traditional indicator that hybridization between longleaf and loblolly pine has occurred (Barnett and others 2020, Jackson and others 2020).

Sonderegger pine's reputation as an unmerchantable species has prevailed for more than 100 years since its description by Chapman (1922) with virtually no support from empirical research. Furthermore, the few studies available comparing Sonderegger to loblolly or longleaf pine are now over 40 years old. For example, van Buijtenen (1969) reported that Sonderegger pine was prone to forking defects and heavy branches on an east Texas site, but presented no data to draw comparisons with longleaf or loblolly pine at the same site with similar silvicultural management. Henderson and Schoenike (1981) found no significant difference between height, diameter at breast height (d.b.h.), volume, branch diameter, stem form, or fusiform rust damage in two mixed stands of 20-year-old Sonderegger and loblolly pines. They concluded that discriminating against Sonderegger pine in stand thinnings and other management practices is unnecessary. While Henderson and Schoenike (1981) did not promote growing Sonderegger pine for such purposes, they stated that hybrid seedlings occurring naturally in southern pine nurseries could be planted satisfactorily along with loblolly pine on many sites in the coastal plain. Schoenike and others (1975) and Saucier and others (1979) believed Sonderegger pine might be suitable for pulpwood production and supported its use in southern forest tree breeding programs that select for rapid growth.

Understanding the economic and ecological values of longleaf and loblolly pine is important in developing a framework to research their hybrid. However, due to an absence of research, heritability of morphological features, wood properties, disease tolerance, and subsequent commercial value of Sonderegger pine remains unknown. Evaluating Sonderegger pine development from the seedling phase throughout maturity may lead to information that provides economic returns from stand management and improved seed efficiency for nurseries. The primary objective of this research was to document Sonderegger pine growth and morphology at age five as it relates to wood quality standards at maturity.

MATERIALS AND METHODS

On March 27, 2018, 400 container-grown Sonderegger pine seedlings were outplanted on the Catahoula Ranger District of the Kisatchie National Forest (near Pollock, LA) on an open field site with a silt-loam soil. These seedlings were operationally culled from longleaf pine container stock grown at the International Forest Company nursery (Evans, LA) during the 2017–2018 production season. Before planting, seedling height and root collar diameter were measured, and each seedling was numbered for future data reference. Seedlings were planted in eight plots, and each plot was planted with 50 seedlings on an 8-foot by 8-foot spacing with a 16-foot spacing between each plot (Jackson and others 2020).

One-year height measurements were conducted on March 1, 2019. In 2022, 4 years after outplanting, total height, d.b.h., and branch angle and diameter were measured on 109 saplings (table 1). Occurrence of stem crooks, forks, ramicorn branches, brown spot needle blight, and fusiform rust were also recorded (table 1). Twenty-four border trees in each plot (120 trees total) were excluded from the study to reduce potential bias caused by increased height and diameter growth compared to interior trees (Cancino 2005, Oliver and Larson 1996).

Correlations between seedling height at planting and height at age two, height at planting and

Table 1—Variables used in this study to measure growth and evaluate defects of 5-year-old Sonderegger pines

Variable	Unit of measure	Description and measurement method	Reference
Height	Feet	Distance from ground-line to tip of terminal bud	Schultz 1997
d.b.h.	Inches	Diameter of bole at 4.5 feet from groundline	Helms 1998
Branch diameter	Inches	Diameter of branches in the stabilization zone (2 to 7 whorls below the canopy) measured 1 inch from the bole. Branch diameter was not evaluated for branches <0.25 inches in diameter and <6 inches long.	Dorman 1976 Snyder 1961
Branch angle	Nearest degree	Angle of branches in the stabilization zone. Branch angle was not evaluated for branches <0.25 inches in diameter and <6 inches long.	Dorman 1976 Snyder 1961
Crook	Binary trait	An abrupt bend in the main bole	Helms 1998
Fork	Binary trait	A separation of the bole into two or more stems of similar sizes	Schermann and others 1997 Xiong and others 2010
Ramicorn branch	Binary trait	A steeply angled branch oriented at <30 degrees with the main stem and having a diameter distinctly less than the main bole	Schermann and others 1997 Xiong and others 2010
Brown-spot needle blight	Binary trait	Necrotic lesions on needles or premature defoliation	Enebak and Starkey 2012a
Fusiform rust	Binary trait	One or more fusoid galls on the bole or branches	Enebak and Starkey 2012b

height at age four, and height and d.b.h. at age five were calculated using Excel (Microsoft, Redmond, WA).

RESULTS

At outplanting, the average seedling height was 6.2 inches. After one growing season, the average seedling height was 10.0 inches. Taller seedlings at time of planting had a growth advantage 1 year after outplanting (r = 0.76, *P* <0.001) (fig. 1), but after 4 years in the field, the correlation between initial seedling height and subsequent height growth had weakened (r = 0.11, *P* = 0.248) (fig. 2). At age five, the trees averaged 7.7 feet tall and 1.6 inches d.b.h. (table 2). At this age, d.b.h. and total height were positively correlated (r = 0.89, *P* <0.001) (fig. 3). Of the 109 study trees, 15 (13 percent) had at least one crook or sinusoidal bole, and another 15 trees had a ramicorn branch or stem fork.

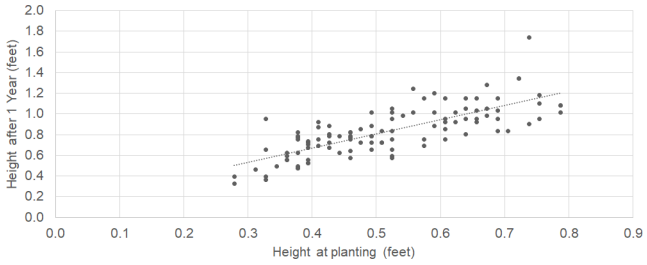

Figure 1—Correlation between seedling height at planting and height after 1 year in the field (r = 0.76).

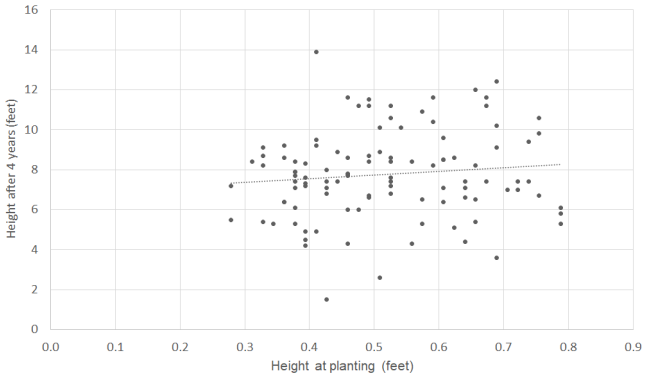

Figure 2—Correlation between seedling height at planting and height after 4 years in the field (r = 0.11).

Table 2—Summary of growth and branch characteristics reported for 5-, 9-, and 20-year-old Sonderegger pines across three study sites in the Southeast

Age	Site	Soil type	Spacing	d.b.h.	Height	Branch diameter	Branch angle
			feet	*inches*	*feet*	*inches*	*degrees*
5	Pollock, LA	Silt loam	8 by 8	1.6	7.7	0.54	72.4
9	Clemson, SC[1]	Clay loam	9 by 9	4.5	24.6	–	70.0
20	Upper coastal plain, South Carolina[2]	Sandy loam	6 by 9	11.5	58.0	2.20	67.4

[1] Source: Schoenike and others (1975).
[2] Source: Henderson and Schoenike (1981).

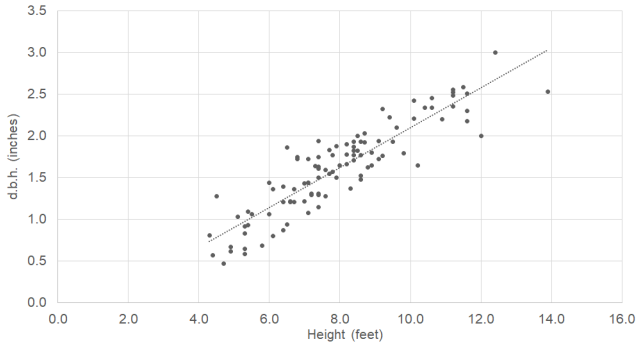

Figure 3—Correlation between height at age five and diameter at breast height (d.b.h.) (r = 0.89).

Branch angle was highly variable, ranging from 40 to 110 degrees, with a site average of 72.4 degrees. Branch diameter was also variable, ranging from 0.25 to 1.50 inches, with a site average of 0.52 inch. To our knowledge, branch diameter has never been reported for Sonderegger pines at age five.

Although fusiform rust has been reported as a primary disease for Sonderegger pine, only six trees (5 percent) had stem or branch cankers. Brown-spot needle blight incidence was noted on all trees, but most infections were minor and did not appear to affect tree vigor.

DISCUSSION

Past observations that indicated Sonderegger pine as having large branches or poor form (Dorman 1976, van Buijtenen 1969) did not account for the silvicultural components that may have influenced these poor growth responses. Philip Wakeley, in a personal communication with Schoemike and others (1975), stated that the large diameter branches of Sonderegger pine made it less desirable than the pure species. However, the crown morphology he observed may have been influenced by the open-grown environment created in a stand of slower growing longleaf pine (Schoenike and others 1975).

Similarly, the natural pruning ability of any tree species is influenced by silvicultural practices such as planting density and fire frequency as well as genetic and environmental interactions (Wright 1976). For example, longleaf pine is one of the best natural pruners of all the southern pines (Wahlenberg 1946), but branch size and angle of orientation can vary substantially in open-pollinated families (Dorman 1976). In loblolly pine, natural pruning may be delayed in under-stocked stands and can subsequently reduce the merchantable volume of wood produced at harvest (Dorman 1976, Schultz 1997). While the genetic and environmental interactions that influence natural pruning ability and knot development in Sonderegger pine are still unknown, Chapman (1922) predicted that the limbs of Sonderegger pine would be less persistent and grow less vigorously than loblolly pine, most likely resulting in the self-pruning of branches as the tree matures in a fashion similar to longleaf pine.

Although conclusions cannot be drawn at this time, mean branch angle at this site was similar to findings from previous studies involving 9- and 20-year-old Sonderegger pines in South Carolina, which may indicate that branch orientation is less dependent on tree age than other factors (table 2). Henderson and Schoenike (1981) found that 20-year-old Sonderegger pine had a branch angle of 67.4 degrees when grown in mixed stands of loblolly pine in South Carolina. In the same study, mean branch angle was found to be less acute in Sonderegger pine than loblolly pine, indicating that Sonderegger pine had the more desirable branch attribute (table 2).

Further research is needed to draw conclusions on branch characteristics and their impacts on woody quality of Sonderegger pine. Studies that evaluate Sonderegger pine's outplanting performance, fire tolerance, wood quality, and potential nontimber values will also add to our understanding of this naturally occurring hybrid.

CONCLUSIONS

It was surmised decades ago that Sonderegger pine was not commercially viable. Despite the lack of research to support this conclusion, Sonderegger pine seedlings have traditionally been culled from the nursery to prevent them from being planted in southern forest plantations.

Studies have reported that Sonderegger pine's growth and morphology is similar to loblolly pine (Henderson and Schoenike 1981, Schoenike and others 1975), but the consensus surrounding the hybrid's reputation has not changed for more

than a century. Although conclusions on branch diameter and knot development cannot be drawn at this time, this study showed that branch orientation at age five was similar to findings for 20-year-old Sonderegger and loblolly pine on a South Carolina site (Schoenike and others 1975). Furthermore, while Sonderegger pine has historically been considered highly susceptible to brown-spot needle blight and fusiform rust (Saucier and others 1979), disease incidence was low at our site.

If Sonderegger pine has commercial value, economic returns for nursery owners may be possible with improved seed efficiency. However, financial gains may not be fully realized unless Sonderegger pine development is examined across various sites and compared with the performance of longleaf and loblolly pine.

ACKNOWLEDGMENTS

We would like to thank James Barnett, Billy Bruce, Jaesoon Hwang, Wood Johnson, James Meeker, Rabiu Olatinwo, Aaron Rachal, Karen Reed, Alan Springer, Chris Steiner, and Brian Strom (USDA Forest Service, Southern Research Station and Forest Health Protection) for their technical support during this ongoing project. We also thank the International Forest Company tree seedling nursery in Evans, LA, for providing the Sonderegger pines used in this study.

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. https://doi.org/10.2737/SRS-GTR-56.

Barnett, J.P.; Olatinwo, R.; Jackson, D.P.; Blomquist, S. 2020. Longleaf pine hybridization: is there a growing problem? In: Bragg, D.C.; Koerth, N.E.; Holley, A.G., eds. Proceedings of the 20th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-253. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 128–134. https://www.fs.usda.gov/research/treesearch/60908.

Bolner, N.G.; Jackson, D.P.; Olatinwo, R.; Barnett, J.P. 2019. Evaluation of sowing methods to determine the role of hypocotyl lift in longleaf pine seedling development. Tree Planters' Notes. 62(1&2): 88–94. https://www.fs.usda.gov/research/treesearch/59713.

Cancino, J. 2005. Modelling the edge effect in even-aged Monterey pine (*Pinus radiata* D. Don) stands. Forest Ecology and Management. 210: 159–172. https://doi.org/10.1016/j.foreco.2005.02.021.

Chapman, H. H. 1922. A new hybrid pine (*Pinus palustris* × *Pinus taeda*). Journal of Forestry. 20: 729–734. https://doi.org/10.1093/jof/20.7.729.

Cumbie, W.P.; Isik, F; McKeand, S.E. 2012. Genetic improvement of sawtimber potential in loblolly pine. Forest Science. 58(2): 168–177. https://doi.org/10.5849/forsci.09-060.

Dorman, K.W. 1976. The genetics and breeding of southern pines. Agric. Handb. 471. Washington, DC: U.S. Department of Agriculture Forest Service. 407 p. https://naldc.nal.usda.gov/download/CAT87208699/pdf.

Edwards, M.B. 1987. Natural regeneration of loblolly pine: A loblolly pine management guide. Gen. Tech. Rep. SE-47. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 17 p. https://doi.org/10.2737/SE-GTR-47.

Enebak, S.A.; Starkey, T. 2012a. Brown spot needle blight. In: Cram, M.M.; Frank, M.S.; Mallams, K.M., tech. coords. Forest nursery pests. Agric. Handb. 680. Washington, DC: U.S. Department of Agriculture Forest Service: 28–30.

Enebak, S.A.; Starkey, T. 2012b. Fusiform rust. In: Cram, M.M.; Frank, M.S.; Mallams, K.M., tech. coords. Forest nursery pests. Agric. Handb. 680. Washington, DC: U.S. Department of Agriculture Forest Service: 41–43.

Helms, J.A. 1998. The dictionary of forestry. Bethseda, MD: Society of American Foresters. 210 p.

Henderson, L.T., Jr.; Schoenike, R.E. 1981. How good is Sonderegger pine? Southern Journal of Applied Forestry. 5(4): 183–186. https://doi.org/10.1093/sjaf/5.4.183.

Hepting, G.H. 1971. Diseases of forest and shade trees of the United States. Agric. Handb. 386. Washington, DC: U.S. Department of Agriculture Forest Service. 658 p.

Jackson, D.P.; Olatinwo, R.; Barnett, J.P. [and others]. 2020. Field observations of longleaf pine seedlings to examine possible hybridization. In: Bragg, D.C.; Koerth, N.E.; Holley, A.G., eds. Proceedings of the 20th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-253. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 137–141. https://www.fs.usda.gov/research/treesearch/60905

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. Update ed. New Haven, CT: Yale University. 520 p.

Saucier, J.R.; Henderson, L.T.; Schoenike, R.E. 1979. Wood properties of plantation-grown Sonderegger pine. Res. Note SE-272. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 4 p.

Schermann, W.T.; Adams, W.T.; Aitken, S.N.; Bastien, J.-Ch. 1997. Genetic parameters of stem form traits in 9-year-old coastal Douglas-fir progeny test in Washington. Silvae Genetica. 46(2): 166–170.

Schoenike, R.E.; Hart, J.D.; Gibson, M.D. 1975. Growth of a nine-year-old Sonderegger pine plantation in South Carolina. Silvae Genetica. 24(1): 10–11.

Schultz, R.P. 1997. Loblolly pine: The ecology and culture of loblolly pine (*Pinus taeda* L.) Agric. Handb. 713. Washington, DC: U.S. Department of Agriculture Forest Service. [irregular pagination].

Snyder, E.B. 1961. Measuring branch characters of longleaf pines. Occas. Pap. 184. New Orleans, LA: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 5 p.

Susaeta, A; Gong, P. 2019. Economic viability of longleaf pine management in the southeastern United States. Forest Policy and Economics. 100: 14–23. https://doi.org/10.1016/j.forpol.2018.11.004.

van Buijtenen, J.P. 1969. Seventeenth progress report of the cooperative forest tree improvement program. Circ. 200. U.S. Department of Agriculture Forest Service. 14 p.

Wahlenberg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth, and management. Washington, DC: Charles Lathrop Pack Forestry Foundation in cooperation with U.S. Department of Agriculture Forest Service. 429 p.

Wahlenberg, W.G. 1960. Loblolly pine: its use, ecology, regeneration, protection, growth, and management. Durham, NC: Duke University, School of Forestry, in cooperation with forest industry and U.S. Department of Agriculture Forest Service. 603 p.

Wright, J.W. 1976. Introduction to forest genetics. New York: Academic Press. 480 p.

Xiong, J.S.; Isik, F.; McKeand, S.E.; Whetten, R.W. 2010. Genetic variation of stem forking in loblolly pine. Forest Science. 56(5): 429–436. https://doi.org/10.1093/forestscience/56.5.429.

Longleaf Pine

DISTURBANCE AND STAND STRUCTURE IN A LONGLEAF PINE WOODLAND

David L. Phillips, J. Davis Goode, and Justin L. Hart

EXTENDED ABSTRACT

All stands experience disturbance events that impact succession and development. Most mortality in longleaf pine (*Pinus palustris*) ecosystems consists of localized, gap-scale events composed of one to several canopy trees (Mueller and others 2022, Palik and Pederson 1996) that free resources including light, soil nutrients, moisture, and growing space (Hart and Kleinman 2018, Palik and others 1997). Available resources allow for tree establishment and recruitment that may result in a mosaic of age and size classes (Richards and Hart 2011, Varner and others 2003). We quantified the stand structure, composition, canopy disturbance history, and prescribed fire effects in a complex-stage reference longleaf pine stand in the desired future condition as identified by U.S. Department of Agriculture, Forest Service personnel (multiple age classes of canopy longleaf pine, presence of advance reproduction, and desired species composition). Canopy disturbance history data such as these can be used to inform current management of longleaf pine ecosystems, and quantifying prescribed fire mortality can provide insight into potentially undesirable fire effects.

The chosen stand was located in the Alabama Fall Line Hills, within the Oakmulgee Ranger District of the Talladega National Forest, AL. The stand was entered in 1979 for timber stand improvement, 1990 for midstory removal to enhance red-cockaded woodpecker (*Leuconotopicus borealis*) foraging habitat, and 1997 for a thinning. Prescribed fire occurred in 1963, 1976, 1980, 1993, 1997, 2000, 2004, 2015, 2019, and 2021. Within a 1-ha plot in the stand, we recorded location (using a sighting compass and Haglof hypsometer and transponder, Sweden), species, height (using a hypsometer and transponder and measured to the nearest 0.1 m), and diameter at breast height (d.b.h., using a diameter tape and measured to the nearest 0.1 cm) of all trees with d.b.h. ≥5 cm and longleaf pine saplings (height ≥1 m, d.b.h. <5 cm). Additionally, we recorded the species and quadrat for saplings of other species. We extracted a single tree core (from outer bark to pith) from every tree using a Haglof increment borer about 50 cm above the ground and parallel to the slope to avoid reaction wood in the tree cores (Hart and others 2010). Tree-ring samples were processed in accordance with the methods of Stokes and Smiley (1996). Cores were air dried for 48 hours, attached to wooden mounts with cells vertically aligned, and sanded using progressively finer grits. Tree rings

David L. Phillips, Graduate Student, University of Alabama, Tuscaloosa, AL 35487
J. Davis Goode, Graduate Student, University of Alabama, Tuscaloosa, AL 35487
Justin L. Hart, Lab Director, University of Alabama, Tuscaloosa, AL 35487

were then dated to their calendar year of formation. We measured raw ring width for all canopy trees to the nearest 0.001 mm using a Velmex measuring stage (Velmex Inc., 2009, Bloomfield, NY) and Measure J2X software (version 4.2, VoorTech Consulting, 2008, Holderness, NH). We used standard dendroecological techniques to reconstruct canopy disturbance history and age structure within the study area. We applied the 10-year running mean method (Nowacki and Abrams 1997) to detect release events (growth 25 percent greater than the mean of the preceding and succeeding 10 years, sustained for a minimum of 3 years; Hart and Grissino-Mayer 2008). To capture a contemporary disturbance impact, we recorded mortality approximately 6 months after a compartment-wide (401-ha) prescribed burn in the spring of 2021.

We observed 11 different species in the tree layer within the 1-ha plot. Longleaf pine dominated the study area, with loblolly pine (*P. taeda*) and blackjack oak (*Quercus marilandica*) having the second and third greatest basal area, respectively (table 1). Of the 189 overstory individuals analyzed for disturbance history, 163 trees (86 percent) exhibited at least one release event, and 84 trees (52 percent) exhibited multiple release events (fig. 1). Three plot-wide release events were documented. The first two events (1878–1885 and 1908–1913) were based on only five trees, three of which experienced the release events. The third release event occurred during 1967–1978 and included over 60 percent of the 160 trees at that time.

We observed mortality in 9 of the 394 trees (2 percent) and 12 of the 57 longleaf pine saplings (21 percent). Of the nine trees, four were longleaf pine, one was loblolly pine, two were turkey oak (*Quercus laevis*), and two were blackjack oak. The nine trees had an average d.b.h. of 10.4 cm, with

Table 1—Density, dominance, and relative importance (average of relative density and relative dominance) of trees in a longleaf pine woodland in the Oakmulgee Ranger District of the Talladega National Forest, AL

Species	Density	Relative density	Dominance	Relative dominance	Relative importance
	stems/ha	*percent*	*m^2/ha*	*percent*	
Pinus palustris	350	88.83	19.5342	93.99	91.41
Quercus marilandica	14	3.55	0.2971	1.43	2.49
Pinus taeda	9	2.28	0.4514	2.17	2.23
Nyssa sylvatic	6	1.52	0.1994	0.96	1.24
Quercus laevis	7	1.78	0.0735	0.35	1.07
Diospyros virginiana	2	0.51	0.1154	0.56	0.53
Pinus echinata	1	0.25	0.0625	0.30	0.28
Liquidambar styraciflua	2	0.51	0.0069	0.03	0.27
Quercus velutina	1	0.25	0.0373	0.18	0.22
Pinus virginiana	1	0.25	0.0029	0.01	0.13
Oxydendrum arboretum	1	0.25	0.0024	0.01	0.13
Total	394	100.00	20.7829	100.00	100.00

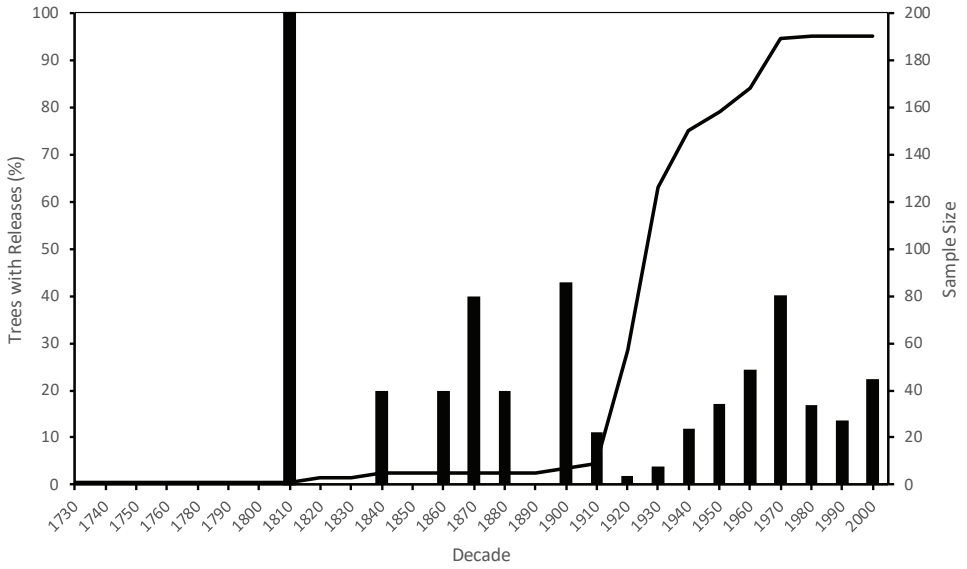

Figure 1—Release events documented in 189 canopy trees in a longleaf pine woodland in the Oakmulgee Ranger District of the Talladega National Forest, AL. Bars represent the percentage of trees in each decade that experienced a release event. The line represents the total sample size in any given decade.

a minimum of 5.2 cm and a maximum of 32.8 cm (both longleaf pine). The 12 saplings had an average d.b.h. of 3.8 cm and an average height of 3.9 m, just below the averages for all longleaf pine saplings (3.9 cm and 4 m, respectively).

In the tree layer, stems other than longleaf pine suffered a greater percentage of mortality (5 out of 44, 11 percent) than longleaf pine (4 out of 350, 1 percent). Although there was a 21-percent mortality rate for longleaf pine saplings, it is likely that other saplings suffered a much greater mortality rate. Further, we did not sample the seedling layer, where additional longleaf pine reside. These findings provide details on the natural patterns of disturbance and may be used to inform silvicultural systems intended to enhance ecosystem resilience of longleaf pine woodlands.

LITERATURE CITED

Hart, J.L.; Austin, D.A.; van de Gevel, S.L. 2010. Radial growth responses of three co-occurring species to small canopy disturbances in a secondary hardwood forest on the Cumberland Plateau, Tennessee. Physical Geography. 31: 270–291. https://doi.org/10.2747/0272-3646.31.3.270.

Hart, J.L.; Grissino-Mayer, H.D. 2008. Vegetation patterns and dendroecology of a mixed hardwood forest on the Cumberland Plateau: implications for stand development. Forest Ecology and Management. 255: 1960–1975. http://dx.doi.org/10.1016/j.foreco.2007.12.018.

Hart, J.L.; Kleinman, J.S. 2018. What are intermediate-severity forest disturbances and why are they important? Forests. 9: 579. 15 p. https://doi.org/10.3390/f9090579.

Mueller, H.L.; Goode, J.D.; Hart, J.L 2022. Gap-scale disturbance patterns and processes in a montane *Pinus palustris* woodland. Forests. 13: 1169. 19 p. https://doi.org/10.3390/f13081169.

Nowacki, G.J.; Abrams, M.D. 1997. Radial-growth averaging criteria for reconstructing disturbance histories from presettlement-origin oaks. Ecological Monographs. 67: 225–249. https://doi.org/10.2307/2963514.

Palik, B.J.; Mitchell, R.J.; Houseal, G.; Pederson, N. 1997. Effects of canopy structure on resource availability and seedling responses in a longleaf pine ecosystem. Canadian Journal of Forest Research. 27: 1458–1464. https://doi.org/10.1139/x97-081.

Palik, B.J.; Pederson, N. 1996. Overstory mortality and canopy disturbances in longleaf pine ecosystems. Canadian Journal of Forest Research. 26(11): 2035-2047. https://doi.org/10.1139/x26-229.

Richards, J.D.; Hart, J.L. 2011. Canopy gap dynamics and development patterns in secondary *Quercus* stands on the Cumberland Plateau, Alabama, USA. Forest Ecology and Management. 262: 2229–2239. https://doi.org/10.1016/j.foreco.2011.08.015.

Stokes, M.A.; Smiley, T.L. 1996. An introduction to tree-ring dating. Tuscon, AZ: University of Arizona Press. 73 p.

Varner, J.M. III; Kush, J.S.; Meldahl, R.S. 2003. Structural characteristics of frequently-burned old-growth longleaf pine stands in the mountains of Alabama. Castanea. 68(3): 211–221.

DOES LONGLEAF PINE CONE PRODUCTION CORRELATE WITH SEED PRODUCTION, SIZE, AND GERMINATION RATE?

Rachel E. Nation, Heather D. Alexander, John L. Willis, Michael Aspinwall, and Jeffery B. Cannon

EXTENDED ABSTRACT

Fire-dependent longleaf pine (*Pinus palustris*) ecosystems in the Southeastern United States require viable seed sources for successful natural regeneration. Most studies focus on cone production as a relatively easy-to-measure variable. However, there is little understanding of whether cone production is a reliable proxy for both seed production and seed viability, which may be decoupled at high cone production levels. Identifying relationships between cone production, seed production, seed size, and germination rate should provide managers and landowners better information to predict the success of natural regeneration efforts more accurately in their longleaf pine stands. The objectives of our study were to 1) determine the relationship between cone production and seed production across individual mature longleaf pine trees, 2) determine whether cone production and seed size characteristics are related across mature longleaf pine trees, and 3) determine whether seed germination rates vary as a function of cone production across mature longleaf pine trees.

To better understand cone-seed relationships, we conducted two rounds of longleaf pine seed collection from trees representing a gradient of cone production during seedfall (mid-October to early December) (Boyer 1963). In 2021, we collected seeds from 10 trees in a longleaf pine stand at The Jones Center at Ichauway (Newton, GA) using ten 0.25-m^2 seed traps placed under each tree. In 2022, we collected seeds in the same stand at The Jones Center at Ichauway and in one stand each at Blackwater River State Forest (Milton, FL) and Eglin Air Force Base (Fort Walton Beach, FL). The 2022 collections were also from 10 trees in each stand, but we used three 3-m^2 seed traps placed under each tree (fig. 1). We tallied seeds to determine seed production and measured seed length, width, and mass to explore seed size trends and their relationship to cone production and seed

Rachel E. Nation, Graduate Student, Auburn University, College of Forestry, Wildlife and Environment, Auburn, AL 36849

Heather D. Alexander, Associate Professor, Auburn University, College of Forestry, Wildlife and Environment, Auburn, AL 36849

John L. Willis, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Auburn, AL 36849

Michael Aspinwall, Affiliate Faculty, Auburn University, College of Forestry, Wildlife and Environment, Auburn, AL 36849

Jeffery B. Cannon, Forest Landscape Ecologist, The Jones Center at Ichauway, Newton, GA 39870

Figure 1—Seed traps placed under a longleaf pine tree at The Jones Center at Ichauway, Newton, GA, in 2022. (Photo courtesy of Rachel Nation)

viability. We also conducted germination trials with the collected seeds in a growth chamber under controlled conditions (20 °C, 16:8 hour light:dark) (AOSA 1993).

Overall, we found that the density of total seeds, developed seeds (wing with seed present), and undeveloped seeds (wing with no signs of viable seed present) all increased with increasing cone production before declining, following a quadratic model. Total seed density increased with cone production reaching a maximum of 120 total seeds collected per square meter trap area at 100 cones per tree before declining with further increases in cone counts. Density of developed seeds also increased with cone production reaching a maximum of 80 developed seeds collected per square meter trap area at 110 cones per tree before declining with higher cone counts. Undeveloped seed density increased with cone production reaching a maximum of 40 undeveloped seeds collected per square meter trap area at 100 cones per tree before declining with higher cone counts.

There were no significant relationships between any of the seed size metrics (seed length, seed width, total length, wing width, total mass, and seed mass) and cone production. We observed that the relationship between probability of germination and seed mass followed a logistic pattern. In addition, we found no definite relationship between cone production and seed germination in longleaf pine seeds.

LITERATURE CITED

Association of Official Seed Analysts [AOSA]. 1993. Rules for testing seeds. Journal of Seed Technology. 16(3): 1–113.

Boyer, W.D. 1963. Longleaf pine seed dispersal. Res. Note SO-3. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 2 p.

EFFECTS OF FERTILIZATION AND ENDOPHYTE APPLICATION ON GROWTH OF PLANTED LONGLEAF PINE

Baylor Doughty, Adam Polinko, and Courtney Siegert

ABSTRACT

Longleaf pine (*Pinus palustris*) is an important economic and ecological species in the Southeast. The removal of fire from southeastern forests and a landowner preference for loblolly pine (*P. taeda*) and slash pine (*P. elliottii*) has resulted in a major decrease of the range of longleaf pine. This study investigates the potential of two fertilization methods and inoculation with diazotrophic endophytes to reduce time in the grass stage and improve diameter growth in longleaf pine. The study area was located in Clarke County, MS, on soils that consisted of a mix of fine sands and sandy clays. A total of four treatments were applied in a randomized complete block design to a recently clearcut stand. The site was chemically prepped prior to establishment. Treatments included control, broadcast fertilizer [nitrogen (N)-phosphorus (P)-potassium (K) 32-14-5] applied at a rate of 35 pounds per acre, and controlled release fertilizer bags (NPK 19-8-3) applied adjacent to each seedling. Diazotrophic endophytes were also applied in a separate treatment adjacent to each planted seedling 1 year following establishment. We measured root collar diameter (r.c.d.), survival, and the number of seedlings that had initiated height growth. Overall r.c.d. and grass stage emergence did not show a significant difference during second year measurements for both fertilization treatments and endophyte inoculation. Survival, however, was significantly lower in the broadcast fertilization treatment compared to the control. Increased competition from shrubs, grasses, and naturally regenerated loblolly pine in broadcast fertilization plots likely contributed to the low survival. Reducing time spent in the grass stage may help re-establish longleaf pine to its native range and increase ecological diversity across the Southeast.

INTRODUCTION

Longleaf pine (*Pinus palustris*) is an ecologically important species in the Southeast (Jose and others 2006). Longleaf pine has lost 95 percent of its historical range due to fire exclusion, its preference for use in naval store production, and landowner preference for loblolly pine (*P. taeda*) and slash pine (*P. elliottii*) due to their quick initial growth and flexibility of use across different site conditions. Longleaf pine exhibits slower initial growth due to the "grass stage," an adaptation to fire where most resources are allocated to root development (Haywood 2000, Ramsey and others 2003). This stage can last from 1 to 15 years (Boyer 1990). Emergence from the grass stage is dependent on genetics, micro- and macro-site conditions, and competition. Grass stage seedlings generally emerge after reaching a root collar diameter (r.c.d.) of 0.5 inches in the absence of competition, though there is considerable variation in that threshold (Ramsey and others 2003). Following the grass stage, a period of rapid height growth and limited lateral expansion may occur.

Competition control is essential to successful establishment of longleaf pine. The use of herbicides alone to control competing vegetation can dramatically influence time in the grass stage (Haywood 2000, Ramsey and others 2003).

Baylor Doughty, Graduate Student, Mississippi State University, Department of Forestry, Forest and Wildlife Research Center, Starkville, MS 39762

Adam Polinko, Assistant Professor, Mississippi State University, Department of Forestry, Forest and Wildlife Research Center, Starkville, MS 39762

Courtney Siegert, Associate Professor, Mississippi State University, Department of Forestry, Forest and Wildlife Research Center, Starkville, MS 39762

Fertilization has the potential to reduce this time but only when competition is reduced or eliminated (Derr 1957, Haywood 2007). Broadcast spraying is the most widespread method of fertilizer application but often results in increased nontree competition (Powers and Ferrell 1996). Localized methods of fertilization, like controlled release point source fertilizers (hereafter "teabag") may improve seedling growth while limiting nutrient availability to nontarget vegetation (Thiffault and others 2005).

Inoculation with diazotrophic endophytes may be another method of reducing time in the grass stage. Endophytes are bacterial communities first derived from poplar (*Populus* spp.) and willow (*Salix* spp.) but later derived from Douglas-fir (*Pseudotsuga menziesii*) and western redcedar (*Thuja plicata*) in the Pacific Northwest. The role of endophytes in plant growth is unclear; however, endophyte inoculations have the potential to improve productivity (Aghai and others 2019, Chanway and Holl 1991) and stress tolerance (Khan and others 2016).

Reducing time spent in grass stage may provide opportunities to shorten rotation length and thus increase adoption of longleaf pine across the landscape. The objective of this study was to evaluate the influence of different fertilization treatments and endophyte inoculation on growth and survival. Specifically, our objectives were to 1) evaluate longleaf pine seedling response to broadcast and teabag fertilization application methods, and 2) quantify response to inoculation of diazotrophic endophytes.

METHODS

The study was located in Clarke County, MS. The property was owned by the Mississippi State Bulldog Forest Foundation. The site was clearcut in March 2020. Prior to harvest, the stand was composed of shortleaf (*Pinus echinata*), loblolly, and longleaf pines in mixtures of hardwoods. The site index was 85 feet (base age 50) with soils consisting of fine sands to silt loams (Lauderdale-Eustis complex) and Shubuta fine sandy loam. The site had slightly rolling topography that resulted in erosion, which formed sandy fans and deep cuts following harvest of some areas.

Chemical site preparation was applied using backpack sprayers in September 2020. Multiple waxy-leafed species were present on site including American holly (*Ilex opaca*) and yaupon (*I. vomitoria*) which are known to resist chemical application, warranting extra consideration for these species. Sprayers were filled with an herbicide mixture containing a solution of 4 pounds imazapyr (2.5 percent), glyphosate (5 percent), nonionic surfactant (0.25 percent), blue dye (0.4 percent), and water (92.1 percent). Herbicide was spot sprayed at a rate not exceeding labelled broadcast rates.

Containerized longleaf pine seedlings were planted by hand in April 2021 at a 10-foot by 10-foot spacing. Following planting, treatments were applied in plots containing 100 trees in a complete block design. Treatments were assigned randomly except for broadcast fertilizer treatments which were oriented to avoid precipitation runoff into other treatment blocks. Treatments were replicated four times in blocks oriented parallel to the slope contour. Treatments included applications of broadcast and controlled release teabag fertilizer, a fungal endophyte treatment, and a no treatment control.

Fertilizer treatments were applied following planting in April 2021. The first treatment utilized broadcast fertilizer applied at a rate of 35 pounds N per acre. (NPK 32-14-5). Broadcast fertilizers were weighed, bagged, and applied using a hand spreader and cart spreader such that the rate was distributed across the plot. The second treatment utilized Reforestation Technologies International (RTI) Silva Pak Worm Blend controlled release fertilizer bags (NPK 19-8-3). Teabags were inserted 2–3 inches deep on the uphill side and approximately 0.5 inches from the seedling using a dibble bar. Diazotrophic endophytes were applied in February 2022. Granular endophytes were applied top dressed in close proximity to each seedling, individually and manually at a rate of one-half teaspoon per seedling.

Trees in each treatment were measured in February 2022 and 2023. To avoid edge effect, only the inner 80 trees were measured for each plot. Measurements included r.c.d. and survival. Emergence from the grass stage was recorded in year 2. Analysis was performed using the

R statistical programming language (R Core Team 2023). A nested linear mixed effects model was developed to analyze r.c.d. To analyze both survival and emergence from the grass stage, a nested mixed effects logistic regression was developed using the lme4 package (Bates and others 2015). For both models, treatment was a fixed effect and block was a random effect.

RESULTS

Second-year survival ranged from 73.5 percent in the broadcast fertilizer treatment to 84.1 percent in the control and was significantly different between control and broadcast treatments (P <0.05, fig. 1). Root collar diameter was statistically similar between treatments (P >0.05, fig. 2) and ranged from 1.28 to 1.49 inches. The proportion of seedlings that had emerged from the grass stage 2 years posttreatment was statistically similar between treatments (mean = 10.6 percent, P >0.05, fig. 3). Teabag fertilization and endophyte inoculation had a considerable amount of variation compared to the control.

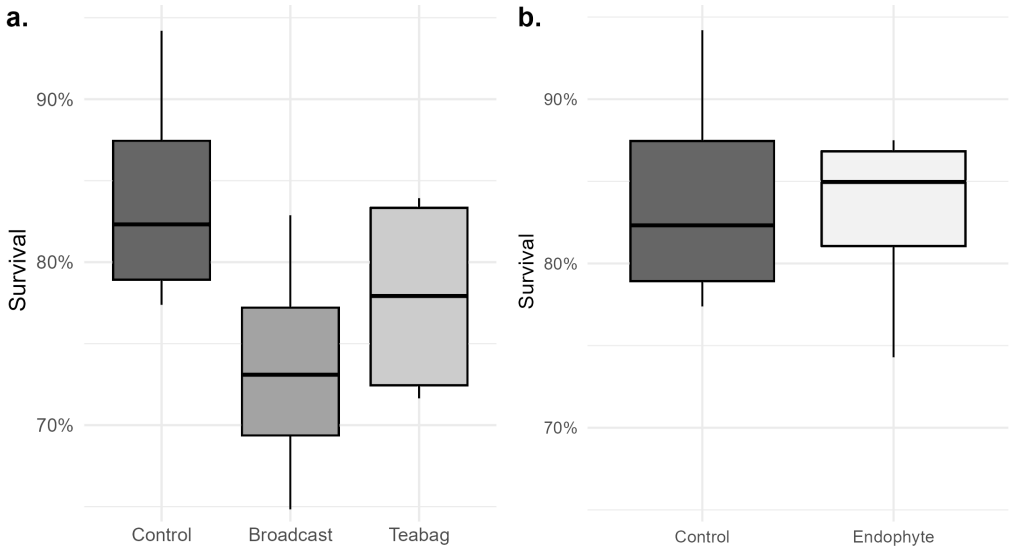

Figure 1—Precent survival of planted longleaf pine in central Mississippi (a) 2 years following treatment of fertilizer and (b) 1 year following endophyte inoculation.

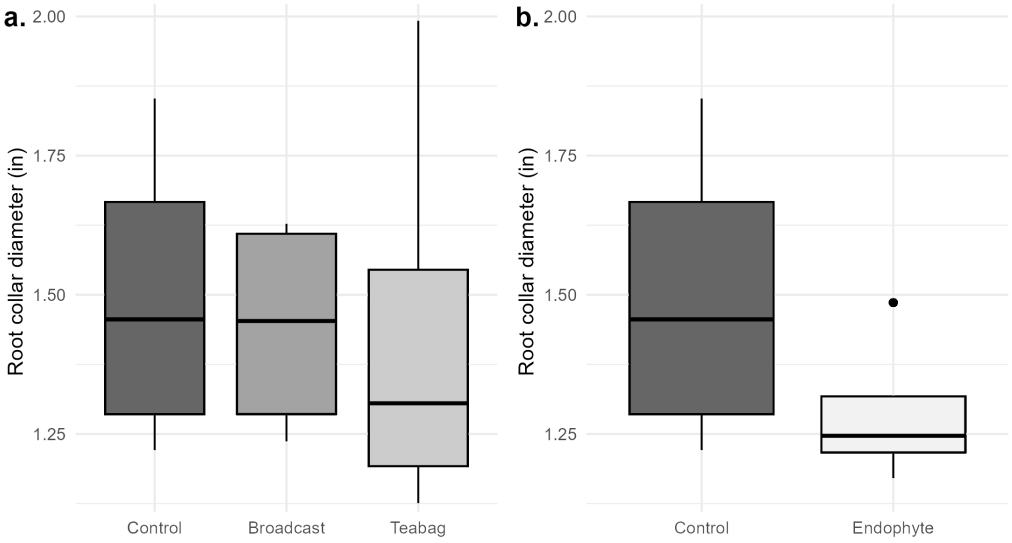

Figure 2—Root collar diameter (inches) of planted longleaf pine in central Mississippi (a) 2 years following treatment of fertilizer and (b) 1 year following endophyte inoculation.

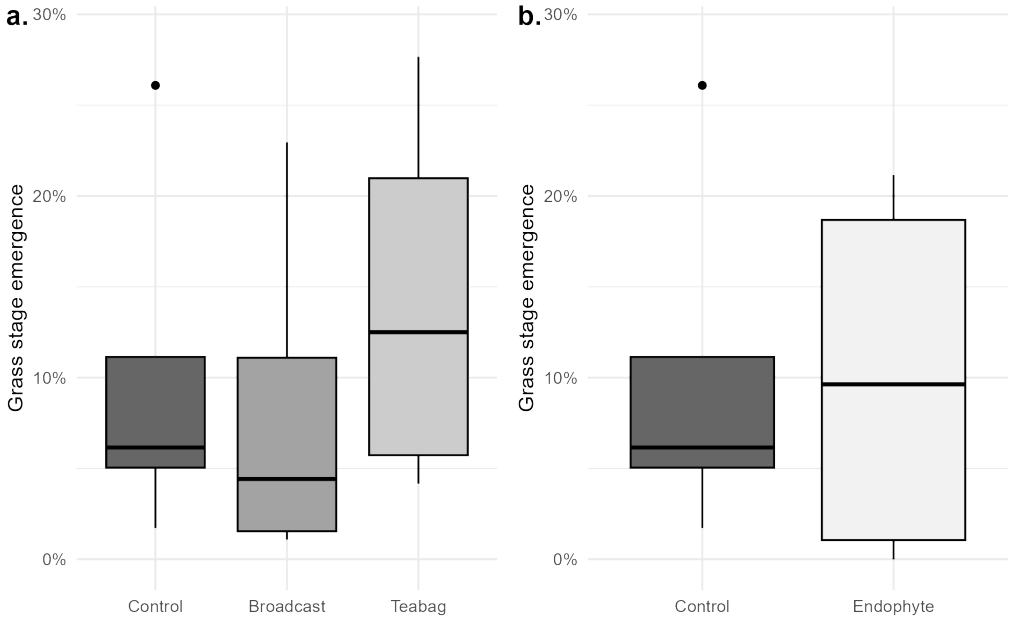

Figure 3—Proportion of grass stage emergence of planted longleaf pine in central Mississippi (a) 2 years following treatment of fertilizer and (b) 1 year following endophyte inoculation.

DISCUSSION

Overall, the results indicated that there was no significant difference between emergence from the grass stage and r.c.d. 2 years posttreatment among the different fertilizer treatments used in this study. However, broadcast fertilization resulted in a significant decrease in survival of seedlings which may have long-term implications for growth and stand development. Application of endophytes produced no growth or survival improvement 1 year following treatment.

While we did not detect significant trends in seedling growth associated with fertilization, seedling survival in the broadcast fertilizer treatment was significantly lower compared to the control. The decreased survival found in the broadcast fertilizer treatment may be associated with increased competition associated with fertilization, where neighboring vegetation likely capitalized on increased nutrient availability. We did not measure nontarget vegetation in this study. Earlier studies of fertilization and competition control in longleaf pine have demonstrated a neutral to negative interaction between fertilization and competing vegetation. For example, survival was significantly lower in fertilized plots and emergence from the grass stage was best predicted by absence of neighboring competition in the Florida panhandle (Ramsey and others 2003). In central Louisiana, a lack of response to fertilization was thought to be a result of increases in competing vegetation (Haywood 2007). In our study, application of chemical site preparation the same year as harvest likely increased competing vegetation as application did not maintain control of competing vegetation throughout the following growing season. Though we found no statistical evidence for growth differences to fertilization 2 years posttreatment, long-term growth and development of longleaf pine trees on plots treated with broadcast fertilizer may be significantly impacted, as competing vegetation can reduce growth and increase mortality.

Like many species, fertilization response in longleaf pine seedlings varies by site, application rate, and existing competition. Root collar diameter was unaffected by applications of fertilizer or endophytes. Ramsey and others (2003) hypothesized that limited response to fertilization in longleaf pine seedlings resulted from poor taproot development associated with lateral root growth stimulated by fertilization or that root development does not require nutrients at the same level of older trees that are putting on both primary and secondary growth. Young grass-stage seedlings do not turn over new needles as quickly as older seedlings and thus are likely not capitalizing as much as older species

or species with more rapid initial height growth. The containerized seedlings used in this study were fertilized during nursery production which likely confounds our results, as fertilization during production can have lasting impacts on survival and growth. Competing vegetation capitalizing on increased nutrient levels may also negatively influence r.c.d. which may explain some of our results. Nutrient uptake in deficient stands can result in increased water use efficiency (Anderson and Johnsen 2009). Variation in soil quality caused by erosion on this site may result in greater heterogeneity in seedling performance, because soil texture is also related to nutrient availability, water holding capacity, and the presence of competing vegetation.

Height growth response to longleaf pine fertilization appears to be directly affected by the level of competition. Derr (1957) only found a fertilizer response in plots that received intensive vegetation control while Ramsey and others (2003) found no fertilizer effect on height growth. Though we did not measure height growth, the initiation of height growth (emergence from the grass stage) was unaffected by either treatment. It is notable that the teabag treatment, though not statistically significant, had a higher percentage of seedlings that had initiated height growth compared to all other treatments. Endophyte applications also appear to have a significant amount of variation relative to the control.

Logistical challenges prevented application of endophytes that coincided with the time of fertilizer application. Unsurprisingly, there was no statistical treatment effect just 1 year following treatment which is consistent with other studies (Aghai and others 2019). It is notable, however, that variability and mean of grass stage emergence was higher compared to broadcast and control treatments. Studies of endophyte inoculation have reported increases in survival, growth, and water use efficiency in western conifers (Aghai and others 2019; Khan and others 2015, 2016; Padda and others 2021; Renninger and others 2021). However, many questions remain as to the influence of endophyte inoculation on seedlings as well as nontarget vegetation.

Studies comparing the effect of broadcast applications to point source, controlled-release fertilizers like teabags are rare. Notably, teabag fertilization did not result in a decrease in survival in this study though mean survival was lower than the control. There is some indication that these targeted methods do result in more efficient uptake of available N (Hangs and others 2003). The differences in application rates between fertilization methods may confound some of our treatment results. As each teabag weighs 0.71 ounces, applying an equivalent broadcast amount would result in an application rate of 1.5 pounds per acre; a rate that is likely insufficient to produce a desired response based on preliminary soil analysis. Further studies will be necessary to evaluate appropriate application rates of targeted fertilizers compared to traditional methods.

CONCLUSIONS

Fertilizer did not improve growth rate or the proportion of seedlings that emerged from the grass stage. We also found no effect of endophyte inoculation 1 year posttreatment. Broadcast fertilization, however, significantly reduced seedling survival while point source teabag fertilizers were similar to the control. Competition from herbaceous vegetation and naturally regenerated loblolly pine likely influenced treatment outcomes in this study. Successful competition control is essential to establishing fast growing longleaf pine.

ACKNOWLEDGMENTS

This project was funded by the Mississippi State University Office of Research and Economic Development Undergraduate Research Program and the Mississippi State University College of Forest Resources Undergraduate Research Scholars Program. RTI provided teabag fertilizers and Intrinsyx Bio provided endophytes. We want to give special thanks to Elizabeth Baach, Zac Chandler, Macy Gosselaar, Austin Himes, Bruno Kanieski da Silva, Kyle Dues, Max Schrimpf, Molly Graham, Justin Gleason, John Brown, Drew Fletcher, Quentin Boccaleri, Jack Fair, Ret Tanner, Jonathan Stoll, Garrett Hendrix, Matthew Harrison, J.P. Cromwell, and College of Forest Resources Forest Operations employees for their assistance throughout the project.

LITERATURE CITED

Aghai, M.M.; Khan, Z.; Joseph, M.R. [and others]. 2019. The effect of microbial endophyte consortia on *Pseudotsuga menziesii* and *Thuja plicata* survival, growth, and physiology across edaphic gradients. Frontiers in Microbiology. 10: 1353. 16 p. https://doi.org/10.3389/fmicb.2019.01353.

Anderson, P.H.; Johnsen, K.H. 2009. Fertilizer responses of volunteer longleaf pine trees within a loblolly pine plantation: separating direct effects from competition effects. Res. Pap. SRS-48. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 6 p. https://doi.org/10.2737/SRS-RP-48.

Bates, D.; Mächler, M.; Bolker, B.M.; Walker, S.C. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software. 67(1): 1–48. https://doi.org/10.18637/jss.v067.i01.

Boyer, W.D. 1990. Growing season burns for control of hardwoods in longleaf pine stands. Res. Pap. SO-256. New Orleans: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 7 p. https://doi.org/10.2737/SO-RP-256.

Chanway, C.P.; Holl, F.B. 1991. Biomass increase and associative nitrogen fixation of mycorrhizal *Pinus contorta* seedlings inoculated with a plant growth promoting *Bacillus* strain. Canadian Journal of Botany. 69(3): 507–511. https://doi.org/10.1139/b91-069.

Derr, H.J. 1957. Effects of site treatment, fertilization, and brownspot control on planted longleaf pine. Journal of Forestry. 55(5): 364–367. https://doi.org/10.1093/jof/55.5.364.

Hangs, R.D.; Knight, J.D.; Van Rees, K.C.J. 2003. Nitrogen accumulation by conifer seedlings and competitor species from ^{15}nitrogen-labeled controlled-release fertilizer. Soil Science Society of America Journal. 67(1): 300–308. https://doi.org/10.2136/sssaj2003.3000.

Haywood, J.D. 2007. Influence of herbicides and felling, fertilization, and prescribed fire on longleaf pine establishment and growth through six growing seasons. New Forests. 33(3): 257–279. https://doi.org/10.1007/s11056-006-9026-3.

Haywood, J.D. 2000. Mulch and hexazinone herbicide shorten the time longleaf pine seedlings are in the grass stage and increase height growth. New Forests. 19(3): 279–290. https://doi.org/10.1023/A:1006673509218.

Jose, S.; Jokela, E.J.; Miller, D.L., eds. 2006. The longleaf pine ecosystem: Ecology, silviculture, and restoration. New York: Springer. 438 p. https://doi.org/10.1007/978-0-387-30687-2.

Khan, Z.; Kandel, S.L.; Ramos, D.N. [and others]. 2015. Increased biomass of nursery-grown Douglas-fir seedlings upon inoculation with diazotrophic endophytic consortia. Forests. 6(10): 3582–3593. https://doi.org/10.3390/f6103582.

Khan, Z.; Rho, H.; Firrincieli, A. [and others]. 2016. Growth enhancement and drought tolerance of hybrid poplar upon inoculation with endophyte consortia. Current Plant Biology. 6: 38–47. https://doi.org/10.1016/j.cpb.2016.08.001.

Padda, K.P.; Puri, A.; Chanway, C.P. 2021. Long-term effects of inoculating lodgepole pine seedlings with plant growth-promoting bacteria originating from a disturbed gravel mining ecosystem. Canadian Journal of Forest Research. 51(4): 533–545. https://doi.org/10.1139/cjfr-2020-0333.

Powers, R.F.; Ferrell, G.T. 1996. Moisture, nutrient, and insect constraints on plantation growth: the “Garden of Eden” study. New Zealand Journal of Forestry Science. 26(1/2): 126–144.

Ramsey, C.L.; Jose, S.; Brecke, B.J.; Merritt, S. 2003. Growth response of longleaf pine (*Pinus palustris* Mill.) seedlings to fertilization and herbaceous weed control in an old field in southern USA. Forest Ecology and Management. 172(2): 281–289. https://doi.org/10.1016/S0378-1127(01)00795-2.

R Core Team. 2023. R: a Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. https://www.R-project.org/. [Date accessed: October 30, 2023].

Renninger, H.J.; Stewart, L.F.; Rousseau, R.J. 2021. Water use, efficiency, and stomatal sensitivity in eastern cottonwood and hybrid poplar varietals on contrasting sites in the Southeastern United States. Frontiers in Forests and Global Change. 4: 704799. 16 p. https://www.frontiersin.org/articles/10.3389/ffgc.2021.704799.

Thiffault, N.; Titus, B.D.; Munson, A.D. 2005. Silvicultural options to promote seedling establishment on *Kalmia-Vaccinium*-dominated sites. Scandinavian Journal of Forest Research. 20(2): 110–121. https://doi.org/10.1080/02827580510008356.

SILVICULTURAL STRATEGIES FOR CONVERTING LONGLEAF PINE PLANTATIONS TO MULTIAGED LONGLEAF PINE STANDS

Joshua J. Puhlick, Adam D. Polinko, and Gabriel F. Nyen

ABSTRACT

To meet objectives such as maintaining continuous forest cover and promoting multiple age classes within stands, there is interest among some landowners to convert longleaf pine (*Pinus palustris*) plantations to multiaged stands. However, silvicultural strategies for converting longleaf pine plantations to multiaged stands need to be developed for local site conditions. This was the aim of our study located at The Jones Center at Ichauway in southwestern Georgia. The study site included eight longleaf pine plantations that were commercially thinned in 2014 and impacted by Hurricane Michael in 2018. In 2015, half of the plantations were seeded with native groundcover species. Our strategies will be informed by inventories of longleaf pine seedlings that established from natural regeneration after partial disturbances. We will also determine the relative success of groundcover restoration efforts, which will be based on biomass of native grasses and forbs 8 years after seeding.

INTRODUCTION

In the late 1990s and early 2000s, incentives for planting longleaf pine (*Pinus palustris*) led to thousands of hectares of mostly marginal agricultural lands being converted to longleaf pine plantations (Oswalt and others 2012) (fig. 1). Many of these plantations were established with Conservation Reserve Program funds to limit the overproduction of certain agricultural crops and to reduce soil erosion and improve water quality. As some landowners consider new objectives, such as improved wildlife habitat and carbon sequestration, they may gradually convert plantations to a more natural condition with new cohorts of longleaf pine developing from natural regeneration (Puhlick and others 2022, 2023). This strategy aligns with objectives of maintaining continuous forest cover and promoting multiple age classes within stands (fig. 2).

However, silvicultural strategies for converting longleaf pine plantations to multiaged stands need to be developed for local site conditions. One approach for converting longleaf pine plantations to a more natural condition is to develop treatments that mimic natural disturbances. In naturally regenerated second-growth stands, Palik and Pederson (1996) found that 5-year mortality rates for longleaf and slash pine (*P. elliottii*) >10 cm diameter at breast height (d.b.h.,

Figure 1—Longleaf pine plantation in 2022 on former agricultural land subjected to row cropping. Photo taken 8 years after the first commercial thinning in 2014, during which every third row of trees was removed along with some poor-quality trees in leave rows, and 4 years after hurricane disturbance in 2018. (Photo courtesy of Joshua Puhlick)

Joshua J. Puhlick, Assistant Scientist, The Jones Center at Ichauway, Newton, GA 39870
Adam D. Polinko, Assistant Professor, Mississippi State University, College of Forest Resources, Mississippi State, MS 39762
Gabriel F. Nyen, Graduate Student, Mississippi State University, College of Forest Resources, Mississippi State, MS 39762

Figure 2—Multiaged longleaf pine stand in 2022 that originated from seedlings planted on marginal agricultural land in 1986. Commercial thinning occurred in 2004 and 2014, and salvage logging was conducted in 2017 after a tornado. (Photo courtesy of Joshua Puhlick)

Figure 3—Recruitment of longleaf pine in a longleaf pine plantation in 2022 that was commercially thinned in 2014 and salvage logged after Hurricane Michael impacted Ichauway in October 2018. (Photo courtesy of Joshua Puhlick)

1.37 m) averaged 2.3 trees/ha or 1.9 percent of original density, with the primary disturbance agents they identified as lightning, competition, and wind. Additional research is needed to determine if the rates of mortality observed in natural stands are the same for plantations with trees that have reached merchantable size and that have been recently impacted by the hurricane.

Studies that quantify the spatial patterns of mortality and associated density reductions within stands would be useful for developing silvicultural treatments that mimic partial disturbance regimes. For instance, within the historical range of longleaf pine, hurricanes can cause large-scale tree mortality across regions but may only result in partial mortality within plantations and natural stands (Zampieri and others 2020). Partial disturbance could favor the recruitment of new cohorts of longleaf pine by increasing the amount of light that reaches the forest floor and stimulating cone production in residual overstory trees (fig. 3).

Restoring native grasses and forbs in plantations is another way to transition plantations to a more natural condition, especially when previous land use was row cropping. One restoration approach has been to seed native grasses and forbs after the first commercial thinning of longleaf pine plantations (Baniya and others 2022). Seeding could also be done during stand establishment, but a potential drawback to this approach is that the tree canopy quickly (within approximately 7 years) begins to close and shade out the understory (Kirkman and Giencke 2017). Studies on the relative success of groundcover restoration efforts are often based on cover estimates of the seeded species. However, biomass may be a better way to evaluate groundcover restoration efforts in longleaf pine ecosystems. This is because there is a high degree of horizontal overlap among plants that complicates estimating cover by species or species group within a given area. New approaches for evaluating the success of groundcover restoration efforts are needed to guide management recommendations for increasing plant diversity and wildlife habitat in longleaf pine plantations.

The overall goal of this project is to develop silvicultural strategies for converting longleaf pine plantations to multiaged longleaf pine stands. Our first objective is to determine the influence of disturbance (commercial thinning and hurricane) on recruitment of new cohorts of longleaf pine in longleaf pine plantations. Our second objective is to determine the success of groundcover restoration efforts that were initiated after the first commercial thinning of longleaf pine plantations. This is an ongoing project with field measurements and sampling being conducted during the summer of 2023. In this paper, we present descriptive statistics on overstory conditions in 2013 (before thinning in 2014) and in May 2023. At the stand scale, we also

present extractable phosphorus (P) concentration results from mineral soils. On completion of the project, our findings will be used to inform strategies for converting longleaf pine plantations to multiaged longleaf pine stands with enhanced groundcover diversity of native species.

MATERIALS AND METHODS

Study Area

The study was located at The Jones Center at Ichauway, which is in the Gulf Coastal Plain of southwestern Georgia. Soils are predominately well-drained loamy sands over sandy loams. The climate features long, hot summers and cool, short winters. Our study was conducted in eight longleaf pine plantations ranging in size from 2.7 to 8.0 ha. The plantations were established on marginal agricultural land from 1991 through 1996. Prior to stand establishment, peanut (*Arachis hypogaea*) was grown as the primary crop and fertilizers with P and potassium (K) were repeatedly applied to soils (Baniya and others 2022). The planting density of longleaf pine was 988 trees/ha at a spacing of 2.74 m by 3.66 m (Baniya and others 2022).

From March to May 2014, all eight plantations were commercially thinned for the first time. The method of thinning was a geometric thinning in which every third row of trees was removed, including some poor quality trees in leave rows. Though shade from the overstory canopy had reduced the presence of understory plants prior to thinning, all of the plantations were sprayed with 1,121 g ha^{-1} isopropylamine salt of imazapyr (Arsenal®, BASF, Florham Park, NJ) and 112 g ha^{-1} metsulfuron-methyl (MetcelTM, Agsurf Corporation, Wilmington, DE) because of the occurrence of pasture grasses such as Bahia grass (*Paspalum notatum* var. *notatum*) and Bermuda grass (*Cynodon dactylon*) (Baniya and others 2022). Based on photo time stamps, this spraying treatment occurred in September 2014 (fig. 4).

Figure 4—The first herbicide treatment that occurred after commercial thinning, 2014. (Photo courtesy of The Jones Center at Ichauway)

In 2015, logging slash was removed from all plantations using a grapple rake so that native groundcover could be sown and hunters could more easily navigate through stands while hunting northern bobwhite quail (*Colinus virginianus*) (fig. 5). During April and May 2015, half of the plantations were seeded with five warm-season C4 grasses and the legume goat's rue (*Tephrosia virginiana*) using a no-till seed drill (Truax Company, New Hope, MN) (fig. 6). Seed would have been sown earlier in the year, but for logistical reasons the seeding was delayed. The seed drills had coulters and openers to create furrows in the soil, into which seed was sown and covered with soil by press wheels. The seeded grasses were wiregrass (*Aristida stricta*), little bluestem (*Schizachyrium scoparium*), yellow Indian grass (*Sorghastrum nutans*), lopsided Indian grass (*Sorghastrum secundum*), and switchgrass (*Panicum virgatum*). For these grasses, the pure live seeding rates were 0.54, 1.93, 1.77, 0.50, and 0.03 kg ha^{-1}. For goat's rue, the seedling rate was 0.27 kg ha^{-1}. The seeded species were sown adjacent to rows of trees that were removed ("take-out rows") during thinning (fig. 6). A treatment of 2,242 g ha^{-1} glyphosate

Figure 5—Removal of logging slash from plantations using a grapple rack. Photo taken in May 2015 after commercial thinning. (Photo courtesy of The Jones Center at Ichauway)

Figure 6—Sowing of native groundcover species with a no-till seed drill while simultaneously applying herbicides to reduce competition of undesired species. Photo taken in May 2015 after commercial thinning and slash removal. (Photo courtesy of The Jones Center at Ichauway)

Figure 7—Frequent, low-intensity prescribed burning in plantations minimizes hardwood recruitment to midstory canopy positions. Grasses and forbs will dominate the understory of this plantation within weeks after dormant season burning. (Photo courtesy of Joshua Puhlick)

(Roundup®, Monsanto, St. Louis, MO), and 280 g ha^{-1} ammonium salt of imazapic (Panoramic, Alligare, Opelika, AL) was applied while simultaneously seeding native groundcover species (Jack 2015) (fig. 6).

For the plantations in this study, the warm season grasses that were sown were chosen because they were found in naturally regenerated, second-growth longleaf pine woodlands at Ichauway, they provided critical fine fuels for use of prescribed fire, and they offered cover for ground-nesting birds (Kirkman and Giencke 2017). Goat's rue is an important nitrogen fixer and a valuable resource for insect pollinators. The plantations were burned biennially between February and May since 1999 (Baniya and others 2022) (fig. 7). They were also impacted by Hurricane Michael in October 2018, and damaged and downed trees were removed from the plantations during salvage logging operations from January through February 2019.

Overstory Trees

In May 2023, we measured overstory tree attributes on a total of sixty-eight 0.04-ha plots (radius = 11.3 m), which had been established in 2013. In each plantation, there were 7–11 plots. For each plot, we recorded species and d.b.h. for trees at least 12.7 cm d.b.h. We also located previous records of trees on these same plots that were inventoried from May through September 2013 before commercial thinning. After this initial inventory, fixed-radius plot sampling was abandoned until our 2023 measurements.

Grasses and Forbs

In the summer of 2023, understory plants were clipped at ground level and collected from 1-m^2 quadrats. The five seeded grasses and goat's rue were bagged separately, oven dried to constant mass at 70 °C, and weighed. Quadrats were systematically located adjacent to take-out rows (i.e., offset 1.37 m from and perpendicular to stumps in the take-out rows) and between rows of overstory trees that were retained. In seeded stands, the location of clip plots adjacent to take-out rows ensured that plants were sampled from locations where seed was sown into the soil (fig. 6). We clipped plants from approximately four quadrats per hectare.

Tree Regeneration

In the summer of 2023, longleaf pine regeneration was measured within a 0.001-ha area at each clip plot location. For trees <1.37 m tall, height from the groundline to the base of current year's growth along the apical meristem was measured and recorded. Diameter at breast height was measured and recorded for trees ≥1.37 m tall and <12.7 cm d.b.h.

Data Analysis

Once the 2023 field measurements are complete and groundcover samples have been collected, we will compare the mean biomass of each seeded species between treatments (i.e., plantations with versus without seeding). Hemispherical photos will be taken at each clip plot location, so that metrics such as canopy openness can be used as explanatory variables in models of seeded species biomass. Plantation and row classification (take-out or leave row) will be included in models as random effects. Hurdle or two-stage models will be considered for modeling longleaf pine regeneration density due to the expected high frequency of zero counts. For this paper, descriptive statistics on overstory attributes in 2013 and 2023 are presented. We also present extractable P concentrations for the mineral soils of each plantation.

RESULTS AND DISCUSSION

For the eight plantations in this study, mean tree densities were 667 trees/ha before thinning and 191 trees/ha after thinning and salvage logging (table 1). During this same time period, quadratic mean diameter increased from 20.9 to 29.9 cm. Before thinning, mean stand density index (SDI) was 197; in 2023, mean SDI was 96. An SDI of 140 represents the lower limit of full site occupancy for longleaf pine, which is 35 percent of the maximum size density (Shaw and Long 2007). In 2023, there was a wide range of tree densities among and within plantations. We observed and are currently measuring longleaf pine seedlings in the eight plantations. These seedlings likely became established after commercial thinning in 2014 and salvage logging following Hurricane Michael. Many of the seedlings have persisted through multiple prescribed fires and will likely be recruited into midstory canopy positions. Hence, partial disturbances of commercial thinning followed by hurricane damage contributed to the development of a two-aged structure in most of the plantations.

Table 1–Grand mean and range of plantation attributes by time period

	2013		2023	
Plantation attribute	Mean (SD)	Range	Mean (SD)	Range
Basal area (m^2/ha)	22.3 (4.0)	13.2–26.2	12.6 (2.3)	8.6–15.0
Density (trees/ha)	667 (153)	341–828	191 (69)	111–331
QMD (cm)	20.9 (1.2)	18.9–22.7	29.9 (2.6)	24.3–32.0
SDI	197 (27)	139–224	96 (18)	70–122

The 2013 values include trees at least 2.5 cm diameter at breast height (d.b.h.) and the 2023 values include trees at least 12.7 cm d.b.h.

Mean values for the eight plantations of this study were used to compute these statistics.

2013 = before the first commercial thinning which occurred in 2014; 2023 = approximately 5 years after Hurricane Michael which impacted Ichauway in October 2018; SD = standard deviation; QMD = quadratic mean diameter; SDI = stand density index.

In 2020, extractable P concentration was determined for the mineral soils of each plantation (Baniya and others 2022). Those authors suggested that elevated stocks of available soil P had allowed old field plants to persist and perhaps inhibit establishment of the seeded groundcover species. We revisited the 2020 extractable P concentration dataset and determined that one plantation (stand 3) had a greater mean P concentration than that of other plantations (fig. 8). For plantations other than stand 3, soil available P may not have been the dominant factor influencing the relative success of groundcover restoration efforts. However, determining the impact of extractable P stocks on aboveground biomass of the seeded species would involve having estimates of soil bulk density or soil mass by depth increment, neither of which were reported by Baniya and others (2022). We propose that location within plantations, specifically in take-out rows versus leave rows, and differences in the light environment within and across plantations may be a better explanation for variation in presence of the seeded species.

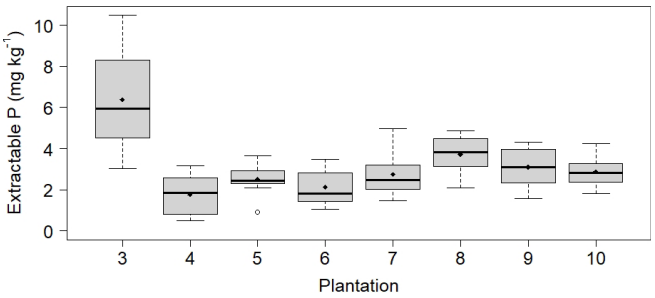

Figure 8—Extractable phosphorus concentration for the 0- to 10-cm depth increment of the mineral soil in 2020 by plantation, which were assigned numbers 3–10. The boxes define the hinge (25–75 percent quartile, and the line is 1.5 times the hinge) and the horizontal line in each box is the median. Solid black dots are means.

CONCLUSIONS

Quantifying stand attributes, including longleaf pine regeneration densities and spatial patterns, after commercial thinning and salvage logging in longleaf pine plantations can inform silvicultural prescriptions for promoting multiaged stand conditions. For the mid-rotation longleaf pine plantations of this study, we will examine how silvicultural treatments and natural disturbance influenced regeneration outcomes. Restoring longleaf pine stands to a more natural condition can also include native groundcover restoration, especially when previous land use was row cropping. In this study, seeding of native groundcover species was done after commercial thinning, which may be a better alternative than seeding grasses and forbs at the time of tree planting when financial resources are limited. The ability of seeded grasses and forbs to spread throughout stands could influence decisions about skid trail layout during future timber harvesting. For groundcover species with limited dispersal, some portions of stands where they occur may require reduced trafficking of machinery. Determining seeded groundcover species and longleaf pine regeneration presence and biomass in take-out and leave rows is crucial for making such decisions.

REFERENCES

Baniya, B.; Bigelow, S.W.; Sharma, A. [and others] 2022. Re-assembly of the longleaf pine ecosystem: effects of groundcover seeding on understory community, fire behavior and soil properties. Forests. 13: 519. 17 p. https://doi.org/10.3390/f13040519.

Jack, S. 2015. Personal communication. Executive Director, Boggy Slough Conservation Area, 204 Champions Drive, Lufkin, TX 75901.

Kirkman, L.K.; Giencke, L.M. 2017. Restoring and managing a diverse ground cover. In: Kirkman, L.K.; Jack, S.B., eds. Ecological restoration and management of longleaf pine forests. Boca Raton, FL: CRC Press: 207–232.

Oswalt, C.M.; Cooper, J.A.; Brockway, D.G [and others]. 2012. History and current condition of longleaf pine in the Southern United States. Gen. Tech. Rep. SRS-166. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 51 p. https://doi.org/10.2737/SRS-GTR-166.

Palik, B.J.; Pederson, N. 1996. Overstory mortality and canopy disturbances in longleaf pine ecosystems. Canadian Journal of Forest Research. 26(11): 2035–2047. https://doi.org/10.1139/x26-229.

Puhlick, J.J.; Brantley, S.T.; O'Halloran, T.L. [and others]. 2022. Perspectives: Carbon markets might incentivize poorer ecological outcomes in longleaf pine ecosystems. Forest Ecology and Management. 520: 120421. 5 p. https://doi.org/10.1016/j.foreco.2022.120421.

Puhlick, J.J.; O'Halloran, T.L.; Starr, G. [and others]. 2023. Opportunities for research on carbon management in longleaf pine ecosystems. Forests. 14: 874. 13 p. https://doi.org/10.3390/f14050874.

Shaw, J.D.; Long, J.N. 2007. A density management diagram for longleaf pine stands with application to red-cockaded woodpecker habitat. Southern Journal of Applied Forestry 31(1): 28–38. https://doi.org/10.1093/sjaf/31.1.28.

Zampieri, N.E.; Pau, S.; Okamoto, D.K. 2020. The impact of Hurricane Michael on longleaf pine habitats in Florida. Scientific Reports. 10: 8483. 11 p. https://doi.org/10.1038/s41598-020-65436-9.

HURRICANES STIMULATE CONE PRODUCTION IN LONGLEAF PINE WOODLANDS

Jeffery B. Cannon, Brandon T. Rutledge, Joshua J. Puhlick, John L. Willis, and Dale G. Brockway

EXTENDED ABSTRACT

Many trees exhibit a reproductive strategy known as masting where seed production is highly variable in time (Janzen 1971, Koenig 2021, Koenig and Knops 2005). Variability in weather patterns can trigger seed production leading to synchrony over large geographic areas (Pearse and others 2014). Disturbances such as drought and fire are known to stimulate masting, but little is known about whether hurricanes also stimulate seed production (Vacchiano and others 2021). Hurricanes may trigger cone production through multiple environmental stimuli associated with windstorms such as additional precipitation, neighbor removal, and sublethal mechanical damage (fig. 1).

Figure 1—Hurricane damage to a longleaf pine forest in Newton, GA, following 2018 Hurricane Michael. (Photo courtesy of The Jones Center at Ichauway)

Jeffery B. Cannon, Landscape Ecologist, The Jones Center at Ichauway, Newton, GA 39870

Brandon T. Rutledge, Conservation Coordinator, The Jones Center at Ichauway, Newton, GA 39870

Joshua J. Puhlick, Ecological Silviculturist, The Jones Center at Ichauway, Newton, GA 39870

John L. Willis, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Auburn, AL 36849

Dale G. Brockway, Research Ecologist, U.S. Department of Agriculture, Forest Service, Southern Research Station, Auburn, AL 36849

Longleaf pine (*Pinus palustris*) is considered a weakly masting species with sporadic seed production and is concentrated in the hurricane-prone Southeastern United States (Chen and Willis 2023, Guo and others 2016), making it suitable for testing whether intense winds stimulate seed production. In addition, longleaf pine is a primary focus for restoration efforts, as the species is expected to be resilient to future climate changes such as increased drought and hurricane intensity (Bigelow and others 2021, McLaughlin and others 2013, Noss and others 1995, Qi and others 2022, Rutledge and others 2021).

Due to widespread interest in restoration and management of longleaf pine, long-term records of cone production are available that can provide valuable insights into the dynamics of cone production (Willis and Brockway 2022). We combined these cone records with hurricane records to test the hypothesis that hurricanes stimulate cone production in longleaf pine. A record of cone production is available for 12 mature longleaf pine sites throughout the Southeastern United States with record lengths ranging from 8 to 67 years (1958 to 2022). At each of these sites, scientists with the U.S. Department of Agriculture, Forest Service conducted annual cone counts using 8- to 10-power binoculars between April and May each year (Willis and Brockway 2022). Annual information on cone production is available for each site as the mean number of cones per tree based on the averaged counts from a set of 10 to 17 trees per site for most sites, and 50 trees at one site. Initially, trees were selected as dominant healthy trees after a shelterwood cut, and consistently sampled each year. The death of a sampled tree was replaced with a nearby healthy tree in a dominant canopy position.

Using the hurricane tracks available from the National Oceanic and Atmospheric Administration [Best Track Data (HURDAT2)] (Landsea and Franklin 2013, NOAA 2022), we identified 96 instances where hurricane winds approached monitored sites, and 34 cases where cone count data was available 1 year before and each of 3 years after a hurricane. To test the hypothesis that hurricanes influence cone production, we assessed changes in cone production based on lag time since a recent hurricane ranging from 1 year before to 3 years after a hurricane. This sampling provided a means to compare the years after a hurricane to the baseline 1 year prior. Over the 4 years, repeated measures analysis of variance was used to test whether time since hurricane had a significant impact on cone production.

Preliminary results reveal that cone production increased by 31 percent in the first year after a hurricane, and 71 percent higher in the second year, before returning to prestorm levels in year three ($P = 0.028$). This suggests that hurricanes may stimulate reproduction in longleaf pine, and more generally provides evidence that intense winds may trigger masting in some tree species. Longleaf pine was once a dominant tree in forests of the hurricane-prone Gulf Coastal Plain (Cannon and others 2023; Dahal and others 2014; Keim and others 2007). Thus, it should not be surprising that the species exhibits a reproductive response to hurricane winds. The posthurricane environment provides opportunities for successful longleaf pine germination, such as bare mineral soil within tip-up mounds

(Beatty1984, Sobhani and others 2014, Ulanova 2000). Large downed logs from wind storms may also temporarily reduce fire continuity (Cannon and others 2017) and provide a fire-free period needed for successful longleaf pine establishment (Robertson and others 2019).

Our results demonstrate that hurricanes can increase reproduction, but the mechanism of this effect remains unknown. Hurricanes may increase cone production by increasing precipitation, freeing up resources from competing vegetation, or by stimulating reproduction via stress mechanisms. For example, anomalous precipitation typically accompanies hurricanes, and can be significant especially in the late summer and early fall (August through September) when precipitation is low in the region (Schwarz 1970). Severe winds can topple trees, thinning forests and providing greater access to resources for surviving mature trees (Brockway and others 2006, Croker and Boyer 1975). Intense winds cause sublethal mechanical damage to trees such as leaf stripping, broken branches, or stem torsion (Doyle and others 1995, Rutledge and others 2021). These disturbances may elicit a physiological response that increases reproduction—similar to what is sometimes observed after other disturbances such as wildfire and drought (Lauder and others 2019). Further research into how cone production may vary with each of these factors is underway and may shed light on the specific mechanism responsible for hurricane-triggered masting events.

There is widespread interest in promoting conservation, management, and restoration of longleaf pine forests and woodlands (McIntyre and others 2018). Highly variable cone and seed production make it challenging for managers of longleaf pine to regenerate the species naturally. Often, prescribed burns must be timed with masting events for successful seedling establishment. Increasing the understanding of the factors driving seed production at stand and regional scales can promote natural regeneration of the species. Further, understanding population dynamics in the face of threats from increasing climate-related disturbances will be critical for formulating silvicultural prescriptions for adapting to anticipated changes.

REFERENCES

Beatty, S.W. 1984. Influence of microtopography and canopy species on spatial patterns of forest understory plants. Ecology. 65: 1406–1419. https://doi.org/10.2307/1939121.

Bigelow, S.W.; Looney, C.E.; Cannon, J.B. 2021. Hurricane effects on climate-adaptive silviculture treatments to longleaf pine woodland in southwestern Georgia, USA. Forestry. 94: 395–406. https://doi.org/10.1093/forestry/cpaa042.

Brockway, D.G.; Outcalt, K.W.; Boyer, W.D. 2006. Longleaf pine regeneration ecology and methods. In: Jose, S.; Jokela, E.J.; Miller, D.L., eds. The longleaf pine ecosystem: ecology, silviculture, and restoration. New York, NY: Springer: 95–133.

Cannon, J.B.; Peterson, C.J.; Godfrey, C.M.; Whelan, A.W. 2023. Hurricane wind regimes for forests of North America. Proceedings of the National Academy of Sciences. 120(42): e2309076120. https://doi.org/10.1073/pnas.2309076120.

Cannon, J.B.; Peterson, C.J.; O'Brien, J.J.; Brewer, J.S. 2017. A review and classification of interactions between forest disturbance from wind and fire. Forest Ecology Management. 406: 381–390. https://doi.org/10.1016/j.foreco.2017.07.035.

Chen, X.; Willis, J.L. 2023. Individuals' behaviors of cone production in longleaf pine trees. Forests. 14: 494. 10 p. https://doi.org/10.3390/f14030494.

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. 21 p.

Dahal, D.; Liu, S.; Oeding, J. 2014. The carbon cycle and hurricanes in the United States between 1900 and 2011. Scientific Reports. 4: 5197. 10 p. https://doi.org/10.1038/srep05197.

Doyle, T.W.; Smith, T.J. III; Robblee, M.B. 1995. Wind damage effects of Hurricane Andrew on mangrove communities along the southwest coast of Florida, USA. Journal of Coastal Research. 21: 159–168.

Guo, Q.; Zarnoch, S.J.; Chen, X.; Brockway, D.G. 2016. Life cycle and masting of a recovering keystone indicator species under climate fluctuation. Ecosystem Health and Sustainability. 2(6): e01226. 11 p. https://doi.org/10.1002/ehs2.1226.

Janzen, D.H. 1971. Seed predation by animals. Annual Review of Ecology and Systematics. 2: 465–492. https://doi.org/10.1146/annurev.es.02.110171.002341.

Keim, B.D.; Muller, R.A.; Stone, G.W. 2007. Spatiotemporal patterns and return periods of tropical storm and hurricane strikes from Texas to Maine. Journal of Climate. 20(14): 3498–3509. https://doi.org/10.1175/JCLI4187.1.

Koenig, W.D. 2021. A brief history of masting research. Philosophical Transactions of the Royal Society B. 376: 20200423. 7 p. https://doi.org/10.1098/rstb.2020.0423.

Koenig, W.D.; Knops, J.M.H. 2005. The mystery of masting in trees. American Scientist. 93: 340–347.

Landsea, C.W.; Franklin, J.L. 2013. Atlantic hurricane database uncertainty and presentation of a new database format. Monthly Weather Review. 141(10): 3576–3592. https://doi.org/10.1175/MWR-D-12-00254.1.

Lauder, J.D.; Moran, E.V.; Hart, S.C. 2019. Fight or flight? Potential tradeoffs between drought defense and reproduction in conifers. Tree Physiology. 39: 1071–1085. https://doi.org/10.1093/treephys/tpz031.

McIntyre, R.K.; Guldin, J.M.; Ettel, T. [and others]. 2018. Restoration of longleaf pine in the southern United States: a status report. In: Kirschman, J.E., comp. Proceedings of the 19th biennial southern silvicultural research conference. e-Gen. Tech. Rep. SRS-234. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 297–302.

McLaughlin, D.L.; Kaplan, D.A.; Cohen, M.J. 2013. Managing forests for increased regional water yield in the southeastern U.S. coastal plain. Journal of the American Water Resources Association. 49: 953–965. https://doi.org/10.1111/jawr.12073.

National Oceanic and Atmospheric Administration [NOAA]. 2022. NHC data archive. National Hurricane Center. https://www.nhc.noaa.gov/data/#hurdat. [Date accessed: August 20, 2022].

Noss, R.F.; LaRoe, E.T. III; Scott, J.M. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Biological Report 28. Washington, DC: U.S. Department of the Interior National Biological Service. 58 p.

Pearse, I.S.; Koenig, W.D.; Knops, J.M.H. 2014. Cues versus proximate drivers: testing the mechanism behind masting behavior. Oikos. 123: 179–184. https://doi.org/10.1111/j.1600-0706.2013.00608.x

Qi, J.; Brantley, S.T.; Golladay, S.W. 2022. Simulated longleaf pine (*Pinus palustris* Mill.) restoration increased streamflow—a case study in the Lower Flint River Basin. Ecohydrology. 15: e2365. 15 p. https://doi.org/10.1002/eco.2365.

Robertson, K.M.; Platt, W.J.; Faires, C.E. 2019. Patchy fires promote regeneration of longleaf pine (*Pinus palustris* Mill.) in pine savannas. Forests. 10: 367. 16 p. https://doi.org/10.3390/f10050367.

Rutledge, B.T.; Cannon, J.B.; McIntyre, R.K. [and others]. 2021. Tree, stand, and landscape factors contributing to hurricane damage in a coastal plain forest: post-hurricane assessment in a longleaf pine landscape. Forest Ecology Management. 481: 118724. 13 p. https://doi.org/10.1016/j.foreco.2020.118724.

Schwarz, F.K. 1970. The unprecedented rains in Virginia associated with the remnants of Hurricane Camille. Monthly Weather Review. 98(11): 851–859. https://doi.org/10.1175/1520-0493(1970)098<0851:TURIVA>2.3.CO;2.

Sobhani, V.M.; Barrett, M.; Peterson, C.J. 2014. Robust prediction of treefall pit and mound sizes from tree size across 10 forest blowdowns in eastern North America. Ecosystems. 17: 837–850. https://doi.org/10.1007/s10021-014-9762-8.

Ulanova, N.G. 2000. The effects of windthrow on forests at different spatial scales: a review. Forest Ecology and Management. 135: 155–167. https://doi.org/10.1016/S0378-1127(00)00307-8.

Vacchiano, G.; Pesendorfer, M.B.; Conedera, M. [and others]. 2021. Natural disturbances and masting: from mechanisms to fitness consequences. Philosophical Transactions of the Royal Society B. 376: 20200384. 13 p. https://doi.org/10.1098/rstb.2020.0384.

Willis, J.L.; Brockway, D.G. 2022. Longleaf pine cone prospects. 22 p. Unpublished report. On file with: U.S. Department of Agriculture Forest Service, Southern Research Station, 521 Devall Drive, Auburn, AL 36849.

SILVICULTURAL STRATEGIES FOR PROMOTING LONGLEAF PINE RECRUITMENT IN MULTIAGED STANDS

Joshua J. Puhlick, Benjamin O. Knapp, Ian Goldberg, Brandon T. Rutledge, R. Scott Taylor, and John L. Willis

ABSTRACT

In longleaf pine (*Pinus palustris*) ecosystems, an objective of some landowners is to develop multiaged stands with a component of large longleaf pine trees. A potential strategy to accelerate recruitment of trees to larger size classes is to purposefully release saplings within dense patches of longleaf pine. At The Jones Center at Ichauway, we thinned patches of sapling-sized trees within mature stands. We thinned from below with the modification that some dominant and codominant trees were also cut. Before thinning, average densities within patches of two harvest areas were 712 and 1,145 trees per acre. Many patches had stand density index values close to those associated with the onset of competition-induced mortality. Thinning was done with a feller-buncher and there was minimal damage to residual saplings. Precommercial thinning in multiaged stands of open pine systems is an investment but may accomplish objectives such as enhanced carbon accumulation and groundcover productivity.

INTRODUCTION

Reducing tree densities within sapling thickets that occur as a component of multiaged stands is often necessary to increase tree vigor (fig. 1). In open pine (*Pinus* spp.) systems with frequent, low-intensity fires, burning can sometimes reduce tree densities within dense patches of pine saplings (Cooper 1960, Weaver 1943). Similar to heavy snow loads causing tree mortality within sapling thickets of ponderosa pine (*P. ponderosa*) (Pearson 1950), we have observed a thinning effect in dense patches of longleaf pine (*P. palustris*) due to wind disturbance (fig. 2).

Emulating these natural processes in open pine systems using precommercial thinning has been underutilized in multiaged stands because of the cost and a lack of knowledge of the potential benefits. For example, thinning dense patches of pine saplings could potentially enhance carbon accumulation, increase native groundcover, and benefit wildlife species associated with open pine systems. Increased tree vigor through reduced competition for resources could also decrease

Figure 1— A thicket of longleaf pine saplings that developed in a canopy gap of a multiaged stand at The Jones Center at Ichauway. (Photo by Joshua Puhlick)

Joshua J. Puhlick, Assistant Scientist, The Jones Center at Ichauway, Newton, GA 39870
Benjamin O. Knapp, Associate Professor, University of Missouri, School of Natural Resources, Columbia, MO 65211
Ian Goldberg, Graduate Student, University of Missouri, School of Natural Resources, Columbia, MO 65211
Brandon T. Rutledge, Conservation Coordinator, The Jones Center at Ichauway, Newton, GA 39870
R. Scott Taylor, Research Associate, The Jones Center at Ichauway, Newton, GA 39870
John L. Willis, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Auburn, AL 36849

Figure 2—Excessive bending and snapping of stems within a dense patch of longleaf pine due to Hurricane Michael, which occurred in October 2018. This photo was taken in September 2021 at The Jones Center at Ichauway. (Photo by Joshua Puhlick)

the vulnerability of residual saplings to drought (D'Amato and others 2013). On poor sites and for tree species with poor height differentiation, saplings in dense thickets may reach the point of stagnation (Smith and others 1997). Releasing some individuals within patches, especially before stagnation occurs, could enhance carbon accumulation in these stands. Recruiting trees with large diameters and crowns faster than without thinning could also be desirable for wildlife species such as the red-cockaded woodpecker (*Leuconotopicus borealis*) (Shaw and Long 2007).

In even-aged stands, density management diagrams (DMDs) are often used to inform thinning schedules and estimate stand trajectories. However, DMDs can also be used for one-age cohorts within multiaged stands (Shaw and Long 2007). First, the stand density index (SDI) is calculated for a group (or patch) of similar-sized trees within a multiaged stand. This is done by using the density and mean tree size (or volume) based on trees within the patch and the patch size. Briefly, SDI is an index of relative density that is independent of stand age and site quality and is useful for evaluating the degree of competitive interaction among trees. The patch's SDI is then compared to various relative density thresholds such as values that correspond to the onset of competition among trees and the onset of competition-induced mortality.

The overall goal of this study was to evaluate the use of precommercial thinning to reduce tree densities within dense sapling patches that are a component of multiaged longleaf pine stands. Specific objectives were to 1) quantify attributes of sapling patches (e.g., tree density and mean tree size) before and after precommercial thinning, and 2) evaluate patch trajectories using a DMD developed for longleaf pine by Shaw and Long (2007).

MATERIALS AND METHODS

Study Area

The study was located at The Jones Center at Ichauway, in the Gulf Coastal Plain of southwestern Georgia. Soils are predominately well-drained loamy sands over sandy loams. The climate features long, hot summers and cool, short winters. The study was conducted in two harvest areas of approximately 300 acres each. The harvest areas were named Whitehead and Malone-Bud Walker and were classified as upland longleaf pine-hardwood woodlands. Groundcover was diverse but dominated by wiregrass (*Aristida beyrichiana*). Aerial photography from 1938 shows that these locations contained sparse tree cover and agricultural fields. No timber harvesting has occurred in these areas since 1989, and no records of previous harvesting within these locations exist. Starting around 1991, the frequency of prescribed burning was changed from annual to biennial to establish new cohorts of longleaf pine. The Whitehead harvest area was cut in April 2022, and the Malone-Bud Walker harvest area was cut in February and March 2023. Both locations contained multiaged stands, and the prescription for both was single-tree selection cutting.

Plot Selection and Tree Measurements

In both harvest areas, we recorded the locations of all sapling patches before timber harvesting. Circular, 0.5-acre plots for measuring trees were installed so that they completely contained one or more patches. This plot size was chosen to evaluate the influence of competition on

sapling-sized trees. Trees were measured on two plots within the Whitehead harvest area to determine if the precommercial thinning of saplings within patches could be reasonably accomplished using a feller-buncher. We also identified 12 potential plot locations within the Malone-Bud Walker harvest area. These locations were classified as having patches with small-, mid-, or large-sized saplings. We randomly chose two plots from each size category for a total of six plots in the Malone-Bud Walker harvest area (table 1). Then, we randomly selected one plot from each size category to be treated (i.e., saplings within patches were to be thinned) and the other plot was not to be treated (i.e., saplings within patches were not to be thinned).

We used data from eight plots across the two harvest areas to quantify the attributes of patches before and after timber harvest and precommercial thinning of trees within patches. Single-tree selection cutting and precommercial thinning of trees within patches was conducted while logging equipment was on site in the winter of 2022 and again in 2023. In this study, postharvest results were based on saplings that were marked to be retained. For control plots in the Malone-Bud Walker harvest area, saplings were identified within patches that would be retained if thinning were to occur. For future studies, the growth and mortality of these individuals will be compared to that of individuals in treated patches.

Table 1—Grand mean height of residual longleaf pine saplings within dense sapling patches of the Malone-Bud Walker harvest area by research plot

Plot	Total height (feet)	Number of patches	Patch size classification
8	30	2	Small
12	35	1	Small
1	49	2	Mid
18	50	4	Mid
14	52	2	Large
16	53	3	Large

Patch size classification was based on sapling height.

For each plot, species, diameter at breast height (d.b.h.), and azimuth and distance from plot center were recorded for trees at least 1 inch d.b.h. This size threshold was used by Shaw and Long (2007) to construct their DMD. For each tree, we also noted if it was to be retained or cut based on tree marking by the silviculturist (see Prescription section). To spatially display trees in ArcMap (ESRI, Redlands, CA), the x- and y-coordinates of each tree were derived using azimuth and distance from plot center, and the location of plot center. The boundaries of sapling patches within plots were hand sketched on field sheets to aid in delineating polygons around patches in ArcMap (fig. 3). Information for trees within these polygons was exported to Excel (Microsoft, Redmond, WA) so that descriptive statistics could be calculated for each patch. These statistics were calculated using only longleaf pines. Larger hardwoods generally occurred outside of patches and across all plots there were only 15 hardwoods ≥5.0 inches d.b.h. and 140 hardwoods <5.0 inches d.b.h.

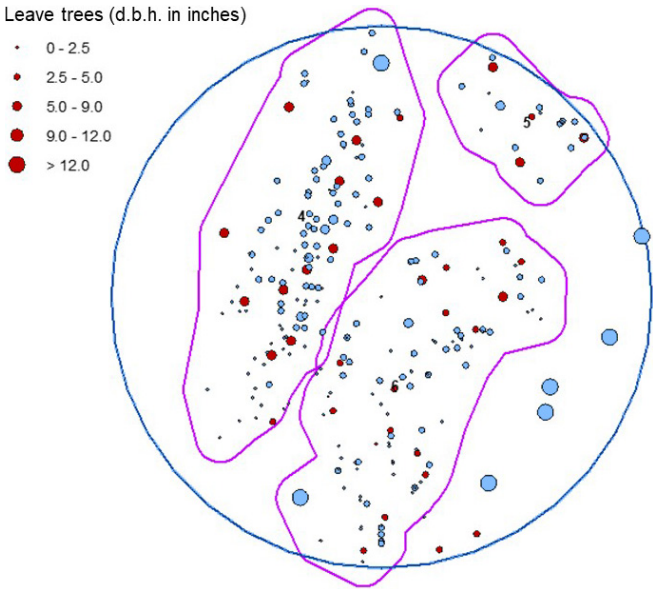

Figure 3—Three distinct patches of trees delineated within a 0.5-acre research plot in the Whitehead harvest area. Blue circles indicate trees to be cut. The size of the blue (light) circles represents the same diameter-at-breast-height range as for the red (dark) circles (trees to be retained).

Prescription

The treatment was a thinning from below with the modification that some dominant and codominant trees were also cut (fig. 4). We retained dominant and codominant trees of good form and with robust crowns. During marking, we also considered the spacing of residual trees based on the average height of dominant trees within patches. We followed recommendations by Wilson (1946), who suggested a spacing of one-fourth to one-fifth of the height for shade-intolerant species. For precommercial thinning, Wilson (1946) indicated that trees could be spaced according to the height they would obtain at the time of the first commercial thinning. Alternatively, he noted that spacing according to heights at the time of precommercial thinning is appropriate when the silviculturist is concerned about a thinning being too severe (e.g., concerns about windthrow). We used the conservative approach to spacing which was based on the average height of dominant trees within patches at the time of precommercial thinning. This decision was also based on the current belief that longleaf pine is more tolerant of shade than once thought (Samuelson and Stokes 2012). This suggests that a narrower spacing between trees may be appropriate. Most hardwood trees were cut so that they would not influence growth of residual pines.

RESULTS

For the two plots in the Whitehead harvest area, there were a total of six sapling patches. For these patches, mean tree densities were 712 trees per acre before thinning and 82 trees per acre after thinning (table 2). Thinning shifted the quadratic mean diameter (QMD) from 3.6 to 5.7 inches. For the six plots in the Malone-Bud Walker harvest area, there were a total of 14 sapling patches. For these patches, mean tree densities were 1,145 trees per acre before thinning and 112 trees per acre after thinning. Thinning shifted the QMD from 2.8 to 4.5 inches. For patches in the Malone-Bud Walker harvest area, patch size was 0.12 ± 0.05 acre (mean ± standard deviation).

Figure 4—Repeat photography of stand conditions (left) before and (right) after thinning a dense patch of longleaf pine in the Malone-Bud Walker harvest area. (Photos by Joshua Puhlick)

Table 2—Mean (standard deviation) and range of patch attributes by harvest area and time period for longleaf pine trees with a diameter at breast height ≥1.0 and <8.0 inches

	Whitehead		Malone-Bud Walker	
Patch attribute	Prethinning	Postthinning	Prethinning	Postthinning
Basal area	48 (16)	14 (5)	48 (19)	12 (5)
	26–69	4–19	23–77	5–19
Density	712 (236)	82 (37)	1,145 (322)	112 (40)
	331–935	23–126	693–1587	43–191
QMD	3.6 (0.4)	5.7 (0.7)	2.8 (0.6)	4.5 (1.1)
	2.8–4.0	4.3–6.3	2.0–3.8	2.7–6.4
SDI	133 (44)	31 (12)	145 (50)	30 (10)
	70–189	10–42	73–220	13–45
SDI_{sum}	125 (40)	31 (12)	137 (47)	29 (10)
	66–175	10–42	69–206	12–44
SDI_{sum}:SDI	0.94 (0.01)	0.99 (0.01)	0.94 (0.01)	0.98 (0.01)
	0.93–0.95	0.98–1.00	0.91–0.97	0.95–1.00

Basal area is in square feet per acre, density is trees per acre.
QMD = quadratic mean diameter (inches); SDI = stand density index; SDI_{sum} = sum of contributions of individual trees to overall stand density (equation 3 in Shaw and Long 2007).

Before thinning patches, 16 of 20 patches had SDI values >100, but none had SDI values >250. An SDI of 100 generally represents a transition from open-grown to competing tree populations, and we expected to see competition-induced mortality at values >250 to a maximum SDI of 400 (Shaw and Long 2007). Several patches in the Malone-Bud Walker harvest area had SDIs approaching the self-thinning line (fig. 5). For both harvest areas, mean SDI_{sum}:SDI was 0.94 before thinning (table 2), which indicates that the size distribution of trees within patches was similar to that of even-aged stands (Ducey 2009, Ducey and Larson 2003). After thinning, SDI_{sum}:SDI ranged from 0.95 to 1.00.

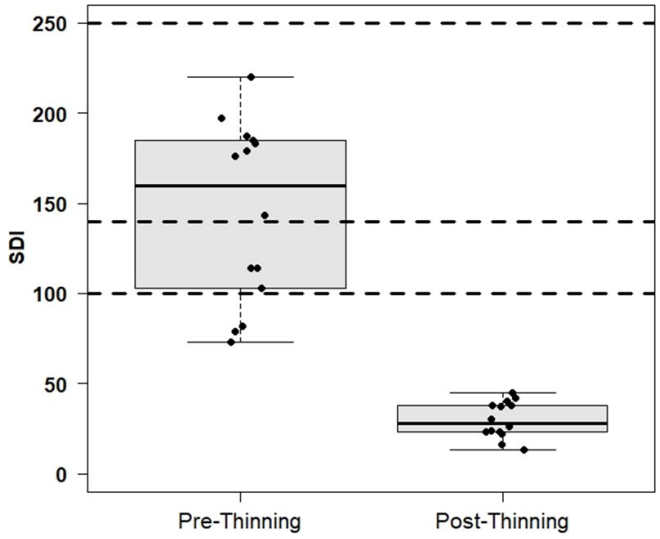

Figure 5—Stand density indices (SDIs) before and after thinning in the Malone-Bud Walker harvest area. The boxes define the interquartile range (IQR, 25–75 percent quartile). The vertical line below the boxes extends to a horizontal line that is the 25th percentile minus 1.5 times the IQR, and the vertical line above the boxes extends to a horizontal line that is the 75th percentile minus 1.5 times the IQR. The horizontal line in each box is the median. Dots represent individual patches. SDIs of 250, 140, and 100 represent the threshold for the self-thinning zone, the lower limit of full site occupancy for longleaf pine, and a transition from open-grown to competing tree populations, respectively (Shaw and Long 2007).

DISCUSSION

Thinning dense patches of longleaf pine saplings that are a component of multiaged stands may be an appropriate treatment for accelerating recruitment of trees to larger size classes and reducing competition-induced mortality. However, the influence of thinning on residual tree growth and mortality needs to be monitored and compared among trees within treated and nontreated patches. Pretreatment SDI_{sum}:SDI values suggested that using the DMD to make decisions about thinning was appropriate. For other multiaged stands with sapling patches, it is important to remember that the distribution

of tree diameters within patches could influence values of SDI_{sum}:SDI and proper use of the DMD. Although pretreatment SDI values were below the self-thinning line, we observed tree mortality in some patches. If the primary cause of death was competition-induced (we cannot rule out that some trees were killed in prescribed fires), then earlier intervention may have been appropriate.

Residual saplings had desired characteristics for wood quality, and we observed no mortality from windthrow or other disturbance agents since thinning. Most residuals were of good form and had branch-free boles for at least the first 16 feet starting from the base of the tree. Residuals also had low height-to-diameter ratios, which are important to maintain wind firmness. Though thinning may accelerate recruitment of saplings to larger diameter classes relative to natural gap dynamics, saplings within patches tended to display good height differentiation. For this reason, choosing not to thin some patches within a stand may be appropriate. Additionally, more research is needed to evaluate growth and mortality of residual trees in thinned patches.

Managers at Ichauway feel that the immediate and future benefits of precommercial thinning justify the costs. The cost of thinning patches was slightly over $100 per acre. In the Malone-Bud Walker harvest area, there were 12 total 0.5-acre plots with sapling patches available for research purposes. Assuming half of those plots were thinned, it would amount to a total cost of approximately $300. Such a cost may be justified in woodlands managed for multiple values such as aesthetics, recreation, or to improve the ability to see through the stand. Another long-term benefit of recruiting trees to larger size classes more quickly at Ichauway was improved wildlife habitat. Overall, our findings indicate this may be a practical treatment for landowners with multiaged longleaf pine stands.

LITERATURE CITED

Cooper, C.F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs. 30(2): 129–164. https://doi.org/10.2307/1948549.

D'Amato, A.W.; Bradford, J.B.; Fraver, S.; Palik, B.J. 2013. Effects of thinning on drought vulnerability and climate response in north temperate forest ecosystems. Ecological Applications. 23(8): 1735–1742. https://doi.org/10.1890/13-0677.1.

Ducey, M.J. 2009. The ratio of additive and traditional stand density indices. Western Journal of Applied Forestry. 24(1): 5–10. https://doi.org/10.1093/wjaf/24.1.5.

Ducey, M.J.; Larson, B.C. 2003. Is there a correct stand density index? An alternate interpretation. Western Journal of Applied Forestry. 18(3): 179–184. https://doi.org/10.1093/wjaf/18.3.179.

Pearson, G.A. 1950. Management of ponderosa pine in the Southwest as developed by research and experimental practice. Agric. Monogr. 6. Washington, DC: U.S. Department of Agriculture Forest Service. 218 p.

Samuelson, L.J.; Stokes, T.A. 2012. Leaf physiological and morphological responses to shade in grass-stage seedlings and young trees of longleaf pine. Forests. 3(3): 684–699. https://doi.org/10.3390/f3030684.

Shaw, J.D.; Long, J.N. 2007. A density management diagram for longleaf pine stands with application to red-cockaded woodpecker habitat. Southern Journal of Applied Forestry. 31(1): 28–38. https://doi.org/10.1093/sjaf/31.1.28.

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

Weaver, H. 1943. Fire as an ecological and silvicultural factor in the ponderosa-pine region of the Pacific Slope. Journal of Forestry. 41(1): 7–15. https://doi.org/10.1093/jof/41.1.7.

Wilson, F.G. 1946. Numerical expression of stocking in terms of height. Journal of Forestry. 44(10): 758–761. https://doi.org/10.1093/jof/44.10.758.

PRESCRIBED BURNING HAS LITTLE EFFECT ON LONGLEAF PINE WOOD QUALITY

Edward O. Andrews and John L. Willis

EXTENDED ABSTRACT

Longleaf pine (*Pinus palustris*) plays a vital role in the economy and ecology of the Southeastern United States. Prescribed burning is an important tool for perpetuating the longleaf pine ecosystem (Van Lear and others 2005); however, surprisingly little is known about how its repeated use affects important aspects of wood properties. Developing this knowledge is important because trees that contain the highest amount of latewood, wood density, and specific gravity will be the most valuable to the producer of structural lumber products (Jane 1970).

In 1973, a study was established to study the effects of prescribed fire on longleaf pine overstory growth at the Escambia Experimental Forest in Brewton, AL. The study was conducted in three pole-sized stands averaging 500 trees ha^{-1}. Each stand was segmented into 1.2-ha blocks and further subdivided into 0.10-ha plots. Plots were randomly assigned a treatment combination examining the effects of fire seasonality (winter, spring) and frequency (0, 2, 3, 5 years). Within each plot (n = 29), we sampled 10 trees (n = 270 total across all treatments) selected in a stratified random design across crown classes (dominant, codominant, intermediate). From each sampled tree, one 5-mm core was extracted from bark to pith. The cores were then placed in straws and transported back to the lab and stored in a freezer at 0 °C until processing. Each core was oven dried at 16 °C until mass remained constant and glued into 2-mm core mounts with a 2-mm spacer. Cores were placed in a conditioned chamber at 22 °C and 52 percent relative humidity to achieve an approximate wood moisture content of 10 percent. We then scanned the cores on a Qtrs-01x tree ring scanner (Quintek Measurement System, Knoxville, TN) with a 0.06-mm radial strip step resolution. The instrument was calibrated to express specific gravity for longleaf pine on an oven-dry weight and green volume basis (basic specific gravity). Latewood was differentiated from earlywood using a threshold specific gravity value of 0.048 (USDA Forest Service 1999). Differences in treatments were examined with a mixed-effects analysis of variance. Separate models were run examining the influence of fire return interval, canopy position, fire season, and the interaction between fire season and fire return interval. Plot was considered a random effect in the model.

Edward O. Andrews, Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Auburn AL 36849

John L. Willis, Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Auburn AL 36849

Preliminary results suggest that fire return interval, canopy position, fire seasonality, and the interaction between fire seasonality and burn interval had negligible effects on wood specific gravity. Similarly, burn interval, fire seasonality, and the interaction between fire seasonality and burn interval had no effect on latewood ring width. In contrast, canopy position significantly influenced latewood ring width, as dominant trees added statistically more latewood than codominant and intermediate trees. Collectively, these preliminary results demonstrate that prescribed burning in the spring and winter at frequent intervals had little impact on longleaf pine wood quality. Consequently, land managers burning within the range of conditions examined here will have minimal impacts on the structural integrity of longleaf pine.

LITERATURE CITED

Jane, F.W.; Wilson, K.; White, D.J.B. 1970. The structure of wood. 2nd ed., revised. London: Adam and Charles Black. 478 p.

U.S. Department of Agriculture, Forest Service [USDA Forest Service]. 1999. Wood handbook: wood as an engineering material. Gen. Tech. Rep. FPL-113. Madison, WI: U.S. Department of Agriculture Forest Service, Forest Products Laboratory. 463 p. https://doi.org/10.2737/FPL-GTR-113.

Van Lear, D.H.; Carroll, W.D.; Kapeluck, P.R.; Johnson, R. 2005. History and restoration of the longleaf pine-grassland ecosystem: implications for species at risk. Forest Ecology and Management. 211(1-2): 150–165. https://doi.org/10.1016/j.foreco.2005.02.014.

Poster Session

EVALUATING BIAS IN TREE-LEVEL MEASUREMENTS OBTAINED WITH A BACKPACK LIDAR UNIT

Michael K. Crosby and Joshua P. Adams

EXTENDED ABSTRACT

The ability to rapidly acquire biometric data in forests allows for timely decisions to be made regarding harvest, fertilization, prescribed fire, or other silvicultural treatments. High resolution tree-level data can be used to assess tree conditions and in calculating tree- or stand-level volume estimates. Using mobile, terrestrial light detection and ranging (LiDAR) units, such data collection is possible and provides an archive of present stand conditions. This study seeks to apply a mobile backpack LiDAR unit to a 3-year-old planted stand in northern Louisiana. The objective of the present study is to compare LiDAR-derived measurements with *in situ* measurements for height and groundline diameter.

An outplanting of American sycamore (*Platanus occidentalis*) was planted in 2019 on Louisiana Tech University's campus using a 6-foot by 10-foot spacing (19 rows and 22 columns). Groundline diameters (GLD) were measured using calipers to the nearest 0.01 inch and a height pole used to measure heights to the nearest 0.1 foot. A terrestrial LiDAR unit was used to measure the trees. The LiDAR backpack consists of a Snoopy A-Series 60 unit incorporating a Velodyne Puck (VLP-16 LITE) generating approximately 600,000 points per second. For this trial, the backpack-housed sensor was walked between the rows of planted trees. Subsequent to the collection and validation of the base-station location via OPUS (Online Positioning User Service; https://geodesy.noaa.gov/OPUS/), the data was processed using Inertial Explorer. Once the LiDAR scenes were created, the LiDAR datasets were imported into ArcGIS Pro and surface and canopy layers were calculated. Subtracting the canopy and surface layers, heights were obtained. A polygon was created to represent crown diameter and a maximum value function was used on each tree with the result used as the height. These were then compared to each measured tree and bias was calculated to determine comparability of the height measurements.

Height measurements were underestimated using the LiDAR dataset. Comparing the two methods, there was average negative bias of 0.43 feet for the LiDAR data. There is a systematic bias with increasing underestimation of heights for taller trees; the bias is twice as great for trees >10 feet tall (fig. 1). This could be the result of some smoothing in

Michael K. Crosby, Associate Professor, Louisiana Tech University, School of Agricultural Sciences and Forestry, Ruston, LA 71272

Joshua P. Adams, Associate Professor, Louisiana Tech University, School of Agricultural Sciences and Forestry, Ruston, LA 71272

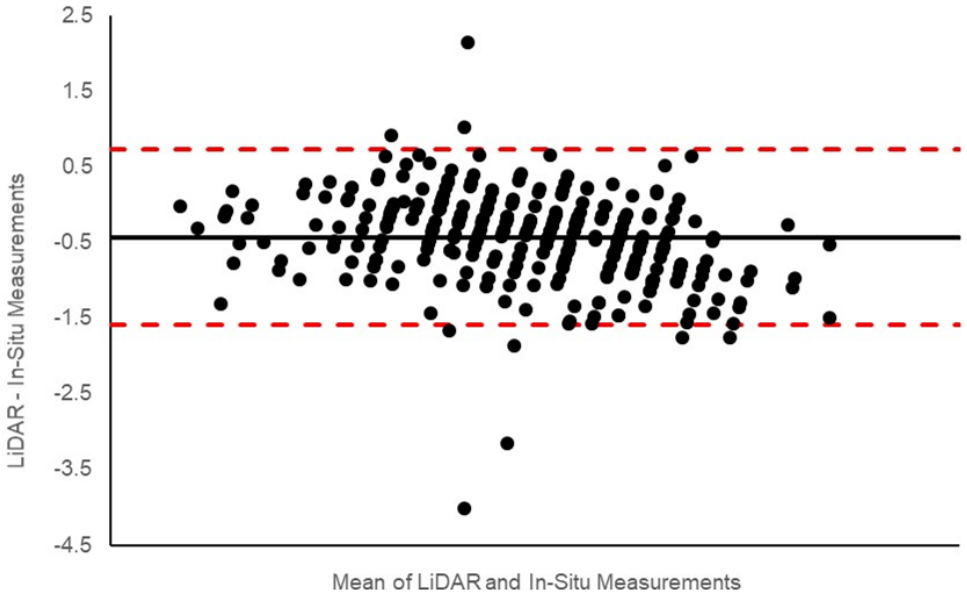

Figure 1—Bland-Altman plot comparing the two methods of measurement (LiDAR and *in situ*) for obtaining tree heights.

generating the raster grids for height estimation (resolution of the surfaces created for height estimation is 1 foot), or an indication sensor capability to detect tree heights in dense plantations. Given the spacing of the trees, small and narrow tops, and propagation of energy from the sensor, there may not be enough returns collected to accurately measure the top of the trees. Having two sensors scanning simultaneously may allow for more returns at height (Xie and others 2020, 2022) and allow for better height estimation.

For GLD measurements, there was a negative bias of 0.02 in for the LiDAR measurements. It is worth noting that given the density of understory vegetation and the presence of limbs at lower heights for some varieties, it was not possible to find enough points to measure an accurate diameter for all trees (hence 30 were selected). This could be an issue for GLD and diameter at breast height estimates if using LiDAR in a new planting because there may not be enough points penetrating foliage to accurately represent the bole of the tree. Leaf-off acquisition may help alleviate this issue. Measuring diameters one at a time is a cumbersome process that is likely most efficiently done in the field unless this can be automated. Artificial intelligence techniques will be explored in the future in an attempt to classify trees into diameter classes.

Previous work using terrestrial LiDAR has focused on mature forests. Incorporating such technology early provides an archive of stand conditions that can be used to monitor a variety of factors throughout the rotation. Future work will look to determine the potential of using a correction factor if bias is consistent and work to develop height-diameter relationships and test their validity in similar plantings as well as those of other species. It will also be important to consider collection and processing time to obtain measurements compared to *in situ* data collection in small plantings.

This study demonstrated the use of a terrestrial LiDAR unit for measuring tree heights and diameters. The results of this study provide a baseline for subsequent assessments using a terrestrial LiDAR unit. Data can be collected quickly and snapshots of trees documented for future comparison. In young stands, care should be taken if assessing total height.

ACKNOWLEDGMENTS

This project was supported by funding from the McIntire-Stennis program (Project # LAX-437626) and funding from the U.S. Department of Agriculture, Forest Service, Southern Research Station (Agreement # 17-JV-11330127-071).

LITERATURE CITED

Xie, Y.; Yang, T.; Wang, X. [and others]. 2022. Applying a portable backpack lidar to measure and locate trees in a nature forest plot: accuracy and error analyses. Remote Sensing. 14: 1806. 15 p. https://doi.org/10.3390/rs14081806.

Xie, Y.; Zhang, J.; Chen, X. [and others]. 2020. Accuracy assessment and error analysis for diameter at breast height measurement of trees obtained using a novel backpack LiDAR system. Forest Ecosystems. 7: 33. 11 p. https://doi.org/10.1186/s40663-020-00237-0.

CORRECTING THE ERRONEOUS GEOGRAPHIC COORDINATES OF THE SOUTHWIDE PINE SEED SOURCE STUDY SOURCE AREAS

Don C. Bragg and Shawna L. Reid

ABSTRACT

The Southwide Pine Seed Source Study (SPSSS) provided a tremendous amount of information from its inception in the early 1950s until its conclusion decades later. The silvicultural value of this genetics study, even given some design weaknesses and other limitations, continues to this day and these data will soon be available through the U.S. Department of Agriculture, Forest Service's Research Data Archive. However, while preparing these data, we discovered in the original study plan documentation that most of the geographic coordinates provided for the seed sources were erroneously located. While some of these errors were very minor, others had significantly different latitudes and longitudes. Such departure errors probably had little to no impact on the earlier interpretations of seed source performance; however, they may prove problematic if used in modern geographic analyses that require more precise sample locations. We demonstrate a number of these errors through a series of species-based tables and maps and provide improved coordinates for future researchers.

INTRODUCTION

As the first large-scale study of southern pine forest genetics, the Southwide Pine Seed Source Study (SPSSS) was developed in the early 1950s to learn about the variation in seed sources and use that knowledge to improve the restoration of the region's forests (Wakeley 1953). Remarkably little was known about the genetics of southern pines to that point, and tree improvement and widespread planting of pines were concepts still in development. Although the SPSSS study design is simple and limited, this research eventually proved highly valuable to forest geneticists and silviculturists, who used this study to refine and improve upon plantation management for decades.

One of the strengths of the SPSSS was its replicated, region-wide study design and statistical analysis of geographic-based variation for the four major southern pine species [longleaf (*Pinus palustris*), shortleaf (*P. echinata*), loblolly (*P. taeda*), and slash (*P. elliottii*)]. The SPSSS was installed across a range of sites designed to follow what were thought to be important regional gradients in temperature, precipitation, and soils for both seed sources and planting locations (Wakeley 1953). This required a monumental amount of coordination with many partners to collect sufficient quantities of pine seed across these gradients to produce meaningful comparisons—not an easy task in the 1950s, especially when faced with limited resources and some major droughts. These challenges led to a number of on-the-fly adjustments to study design and later analyses, which limited some of the interpretation of the results. Another limitation inherent to the SPSSS is that the seeds gathered for this study did not come from controlled collections from known parent trees, but rather were taken from rather large seed production areas that could encompass many thousands of acres. While this undoubtedly facilitated the collection of enough seed to install the planned replicates of outplantings, it also limited the degree of control in determining the actual

Don C. Bragg (deceased), Project Leader and Research Forester, U.S. Department of Agriculture, Forest Service, Southern Research Station, Monticello, AR 71656

Shawna L. Reid, Resource Information Specialist, U.S. Department of Agriculture, Forest Service, Southern Research Station, 233 Lehotsky Hall, Clemson University, Clemson, SC 29634

seed source. In other words, instead of having a handful of known trees specifically located on the ground, collections were made from a much broader seed production area.

The SPSSS data collection started in the 1950s and ran into the 1980s, and this extensive and consistently measured study represents a remarkable investment by the U.S. Department of Agriculture, Forest Service. Not surprisingly, the SPSSS has considerable long-term value as a research program. For example, the SPSSS loblolly pine outplantings were studied for decades on a variety of topics, from growth and yield, forest health, mortality, wood quality differences, and pine genetics (Buford 1986, Henry and Hepting 1957, McAlister and Clark 1991, Schmidtling and Froelich 1993, Wells 1983). Given that in recent years, geographic coordinates of genetic source materials have been used to further large-scale analyses (e.g., comparing wood properties with their origins and proximity to geographic features such as oceans or mountains or even latitude) (Jordan and others 2008), the Forest Service has continued to maintain the SPSSS datasets (Nelson and Schmidtling 2022) and has started to make them publicly available on the agency's Research Data Archives (project in progress).

As a part of this data archiving exercise, available metadata—information on the study beyond the data itself—are required to better document the work. This includes good geospatial information on seed source areas. For the SPSSS, locational information in the form of place descriptions (state, county, latitude, longitude, and a physiographic description) for each seed source was provided in a table from a supplement to the original working plan (SGSS 1956: table 1). We decided to use that information to develop a series of digital maps for the included southern pines and add the maps to the materials to be sent to the Forest Service's Research Data Archive. However, to our surprise, some of these plotted source locations taken from the latitude and longitude provided in SGSS (1956) were noticeably incorrect. If a researcher was to try to use seed source locations as a proxy for environmental conditions in some future silvicultural research, these errors may cause significant problems. Hence, this paper is our improvement of the geospatial information for these seed source locations.

METHODS

Our first step was to extract the original latitude and longitude (in degrees, minutes, and seconds) for the seed source (referred to as stock, as in seed or seedling stock) locations from table 1 in SGSS (1956). These coordinates were converted to decimal degrees and entered into a geographic information system (GIS) project using ArcGIS Pro software as X-Y tables; shapefiles were then created from each table. A map showing shortleaf pine seed source locations from the original 1956 coordinates was created, and immediately it became apparent that some kind of problem existed—some seed source locations were produced that made no sense. For instance, the 1956 coordinates situated some seed sources outside of their accepted natural distributions—this was the case for one shortleaf pine seed source area, which was placed in the Mississippi River Alluvial Plain (shortleaf pine is not native to that frequently flooded region). After double-checking the digital data to ensure their fidelity with the original table, maps were also created for longleaf, loblolly, and slash pines, and each of these also provided examples of substantial discrepancies from expectations. These likewise had obvious issues—for example, some coastal seed sources of loblolly and longleaf pine were placed in the Atlantic Ocean or Gulf of Mexico using the 1956 coordinates. While there did not appear to be any clear patterns of systemic displacements, the sheer number of anomalies suggested an underlying problem.

In order to identify and resolve an avoidable source of error—inaccurate seed area coordinates—we looked to the 1950s study documentation to reconstruct their coordinate assignment approaches. Obviously, when the original seed collections were made in the 1950s, precision, satellite-based global position systems (GPS) or comparable technologies were not available to those gathering the seeds in the field. The seed collectors knew their approximate

locations and were gathering seeds from a number of trees within a source area. Because accurate and precise locational information was not required under the original intent of the SPSSS, geographic references were only expected to be accurate to a county (or counties, for some source areas). Hence, the original approach was to assign a geographic coordinate (latitude and longitude) for the county seat in which the seed was sourced. There were a number of exceptions to that rule, however. For instance, when the seed was collected from a known smaller location (for example, the Crossett Experimental Forest), that more refined source area was to use those coordinates, rather than the county seat. Another exception is when seeds were gathered from several counties; in that case, a spot (sometimes a county seat, sometimes another location) was chosen as the seed source coordinate. As an example, a point along the boundary between the two counties was chosen for loblolly pine seed stock #309. Under a different application, for longleaf pine seed stock #107 the original coordinates were for the county between the two counties that were the actual source of the seeds. In a final example, longleaf pine seed stock #127's location was recorded as the common latitude and longitude for a point where three counties met.

Using those locational principles, modern coordinates for each county seat were extracted by taking official county seat locations from those published online or an alternative source, if needed. Because the original coordinates were in a nonprojected geographic coordinate system, all generated coordinates using the USA Contiguous Albers Equal Area Conic projection were used to produce a comparable set of coordinates. After creating a "modified" table with these updated locations for each seed source, an ArcGIS Pro process was used to calculate the distance, in miles, between the original coordinates and the county seat coordinates. A geoprocessing tool (Generate Near Table) was used to calculate these distances. After this processing was completed, a departure field was added for distance between the original (1956) coordinate data and the modified locations (in miles).

RESULTS AND DISCUSSION

Of the 87 SPSSS seed source areas considered, only one had the same latitude and longitude coordinates (within 0.1 mile of each other) between the original and modified values (table 1). Almost 83 percent were at least 10 miles different, and 57 percent were at least 20 miles apart. The highest departures for each species were more than 31 miles, and loblolly pine had one seed source area (#309) that was 36 miles different (table 2). Shortleaf pine had the highest average departure in its seed source coordinates (fig. 1, table 1). As noted earlier, loblolly pine (fig. 2) had the greatest individual departure but had the second lowest average departure. With a couple of notable exceptions, slash pine seed sources were not markedly different from their original placements and had (by more than 5 miles) the lowest average departure of these four southern pines (fig. 3, table 1).

Table 1—Description and distribution of departures between original and modified coordinates by species for the Southwide Pine Seed Source Study source locations

						Displacement category (miles)				
Species	Seed sources	Average	SD	Min	Max	0	>0–5	>5–10	>10–20	>20
	number	*~~~~~~~~~~~~ miles ~~~~~~~~~~~~*				*~~~~~~~~~~ number of source areas ~~~~~~~~~~*				
Longleaf	32	20.7	8.2	0.0	31.5	1	2	1	9	19
Shortleaf	30	21.3	8.7	3.8	32.8	0	3	1	8	18
Loblolly	16	20.5	7.2	3.8	35.9	0	1	1	4	10
Slash	9	15.3	13.0	1.8	32.8	0	3	2	1	3
Totals	87					1	9	5	22	50

SD = standard deviation; min = minimum; max = maximum.

Table 2—Departures between original SGSS (1956) and current (this study) seed source location assignments using county seats (in miles)

		Original		Modified (this study)		
Species	Stock number	Latitude	Longitude	Latitude	Longitude	Difference
		~~ *degrees* ~~~~~~		~~~~~~ *degrees* ~~~~~~		*miles*
Longleaf	102	36.43	-76.35	36.70	-76.63	24.6
Longleaf	103	34.37	-78.36	34.63	-78.61	22.8
Longleaf	104	34.37	-78.36	34.63	-78.61	22.8
Longleaf	105	34.56	-79.46	34.94	-79.76	31.5
Longleaf	107	34.11	-79.45	34.11	-79.45	0.0
Longleaf	108	34.11	-79.45	34.19	-79.78	19.3
Longleaf	109	34.44	-80.05	34.74	-80.09	20.7
Longleaf	111	32.22	-82.35	32.38	-82.59	17.8
Longleaf	112	32.22	-82.35	32.38	-82.59	17.8
Longleaf	114	27.57	-82.27	27.95	-82.46	28.4
Longleaf	115	30.45	-86.34	30.76	-86.57	25.5
Longleaf	117	30.53	-87.46	30.88	-87.77	30.7
Longleaf	119	33.39	-85.35	33.65	-85.59	22.5
Longleaf	122	30.36	-89.03	30.37	-89.09	3.8
Longleaf	123	30.50	-90.09	30.85	-90.15	24.3
Longleaf	125	31.18	-92.26	31.30	-92.46	14.6
Longleaf	126	31.18	-92.26	31.30	-92.46	14.6
Longleaf	127	30.42	-94.56	30.53	-94.55	7.4
Longleaf	151	36.43	-76.35	36.70	-76.63	24.6
Longleaf	153	34.37	-78.36	34.63	-78.61	22.8
Longleaf	155	34.56	-79.46	34.94	-79.76	31.5
Longleaf	157	34.11	-79.45	34.19	-79.78	19.3
Longleaf	161	34.44	-80.05	34.74	-80.09	20.7
Longleaf	163	32.22	-82.35	32.38	-82.59	17.8
Longleaf	167	27.57	-82.27	27.95	-82.46	28.4
Longleaf	169	30.45	-86.34	30.76	-86.57	25.5
Longleaf	171	30.53	-87.46	30.88	-87.77	30.7
Longleaf	175	32.38	-87.19	32.63	-87.32	19.0
Longleaf	177	30.36	-89.03	30.37	-89.09	3.8
Longleaf	179	30.50	-90.09	30.85	-90.15	24.3
Longleaf	181	31.18	-92.26	31.30	-92.46	14.6
Longleaf	183	30.42	-94.56	30.71	-94.94	29.8
Shortleaf	401	39.56	-77.39	39.93	-77.65	29.1
Shortleaf	403	39.59	-74.47	40.00	-74.79	32.8
Shortleaf	409	34.16	-81.37	34.38	-81.62	20.9

Species	Stock number	Original		Modified (this study)		Difference
		Latitude	Longitude	Latitude	Longitude	
		~~ *degrees* ~~~~~~		~~~~~~ *degrees* ~~~~~~		*miles*
Shortleaf	413	33.14	-84.16	33.25	-84.26	9.5
Shortleaf	415	32.50	-85.45	32.83	-85.76	29.3
Shortleaf	417	33.07	-89.03	33.12	-89.06	4.0
Shortleaf	419	34.22	-89.31	34.37	-89.52	15.6
Shortleaf	421	30.50	-90.40	30.83	-90.67	27.9
Shortleaf	423	31.20	-94.43	31.33	-94.72	19.7
Shortleaf	425	34.14	-95.37	34.24	-95.62	15.7
Shortleaf	427	34.07	-93.03	34.12	-93.05	3.8
Shortleaf	429	33.13	-91.48	33.03	-91.94	27.3
Shortleaf	431	35.52	-92.07	35.87	-92.12	24.4
Shortleaf	433	37.38	-91.32	37.65	-91.54	21.9
Shortleaf	435	36.06	-84.36	36.10	-84.60	13.6
Shortleaf	451	39.56	-77.39	39.93	-77.65	29.1
Shortleaf	453	39.59	-74.47	40.00	-74.79	32.8
Shortleaf	455	36.43	-77.04	36.72	-77.07	19.9
Shortleaf	457	34.43	-81.37	34.72	-81.63	24.9
Shortleaf	461	33.57	-83.22	33.96	-83.38	28.6
Shortleaf	463	33.19	-83.23	33.33	-83.39	13.2
Shortleaf	465	32.04	-84.32	32.07	-84.54	12.8
Shortleaf	467	32.50	-85.45	32.83	-85.76	29.3
Shortleaf	473	30.50	-90.40	30.83	-90.67	27.9
Shortleaf	475	31.47	-95.09	31.80	-95.15	22.8
Shortleaf	477	34.14	-95.37	34.24	-95.62	15.7
Shortleaf	478	33.53	-94.49	33.89	-94.83	31.8
Shortleaf	481	33.13	-91.48	33.03	-91.94	27.3
Shortleaf	485	37.38	-91.32	37.65	-91.54	21.9
Shortleaf	487	36.06	-84.08	36.11	-84.13	4.7
Loblolly	301	38.12	-75.41	38.20	-75.69	16.3
Loblolly	303	34.45	-77.26	34.75	-77.42	22.7
Loblolly	305	35.08	-76.46	35.14	-76.77	17.9
Loblolly	306	35.08	-76.46	35.14	-76.77	17.9
Loblolly	307	34.16	-81.37	34.38	-81.62	20.9
Loblolly	309	31.59	-83.18	31.96	-83.61	35.9
Loblolly	311	33.57	-83.22	33.96	-83.38	28.6
Loblolly	315	34.10	-86.50	34.17	-86.84	20.2
Loblolly	317	33.16	-85.50	33.27	-85.84	20.9
Loblolly	319	33.31	-86.48	33.52	-86.80	23.6

Species	Stock number	Original		Modified (this study)		Difference
		Latitude	Longitude	Latitude	Longitude	
		~~ degrees ~~~~~~		~~~~~~ degrees ~~~~~~		miles
Loblolly	321	34.39	-88.34	34.66	-88.57	22.6
Loblolly	323	30.30	-90.45	30.50	-90.75	22.8
Loblolly	325	31.20	-94.43	31.33	-94.72	19.7
Loblolly	327	34.07	-93.03	34.12	-93.05	3.8
Loblolly	329	35.15	-88.59	35.26	-89.00	24.1
Loblolly	331	33.14	-84.16	33.25	-84.26	9.5
Slash	201	32.54	-80.40	32.90	-80.67	29.2
Slash	203	30.17	-82.07	30.28	-82.12	8.3
Slash	204	30.17	-82.07	30.28	-82.12	8.3
Slash	205	27.53	-81.50	27.90	-81.84	32.8
Slash	206	27.53	-81.50	27.90	-81.84	32.8
Slash	207	31.31	-87.19	31.53	-87.32	17.0
Slash	209	30.36	-89.03	30.37	-89.09	3.8
Slash	210	30.36	-89.03	30.37	-89.09	3.8
Slash	211	30.28	-90.06	30.48	-90.11	1.8

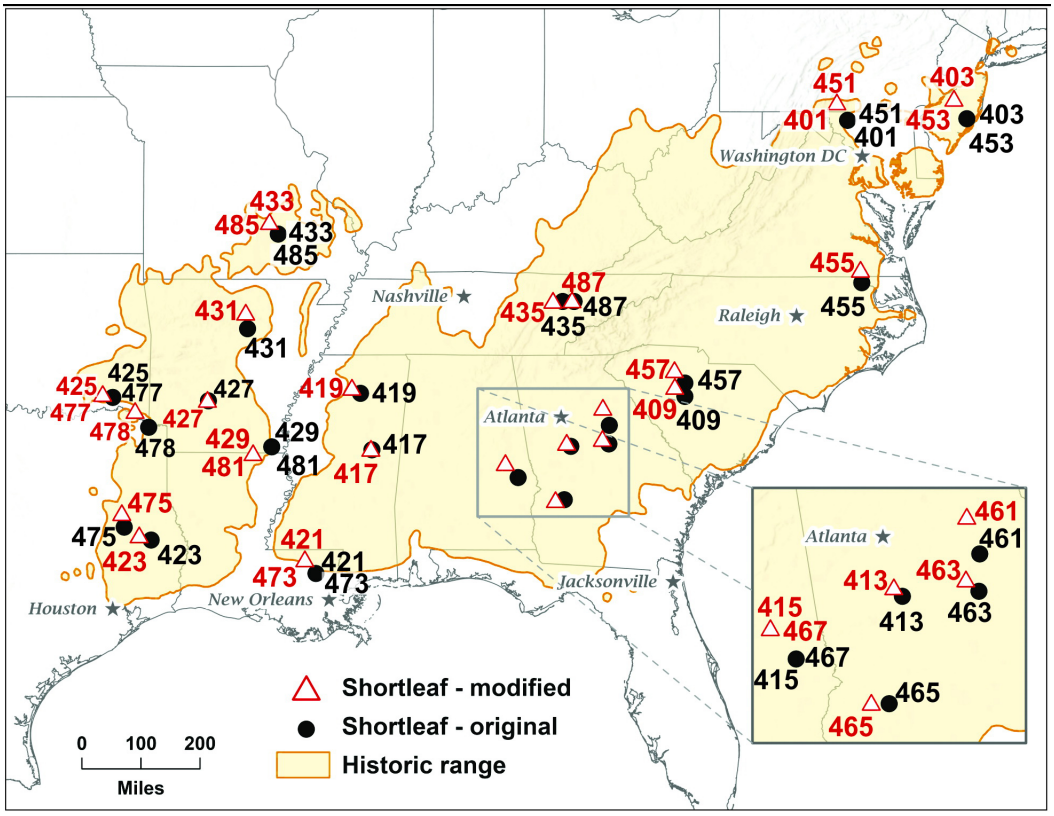

Figure 1—Map of the original Southwide Pine Seed Source Study shortleaf pine seed source locations using coordinates provided in SGSS (1956) and those using digitized latitude and longitude from the county seat locations as determined by a geographic information system. Inset map near Atlanta, GA, helps to clarify this dense cluster of seed sources.

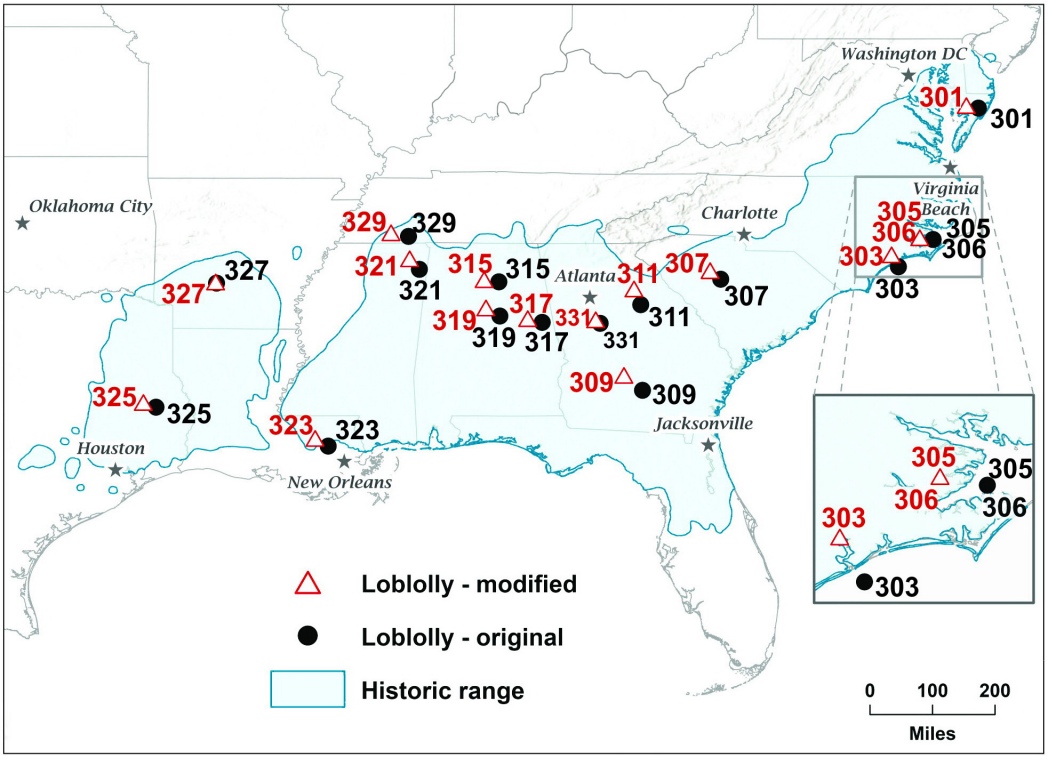

Figure 2—Map of the original Southwide Pine Seed Source Study loblolly pine seed source locations using coordinates provided in SGSS (1956) and those using digitized latitude and longitude from the county seat locations as determined by a geographic information system. The inset map of the North Carolina coast shows why the original, uncorrected latitude and longitudes are potentially quite problematic—three of the loblolly seed sources (#303, #305, and #306) were mapped using coordinates in the Atlantic Ocean.

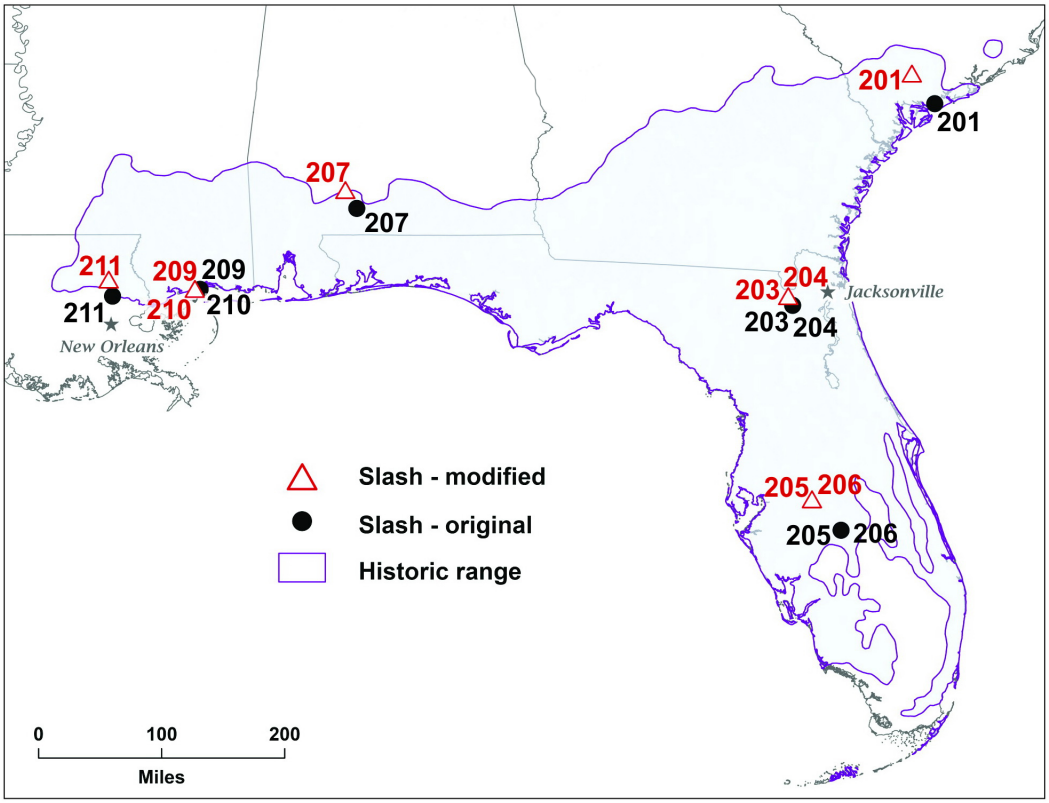

Figure 3—Map of the original Southwide Pine Seed Source Study slash pine seed source locations using coordinates provided in SGSS (1956) and those using digitized latitude and longitude from the county seat locations as determined by a geographic information system.

Given these were seed source areas rather than specific sites or locations, they were never intended to be very precise. However, such disparities between the intended locations and those coordinates provided in 1956 can be problematic. For instance, when longleaf pine was mapped (fig. 4), a high proportion of seed source areas were collected near the margins of the accepted botanical distribution of this species, and this could influence later interpretations of the performance of those sources.

A more critical issue than the distance between original and modified locations occurred in numerous locations when the assigned 1956 coordinates were actually in the ocean or other major waterbodies, or moved from one ecoregion (e.g., Upper West Gulf Coastal Plain) to another (Mississippi River Alluvial Plain). Each species had its own examples of these problematic displacements; rather than describing all of them, we selected a couple of examples to show why these displacements are problematic. For some reason, shortleaf pine seeds sourced from the Crossett Experimental Forest (#429 and #481) were not given coordinates of either the experimental forest or the county seat for which this experimental forest is found. Both the Crossett Experimental Forest and the Ashley County seat (Hamburg) are well situated on the Upper West Gulf Coastal Plain, which is good habitat for shortleaf pine. Rather, the #429 and #481 coordinates are for eastern Ashley County in the Mississippi River Alluvial Plain. The heavier textured, frequently flooded soils of the Mississippi River floodplain are highly unfavorable for shortleaf pine germination and growth, meaning that virtually no shortleaf pine of natural origin are found in this ecosystem. Any post-hoc geospatial analysis considering factors such as soil type and site hydrology would have gotten incorrect information for shortleaf pine.

All the other species had similar problems. For example, given the limited number of slash pine seed source areas in the SPSSS, placement near the coast (as for #201) versus a more appropriate location could markedly skew analyses. Several loblolly pine seed sources (for example, #303, #305, and #306) were originally placed in the

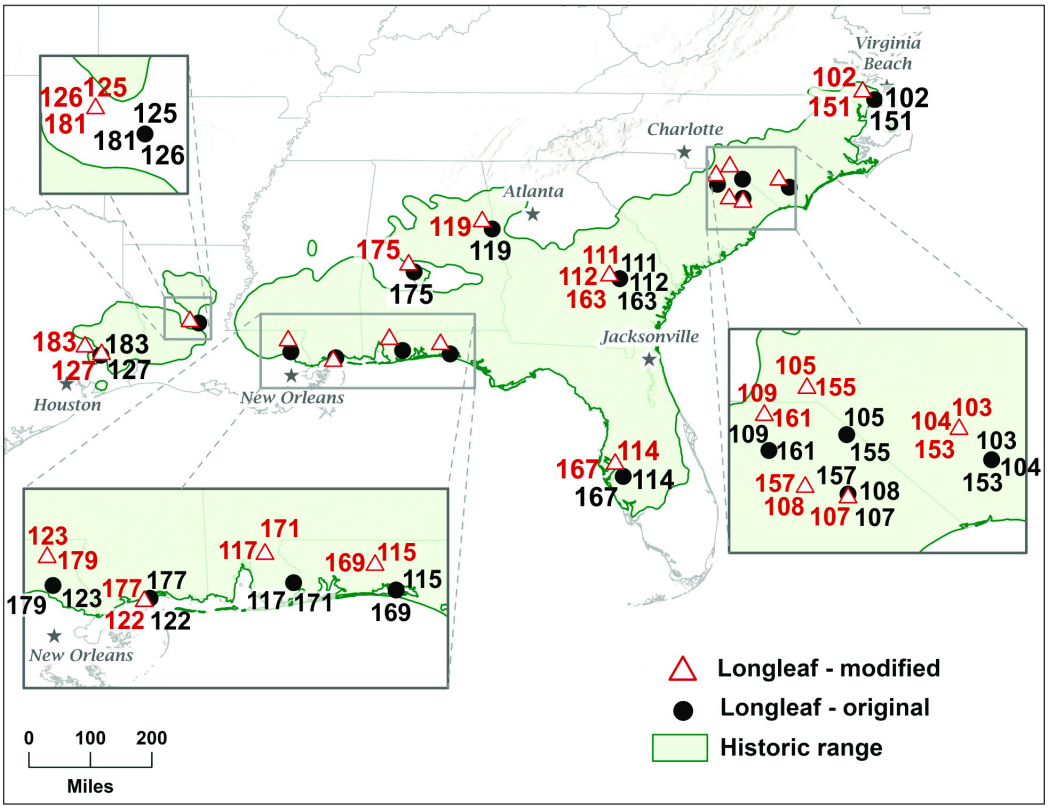

Figure 4—Map of the original Southwide Pine Seed Source Study longleaf pine seed source locations using coordinates provided in SGSS (1956) and those using digitized latitude and longitude from the county seat locations as determined by a geographic information system. The three inset maps are intended to make dense clusters of seed sources more visible and highlight how location errors could move collection points from the land to a body of water.

Atlantic Ocean off of the North Carolina coast when they should have been placed some miles inland (see inset in fig. 2). Such erroneous placements can produce significant errors in post-hoc geospatial or ecological analysis.

CONCLUSIONS

Unfortunately, there is no way of knowing why any of these errors arose. It seems likely that at least some were the result of incorrect readings or recordings of latitude and longitude. Others may have resulted from differing interpretations of precisely which county (or counties) to assign the original field samples to; these were seed source *areas*, after all, and precise sampling coordinates using satellite-based technology was still decades away. Regardless of the reasons, the updated coordinates provided in table 2 are more consistent with the stated intentions of the original SPSSS.

The SPSSS offers a wealth of data on the major southern pines and this "old" information could still provide even more insights into a dynamic, highly managed region of the country. With silvicultural research work increasingly dominated by meta-analyses of existing data sets, elimination of known errors is critical for improving these analyses. Hence, our correction of these seed source locations should help current and future silvicultural research.

ACKNOWLEDGMENTS

We would like to thank the following for their contributions to this project: Dana Nelson, Ron Schmidtling, and Chuck Burdine of the Southern Institute of Forest Genetics, Saucier, MS.

REFERENCES

Buford, M.A. 1986. Height-diameter relationships at age 15 in loblolly pine seed sources. Forest Science. 32(3): 812–818. https://doi.org/10.1093/forestscience/32.3.812.

Henry, B.W.; Hepting, G.H. 1957. Pest occurrences in 35 of the Southwide Pine Seed Source Study plantations during the first three years. Southern Forest Tree Improvement Committee Tech. Pap. 18. New Orleans, LA: U.S. Department of Agriculture, Forest Service, Southern and Southeastern Forest Experiment Stations in cooperation with Southern Forest Tree Improvement Committee. 7 p.

Jordan, L.; Clark, A., III; Schimleck, L.R. [and others]. 2008. Regional variation in wood specific gravity of planted loblolly pine in the United States. Canadian Journal of Forest Research. 38(4): 698–710. https://doi.org/10.1139/X07-158.

McAlister, R.H.; Clark, A., III. 1991. Effect of geographic location and seed source on the bending properties of juvenile and mature loblolly pine. Forest Products Journal. 41(9): 39–42.

Nelson, C.D.; Schmidtling, R.C. 2022. Personal communication. Research geneticists, Southern Institute of Forest Genetics, Saucier, MS 39574.

Schmidtling, R.C.; Froelich, R.C. 1993. Thirty-seven year performance of loblolly pine seed sources in eastern Maryland. Forest Science. 39(4): 706–721. https://doi.org/10.1093/forestscience/39.4.706.

Subcommittee on Geographic Source of Seed [SGSS]. 1956. Supplement No. 1 to the original working plan of September 12, 1952 for the Southwide Pine Seed Source Study. Report 15. New Orleans, LA: Southern Forest Tree Improvement Committee. 110 p.

Wakeley, P.C. 1953. The South establishes a major pine geographic seed source study. In: Proceedings of the second conference on southern forest tree improvement. New Orleans, LA: Southern Forest Tree Improvement Committee. 7 p.

Wells, O.O. 1983. Southwide Pine Seed Source Study—loblolly pine at 25 years. Southern Journal of Applied Forestry. 7(2): 63–71. https://doi.org/10.1093/sjaf/7.2.63.

DEVELOPING SUCCESSFUL MANAGEMENT STRATEGIES FOR INVASIVE PLANT SPECIES AT TOWNSEND BOMBING RANGE, GEORGIA

G. Ryan Bohannon, Trisha M. Markus, Yingzhou Tang, Gary S. Herndon, and G. Geoff Wang

ABSTRACT

Nonnative, invasive plant species threaten the ecological and military values provided by forests at Townsend Bombing Range in southeastern Georgia. To inform effective management of invasive plants, installation-wide transect surveys were conducted in the summer of 2022 to determine the presence and abundance of invasive plant species in multiple land-cover classes. Based on these surveys, densities of major invasive plant species were greater in stand edges than in stand interiors. Japanese climbing fern (*Lygodium japonicum*) and Chinese tallow (*Triadica sebifera*) were concentrated on the edges of hardwood forest stands and managed pine (*Pinus* spp.) stands, respectively. Control methods for recorded species will be tested to develop targeted, effective management of high-priority invasive plants.

INTRODUCTION

Nonnative, invasive plant species are altering the composition and function of ecosystems throughout the Southern United States (Pile and others 2017b, Webster and others 2006). Invasive plants impact forest productivity and ecosystem services, making invasive species management essential in silviculture and ecological restoration (Webster and others 2006, Weidlich and others 2020). Invasive plants can also interfere with land management for military training on installations by impeding access and increasing fire risks. Townsend Bombing Range, an installation in southeastern Georgia, is managed to preserve its military mission while conserving natural resources through ecologically appropriate land management practices (DOD 2016). The impacts of invasive plant species threaten the ecological and military values provided by Townsend Bombing Range and impede conservation and restoration of the natural resources it holds. By outcompeting native plant species, invasive plants can reduce native plant growth and diversity and necessitate increasingly high management costs (Camarillo and others 2015, Miller and others 2010, Pile and other 2017b, Wang and others 2012). These species can also alter ecosystem functions such as fire regimes, interfering with the installation's ability to manage wildfire risk. For example, Japanese climbing fern (*Lygodium japonicum*) can increase the risk of high-intensity crown fires by acting as a ladder fuel (Minogue and others 2016), while Chinese tallow (*Triadica sebifera*) can suppress the use of prescribed fire for fuel management (Grace and others 2005; Pile and others 2017b, 2017c).

To sustain the ecological and military values provided by forests at Townsend Bombing Range, this project was initiated to develop an effective management plan for invasive plant species present on this land. A thorough understanding of the identity, abundance, and location of invasive plants is necessary to inform successful

G. Ryan Bohannon, Graduate Student, Clemson University, Department of Forestry and Environmental Conservation, Clemson, SC 29634
Trisha M. Markus, Graduate Student, Clemson University, Department of Forestry and Environmental Conservation, Clemson, SC 29634
Yingzhou Tang, Visiting Graduate Student, Clemson University, Department of Forestry and Environmental Conservation, Clemson, SC 29634
Gary S. Herndon, Natural and Cultural Resources Manager, U.S. Marine Corps Air Station Beaufort, Beaufort, SC 29904
G. Geoff Wang, Professor, Clemson University, Department of Forestry and Environmental Conservation, Clemson, SC 29634

management. An updated inventory of invasive plants would facilitate the identification of the highest-priority invasive species and the development of targeted management strategies to control them. To this end, we developed and implemented field surveys to inventory the abundance and distribution of invasive plant species across the installation.

MATERIALS AND METHODS

Study Site

This study took place at Townsend Bombing Range, an installation located in Long and McIntosh Counties, GA, and used for military training by U.S. Marine Corps Air Station Beaufort. Townsend Bombing Range lies within the Southern Coastal Plain ecoregion (EPA 2022) and is adjacent to the Altamaha River. The study area comprises approximately 13 692 ha after the original 2097-ha range was expanded in 2017. To represent the diversity of the focal landscape, the study area was divided into four land-cover classes: managed pine (*Pinus* spp.) stands, hardwood forest stands, impact areas, and fields. Managed pine stands consisted primarily of former industrial forest lands that were managed for pine production before their acquisition in 2017. Hardwood stands consisted of mixed upland and bottomland hardwood forests. Impact areas were actively used for air-to-ground military training and fields were open, nonforested land.

Surveys

Transect surveys were conducted in the summer of 2022. Starting points of transects were established within the four defined land-cover classes. We attempted a stratified random sampling method based on the land-cover classes, although this was limited in practice by a lack of accurate land-cover data prior to the surveys. Randomly located survey starting points were moved to the nearest road for easier access. Originating at each survey starting point, we paced 250 m into the interior of each stand perpendicular to the road. We then paced 250 m parallel to the road, 250 m to return to the road, and 250 m along the road to return to the starting point. This created a square perimeter of four 250-m transects at 90-degree angles to one another. The width of transects varied based on vegetation density. Transects in recently burned, mature pine stands were 10 m wide, while transects in denser areas were 5 m wide. This choice of transect width ensured that invasive plants could be easily detected while traveling efficiently along each transect. When an invasive plant was encountered, we recorded coordinates and either the estimated coverage area for fern, shrub, and vine species or the number of stems and largest stem diameter for tree species.

Data Analysis

Invasive plants may be more abundant along roads and forest edges due to increased propagule pressure and disturbance (Fan and others 2021, Huebner and Tobin 2006, Iverson and others 2019). To account for this potential edge effect, survey results were analyzed separately for the edge and interior of each land-cover class. First, a buffer was created at the edges of each stand beginning at the road boundary and extending 50 m into the stand. This 50-m buffer was classified as edge, and the remaining area in the core of each stand was classified as interior. All survey data that was collected within the 50-m edge of a stand was then separated from data collected in the stand interior. Survey transects that extended across both the edge and interior of a stand were divided into two segments, and edge and interior segments were considered separately.

Edge and interior density estimates of each plant species were calculated across each land-cover class. The total abundance of each plant species in stand edges and stand interiors was summed for each class. Based on the growth form of the plant species, abundance was calculated as either the coverage area for fern, shrub, and vine species or the number of stems for tree species. These totals were then divided by the total sampled area in stand edges or stand interiors for each land-cover class to produce density estimates for the entire study area.

RESULTS AND DISCUSSION

Five invasive plant species were recorded in transect surveys across 186.7 ha in 2022 (table 1). Japanese climbing fern and Chinese tallow were the most commonly recorded species, and two or fewer observations of mimosa (*Albizia julibrissin*), thorny olive (*Elaeagnus pungens*), and Callery pear (*Pyrus calleryana*) were recorded. Every recorded invasive plant species was observed in managed pine stands, and every species except thorny olive and Callery pear was observed in hardwood stands. None of the species were observed in impact areas or fields.

Density estimation revealed important trends in the abundance and distribution of different plant species both within and among stands of different land-cover classes. Overall, installation-wide density estimates for plant species with more than two observations were greater in stand edges than in stand interiors (figs. 1 and 2). Forests in the study area are dissected by a network of roads, and distance from roads is an important driver of forest invasibility by creating pathways for propagules to enter ecosystems (Fan and others 2021, Vieira and others 2014, Wang and others 2011). Japanese climbing fern was found almost exclusively on the edges of hardwood forest stands (fig. 1). Japanese climbing fern can establish dense populations in diverse forest types, including planted pine stands, bottomland hardwood-cypress (*Taxodium distichum*) forests, and longleaf pine (*Pinus palustris*) forests (Bohn and others 2011, Minogue and others 2010, Van Loan 2006), so the current distribution of Japanese climbing fern in the study area may be a result of its recent establishment rather than ecological limitations.

Table 1—Invasive plant species recorded at Townsend Bombing Range, GA, from transect surveys in summer 2022

Common name	Scientific name	Number of observations
Mimosa	*Albizia julibrissin*	2
Thorny olive	*Elaeagnus pungens*	1
Japanese climbing fern	*Lygodium japonicum*	11
Callery pear	*Pyrus calleryana*	1
Chinese tallow	*Triadica sebifera*	13

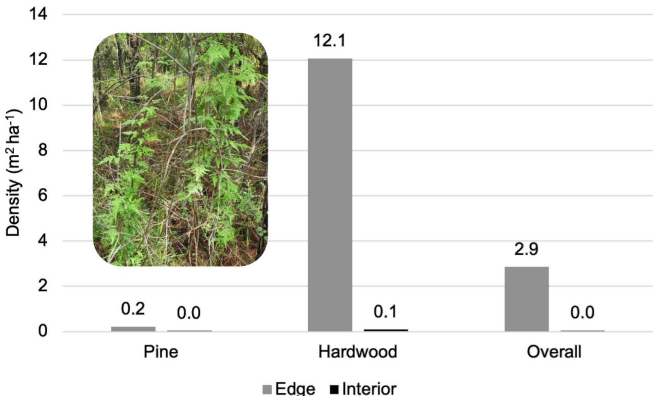

Figure 1—Density estimates of Japanese climbing fern (*Lygodium japonicum*) in the edge and interior of managed pine and hardwood forest stands. (Photo courtesy of Trisha Markus, Clemson University)

Chinese tallow, the most abundant invasive tree species recorded at Townsend Bombing Range, was heavily concentrated on the edges of managed pine stands (fig. 2). Many pine stands at Townsend Bombing Range are regenerating following harvesting and planting, and high light availability and soil disturbance in such stands may favor invasive plant establishment. Management activities such as harvests and thinnings can promote the establishment of invasive trees by altering resource availability and competition with native plants (Iverson and others 2019, Lockwood and others 2013, Pile and others 2017a, Rebbeck and others 2017). Chinese tallow is tolerant of a wide range of environmental conditions and has a prolific reproductive capacity (Pile and others 2017b), so further dispersal of this species into other land-cover types is likely despite its limited current distribution.

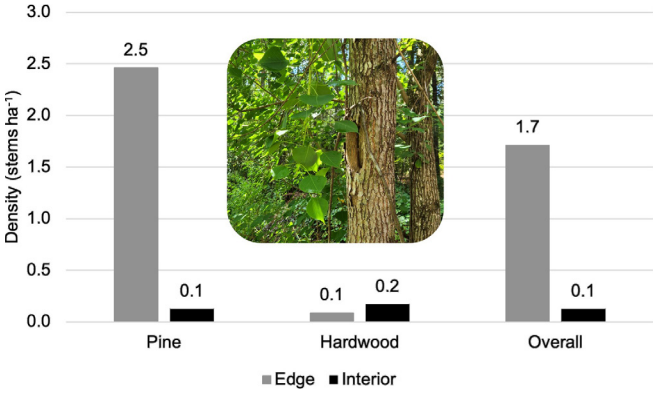

Figure 2—Stem density estimates of Chinese tallow (*Triadica sebifera*) in the edge and interior of managed pine and hardwood forest stands. (Photo courtesy of Trisha Markus, Clemson University)

CONCLUSIONS

Dominant and emerging invasive plant species identified at Townsend Bombing Range will have increasingly negative ecological and operational impacts if not effectively managed. While successful management of invasive plant species can be challenging to achieve (Weidlich and others 2020), a number of potential control options are available for these species (Miller and others 2013). Survey results characterized the ecology, abundance, and distribution of important invasive species in the study area, and this information can inform the judicious use of multiple strategies, including mechanical control, chemical control, and prescribed fire, for their management. Populations of many species detected in transect surveys appear to be currently present in the study area at low densities or small spatial extents. Continued monitoring, especially along stand edges, would support early detection and rapid response to eradicate such populations or maintain small populations below a damaging level. To inform an integrated pest management plan for invasive plant species, control methods identified for established species will be tested in the field to determine their effectiveness. By developing targeted, effective management techniques for high-priority species, this work will inform the sustained, successful management of invasive plants at Townsend Bombing Range.

ACKNOWLEDGMENTS

This project was funded by the U.S. Department of Defense. We thank Brandon Pritchett (Townsend Bombing Range), Christopher Petersen (Naval Facilities Engineering Systems Command), and range control personnel for assistance. We also thank Robert Jetton and Kelly Oten for providing helpful comments that improved this manuscript.

LITERATURE CITED

Bohn, K.K.; Minogue, P.J.; Pieterson, E.C. 2011. Control of invasive Japanese climbing fern (*Lygodium japonicum*) and response of native ground cover during restoration of a disturbed longleaf pine ecosystem. Ecological Restoration. 29: 346–356. https://doi.org/10.3368/er.29.4.346.

Camarillo, S.A.; Stovall, J.P.; Sunda, C.J. 2015. The impact of Chinese tallow (*Triadica sebifera*) on stand dynamics in bottomland hardwood forests. Forest Ecology and Management. 344: 10–19. https://doi.org/10.1016/j.foreco.2015.02.013.

Environmental Protection Agency [EPA]. 2022. Ecoregions of North America. https://www.epa.gov/eco-research/ecoregions-north-america. [Date accessed: May 28, 2023].

Fan, Z.; Yang, S.; Cheng, N. [and others]. 2021. Invasibility of fire-managed ecosystems to the Chinese tallow tree (*Triadica sebifera*) in the lower Gulf Coastal Plain, USA: mechanisms and key factors at the landscape level. Forest Ecology and Management. 479: 118539. 10 p. https://doi.org/10.1016/j.foreco.2020.118539.

Grace, J.B.; Allain, L.K.; Baldwin, H.Q. [and others]. 2005. Effects of prescribed fire in the coastal prairies of Texas. Open File Rep. 2005-1287. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. 46 p.

Huebner, C.D.; Tobin, P.C. 2006. Invasibility of mature and 15-year-old deciduous forests by exotic plants. Plant Ecology. 186: 57–68. https://doi.org/10.1007/s11258-006-9112-9.

Iverson, L.R.; Rebbeck, J.; Peters, M.P. [and others]. 2019. Predicting *Ailanthus altissima* presence across a managed forest landscape in southeast Ohio. Forest Ecosystems. 6: 41. 13 p. https://doi.org/10.1186/s40663-019-0198-7.

Lockwood, J.L.; Hoopes, M.F.; Marchetti, M.P. 2013. Invasion ecology. 2nd ed. West Sussex, UK: John Wiley. 667 p.

Miller, J.H.; Chambliss, E.B.; Lowenstein, N.J. 2010. A field guide for the identification of invasive plants in southern forests. Gen. Tech. Rep. SRS-119. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 126 p.

Miller, J.H.; Manning, S.T.; Enloe, S.F. 2013. A management guide for invasive plants in southern forests. Gen. Tech. Rep. SRS-131. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 120 p. https://doi.org/10.2737/SRS-GTR-131.

Minogue, P.J.; Bohn, K.K.; Osiecka, A.; Lauer, D.K. 2010. Japanese climbing fern (*Lygodium japonicum*) management in Florida's Apalachicola bottomland hardwood forests. Invasive Plant Science and Management. 3: 246–252. https://doi.org/10.1614/IPSM-D-09-00023.1.

Minogue, P.J.; Jones, S.; Bohn, K.K.; Chevasco, E.D. 2016. Biology and control of Japanese climbing fern (*Lygodium japonicum*). FOR 218. Revised. Gainesville, FL: University of Florida, Institute of Food and Agricultural Sciences Extension. 5 p.

Pile, L.S.; Wang, G.G.; Knapp, B.O. [and others]. 2017a. Chinese tallow (*Triadica sebifera*) invasion in maritime forests: the role of anthropogenic disturbance and its management implication. Forest Ecology and Management. 398: 10–24. https://doi.org/10.1016/j.foreco.2017.05.006.

Pile, L.S.; Wang, G.G.; Stovall, J.P. [and others]. 2017b. Mechanisms of Chinese tallow (*Triadica sebifera*) invasion and their management implications—a review. Forest Ecology and Management. 404: 1–13. http://dx.doi.org/10.1016/j.foreco.2017.08.023.

Pile, L.S.; Wang, G.G.; Waldrop, T.A. [and others]. 2017c. Managing an established tree invader: developing control methods for Chinese tallow (*Triadica sebifera*) in maritime forests. Journal of Forestry. 115(6): 522–529. https://doi.org/10.5849/JOF-2016-022R2.

Rebbeck, J.; Hutchinson, T.; Iverson, L. [and others]. 2017. Distribution and demographics of *Ailanthus altissima* in an oak forest landscape managed with timber harvesting and prescribed fire. Forest Ecology and Management. 401: 233–241. http://dx.doi.org/10.1016/j.foreco.2017.06.050.

U.S. Department of Defense [DOD]. 2016. Integrated natural resources management plan, Townsend Bombing Range, McIntosh and Long Counties, Georgia, Plan Years 2017–2022. Townsend, GA: U.S. Department of Defense. 180 p.

Van Loan, A.N. 2006. Aspects of the invasion and management of Japanese climbing fern (*Lygodium japonicum*) in southeastern forests. M.S. thesis. Gainesville, FL: University of Florida. 119 p.

Vieira, R.; Finn, J.T.; Bradley, B.A. 2014. How does the landscape context of occurrence data influence models of invasion risk? A comparison of independent datasets in Massachusetts, USA. Landscape Ecology. 29: 1601–1612. http://dx.doi.org/10.1007/s10980-014-0080-5.

Wang, H.-H.; Grant, W.E.; Gan, J. [and others]. 2012. Integrating spread dynamics and economics of timber production to manage Chinese tallow invasions in southern U.S. forestlands. PLOS ONE. 7(3): e33877. 15 p. https://doi.org/10.1371/journal.pone.0033877.

Wang, H.-H.; Grant, W.E.; Swannack, T.M. [and others]. 2011. Predicted range expansion of Chinese tallow tree (*Triadica sebifera*) in forest lands of the southern United States. Diversity and Distributions. 17(3): 552–565. https://doi.org/10.1111/j.1472-4642.2011.00760.x.

Webster, C.R.; Jenkins, M.A.; Jose, S. 2006. Woody Invaders and the Challenges They Pose to Forest Ecosystems in the Eastern United States. Journal of Forestry. 104(7): 366–374. https://doi.org/10.1093/jof/104.7.366.

Weidlich, E.W.A.; Flórido, F.G.; Sorrini, T.B.; Brancalion, P.H.S. 2020. Controlling invasive plant species in ecological restoration: a global review. Journal of Applied Ecology. 57: 1806–1817. https://doi.org/10.1111/1365-2664.13656.

EFFECTIVENESS OF SHELTERWOOD AND STRIP CLEARCUTS FOR REGENERATING BOTTOMLAND OAKS IN NORTHEAST TEXAS

Cason Brewster, Robert Heinemann, Randal Holeman, Cassandra Meek, Keith Anderson, Josh Everett, Omkar Joshi, and Rodney Will

ABSTRACT

Oaks (*Quercus* spp.) are frequently the most valuable bottomland species, but are often hard to regenerate. In northeastern Texas, operational shelterwoods and strip clearcuts were established in 2014 to test their effectiveness for establishing oak regeneration. Measured in 2022, basal area of the shelterwood areas recovered to 96.2 square feet per acre and the sapling size class [diameter at breast height (d.b.h.) <2 inches] increased from an initial 270 trees per acre (TPA) to 580 TPA with 26 percent oaks. The seedling layer increased to 7,371 TPA with 67 percent willow oak (*Q. phellos*), 10 percent overcup oak (*Q. lyrata*), and 9 percent water oak (*Q. nigra*). In the strip clearcuts, no seedlings occurred in the sampling plots. However, there were 961 TPA in the small sapling size class (<2 inches d.b.h.) with 62 percent *Quercus* spp. These results indicate that both techniques successfully enhanced oak regeneration. However, additional treatments are likely needed to ensure recruitment of oaks into the overstory.

INTRODUCTION

Oaks (*Quercus* spp.) are among the most valuable bottomland species but are often difficult to regenerate. The typical regeneration prescription for oaks is a shelterwood, which partially opens the canopy to provide intermediate light conditions. However, planning and implementation of shelterwoods can be challenging; other practices such as strip clearcuts may provide a simpler alternative. The goal of this study was to compare the effectiveness of operational strip shelterwoods and strip clearcuts for establishing oak regeneration in a bottomland area of northeastern Texas.

METHODS

The study was located near Daingerfield, TX (32°58’40” N, 94°48’23” W). Average annual precipitation was 46.8 inches per year and mean annual air temperature was 64.4 °F. The soils were mostly Estes series (Fine, smectitic, thermic, Aeric Dystraquerts) which are very deep, somewhat poorly drained floodplain soils with clay/clay loam texture. The site typically floods between 2 and 60 days per year during the dormant season.

In 2014, alternating areas of strip clearcuts and strip shelterwoods (three each) were installed. Sizes ranged from 80 to 100 m wide by 400 to 800 m long. In the shelterwood strips, the establishment cut reduced basal area from 94 square feet per acre to between 50 and 60 square

Cason Brewster, Forester, Hurliman Forestry Services, Broken Bow, OK 74728
Robert Heinemann, Superintendent, Oklahoma State University, Kiamichi Forest Research Center, Idabel, OK 74745
Randal Holeman, Senior Research Specialist, Oklahoma State University, Kiamichi Forest Research Center, Idabel, OK 74745
Cassandra Meek, Assistant Superintendent, Oklahoma State University, Kiamichi Forest Research Center, Idabel, OK 74745
Keith Anderson, Senior Research Specialist, Oklahoma State University, Kiamichi Forest Research Center, Idabel, OK 74745
Josh Everett, Graduate Student, Oklahoma State University, Department of Natural Resources and Management, Stillwater, OK 74078
Omkar Joshi, Associate Professor, Oklahoma State University, Department of Natural Resources and Management, Stillwater, OK 74078
Rodney Will, Professor, Oklahoma State University, Department of Natural Resources and Management, Stillwater, OK 74078

feet per acre. Five or six sampling plots were established in each cutting unit. Preharvest data were collected in the summer of 2014. Posttreatment data were collected in October 2022. For preharvest data and shelterwood posttreatment data, stems >2 inches d.b.h. (overstory and midstory) were sampled within 0.1-acre plots, stems <2 inches d.b.h. (small saplings) were sampled in 0.01-acre plots nested within the larger plot, and seedlings (<4.5 feet) were sampled from five 0.001-acre subplots. Subplots were located at plot center and 10 feet in each cardinal direction from plot center. Because of a general lack of overstory within the strip clearcut units, all stems taller than 4.5 feet were sampled within 0.01-acre plots and five 0.001-acre subplots were used to sample seedlings. Diameters, heights, and species were measured for the overstory and sapling layers. The heights of all seedlings were measured. Plot-level data within a given cutting unit were averaged before analysis, such that the cutting unit served as the experimental unit (n = 3).

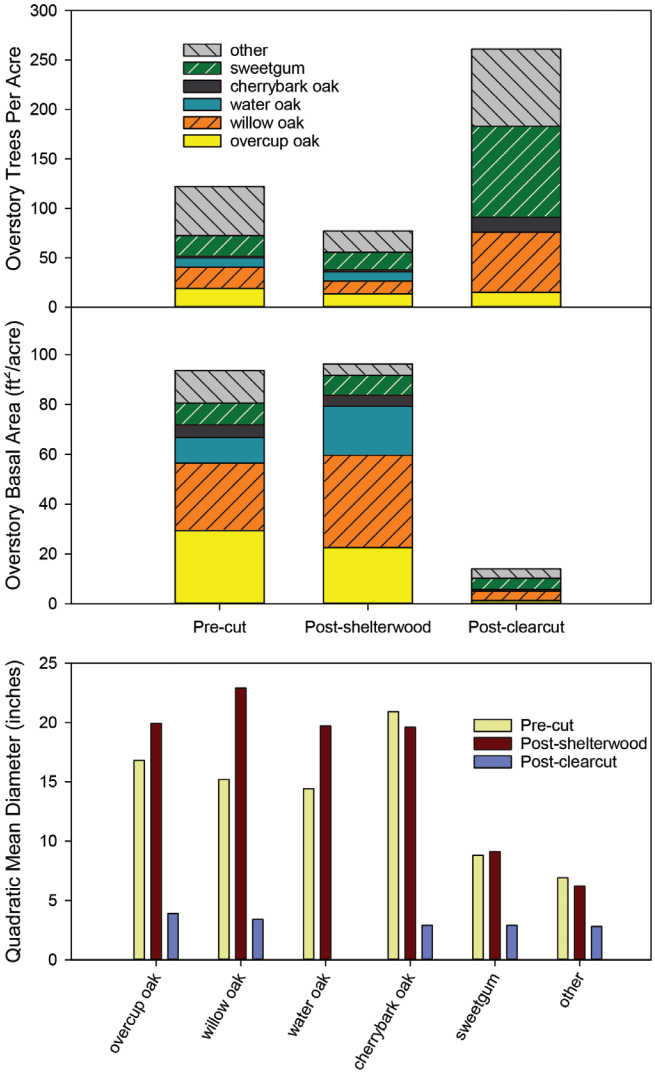

Figure 1—Overstory and midstory trees (>2 inches d.b.h.) measured in 2014 (precut) and in 2022; (top) trees per acre, (middle) basal area, and (bottom) quadratic mean diameter.

RESULTS

In 2014, the initial unmanaged stand had 93.6 square feet per acre basal area (BA) and 122 TPA. The basal area of the overstory and midstory was composed of 31 percent overcup oak (*Quercus lyrata*), 29 percent willow oak (*Q. phellos*), 11 percent water oak (*Q. nigra*), 9 percent sweetgum (*Liquidambar styraciflua*), 5 percent cherrybark oak (*Q. pagoda*), and 14 percent other species. Quadratic mean diameter of trees >2 inches was 11.9 inches with the mean for oak species exceeding 14 inches (fig. 1). The small sapling layer consisted of 270 TPA. Possumhaw (*Ilex decidua*) was most abundant (29 percent), and oaks composed only 6 percent (fig. 2). Before treatment, there were 1,965 seedlings per acre with willow oak most abundant (38 percent). Other common species were hophornbeam (*Ostrya virginiana*) (11 percent), *Ulmus* spp. (9 percent), hornbeam (*Carpinus caroliniana*) (6 percent), overcup oak (6 percent), *Celtis* spp. (5 percent), water oak (3 percent), green ash (*Fraxinus pennsylvanica*) (3 percent), and sweetgum (2 percent) (fig. 2).

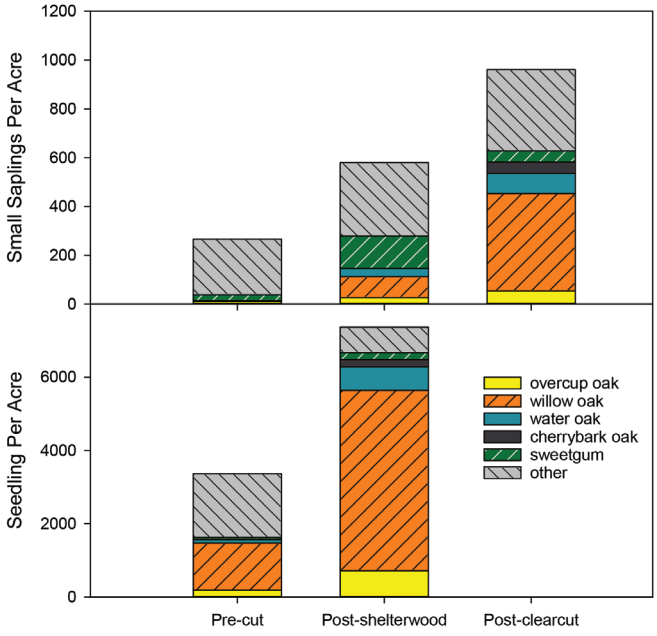

Figure 2—(top) Small saplings (<2 inches d.b.h) and (bottom) seedlings (shorter than 4.5 feet) measured in 2014 (precut) and in 2022.

Measured in 2022, BA of the shelterwood areas recovered to 96.2 square feet per acre in the overstory and midstory and was composed of 38 percent willow oak, 24 percent overcup oak, 20 percent water oak, 8 percent sweetgum, 5 percent cherrybark oak, and 5 percent other species. The TPA was 77. Quadratic mean diameter had increased to 15.1 inches and the mean of the oak species exceeded 19 inches (fig. 1). BA of the sapling layer was 2.73 square feet per acre consisting of 580 TPA. Of the trees, there were 23 percent sweetgum, 18 percent *Ulmus* spp., 17 percent *Cornus* spp., 15 percent willow oak, 9 percent possumhaw, 6 percent water oak, 5 percent overcup oak, and 7 percent other species. The seedling layer increased to 7,371 seedlings per acre and consisted of 67 percent willow oak (9.4 inches average height), 10 percent overcup oak (10 inches average height), 9 percent water oak (10 inches average height), 5 percent *Ulmus* spp. (13.2 inches average height), 3 percent cherrybark oak (9.7 inches average height), 3 percent sweetgum (17 inches average height).

For the strip clearcut areas measured in 2022, the overstory and midstory had BA of 15.4 square feet per acre, TPA of 261, and quadratic mean diameter of 3.3 inches. Of the TPA, there were 35 percent sweetgum, 24 percent willow oak, 21 percent black willow (*Salix nigra*), 6 percent cherrybark oak, 6 percent overcup oak, and 6 percent *Ulmus* spp. Basal area of sapling layer was 6.6 square feet per acre with 961 TPA. Of the trees, there were 41 percent willow oak (16 feet average height), 11 percent *Ulmus* spp. (13 feet average height), 10 percent water oak (17 feet average height), 6 percent overcup oak (18 feet average height), 5 percent cherrybark oak (9 feet average height), 5 percent sweetgum (10 feet average height), 5 percent green ash (11 feet average height), and 17 percent other. No seedlings were observed. The understory was composed of abundant herbaceous plants.

DISCUSSION

Given the shade intolerance of willow oak (Schlaegel 1990), water oak (Vozzo 1990), and cherrybark oak (Krinard 1990) and intermediate shade tolerance of overcup oak (Soloman 1990), very few oaks initially occurred in the sapling or seedling layers, indicating the need for increasing light availability to facilitate natural oak regeneration. Following the reduction in BA to 50–60 square feet per acre, the shelterwood establishment cut increased oak saplings from 16 per acre to 150 per acre. Seedlings increased 3.75 times, from 1,570 per acre to 6,484 per acre. However, the majority of the seedlings were short; 64 percent were less than 10 inches tall. In addition, growth of residual overstory trees was rapid and BA exceeded precut levels within 8 years. Because shade intolerant bottomland oaks, such as cherrybark oak, need 50 percent full sun for optimal growth (Gardiner and Hodges 1998), a second shelterwood establishment cut is probably necessary for continued development of the current sapling and seedling cohorts. Alternatively, a removal cut could be considered with reliance on existing oak saplings, release of current oak seedlings, and stump sprouts to provide adequate regeneration. Targeted herbicide application to control species such as sweetgum, possumhaw and *Cornus* spp. will help recruitment of oak into the sapling and midstory layers.

The strip clearcut produced about 600 oaks per acre in the sapling layer (between 0.1 and 2 inches d.b.h.) that likely will contribute to the next stand. This should be sufficient given a goal of 100–200 TPA of a mature overstory. Given the general lack of oaks in the sapling layer precut (<20 per acre), the oak saplings came from the release of seedlings or stump sprouts. Aggressive herbaceous growth following the clearcut resulted in a lack of seedlings and increased the reliance of the future stand on the current sapling layer. Therefore, targeted control of less desirable species, such as *Ulmus* spp., green ash, sweetgum, and black willow, is of particular importance to facilitate oak establishment.

ACKNOWLEDGMENTS

Funding for this project was provided by the Oklahoma Agricultural Experiment Station, the Department of Natural Resources, Ecology and Management at Oklahoma State University, U.S. Department of Agriculture McIntire-Stennis, and the endowment for the Sarkeys Distinguished Professorship. Thanks to Ed Hurliman for providing access, expertise, and for his inquisitive approach to forest management.

LITERATURE CITED

Gardiner, E.S.; Hodges, J.D. 1998. Growth and biomass distribution of cherrybark oak (*Quercus pagoda* Raf.) seedlings as influenced by light availability. Forest Ecology and Management. 108: 127—134. https://doi.org/10.1016/s0378-1127(98)00220-5.

Krinard, R.M. 1990. *Quercus falcata* var. *pagodifolia* Ell. Cherrybark oak. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. Vol. 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service: 644–649.

Schlaegel, B.E., 1990. *Quercus phellos* L. Willow oak. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. Vol. 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service: 715–720.

Soloman, J.D. 1990. *Quercus lyrata* Walt. Overcup oak. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. Vol. 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service: 681–685.

Vozzo, J.A. 1990. *Quercus nigra* L. Water oak. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. Vol. 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service: 701–703.

REFORESTATION TAX INCENTIVE IMPACTS ON FINANCIAL RETURNS OF LOBLOLLY PINE PLANTATIONS FOR NONINDUSTRIAL PRIVATE FOREST LANDOWNERS IN MISSISSIPPI

Curtis L. VanderSchaaf and Yanshu Li

ABSTRACT

Forest plantation investment returns depend not only on survival and growth rates, but also on costs and revenues associated with various practices. Beyond that, tax-related issues are very important, but are often not addressed and thus most forest financial assessments can be defined as before-tax. Mississippi landowners can reduce reforestation cost burdens through two important tax-related opportunities. Firstly, the Federal reforestation deduction and amortization provisions, and secondly the State-based reforestation tax credit. Impacts of these two income tax reduction opportunities on unthinned loblolly pine (*Pinus taeda*) financial returns were examined for three planting densities (454, 519, and 605 seedlings per acre) for an 80-acre clearcut with a site index of 65 feet (base age 25) at age 26. For Federal tax purposes, a landowner classified as an investor within the 22 percent Federal income tax bracket was assumed. Whether before- or after-tax assessment, the financially optimal planting density was 454 seedlings per acre.

INTRODUCTION

Federal and State cost-share programs and reforestation tax incentives are extremely important for landowners to obtain reasonable financial returns on investment and in helping to maintain this important supply of wood for the State's economy. The Federal Income Tax Reforestation Deduction and Amortization incentives are available to many landowners across the country (Li and others 2022). In addition, the State of Mississippi currently has the unique Reforestation Tax Credit, which provides a tax credit against State income tax liability of up to $10,000 annually to offset costs of practices leading to the reforestation of a property. This analysis examines how much these two reforestation incentives can help reduce the costs of establishing loblolly pine (*Pinus taeda*) plantations, thus leading to maximizing financial returns in Mississippi. Additionally, for these analyses, it is assumed that a landowner is not seeking any reforestation payments from Federal or State cost-share programs.

Federal Income Tax Reforestation Deduction and Amortization Incentives

For any justified (qualified) reforestation expenditure in a particular year, a landowner can deduct up to $10,000 for any qualified timber property (QTP). For any amount of justified (qualified) reforestation costs greater than $10,000 in a particular year, a landowner can amortize that amount over an 84-month period, over 8 tax years due to the half-year convention. In the first tax year (the year in which the expense occurred, or tax year 0) and the last, or eighth, tax year (tax year 7), 1/14th of the reforestation expenses in excess of $10,000 can be deducted, but in tax years 1 through 6, 1/7th of the expenses can be deducted.

Curtis L. VanderSchaaf, Assistant Professor, Mississippi State University, Central Mississippi Research and Extension Center, Raymond, MS 39154

Yanshu Li, Associate Professor, Forest Economics and Taxation Outreach Specialist, University of Georgia, Athens, GA 30602

There are some limitations in the applicability of these incentives to a landowner. Landowners with forests classified as investment (investors) or business (or trade) can use these two provisions. A landowner with a forest classified as a hobby, or personal use, could justify the use of these two provisions by claiming they are planting and caring for the trees for production of timber products, but not for profit. Any use of these two incentives, or provisions, reduces the landowner's timber tax basis by the amount deducted or amortized.

These Federal provisions are highly advantageous as well because they can be used to offset any source of income, whether it be wage, salary, or capital gain. Ironically, these provisions (particularly the amortization) may reduce taxes owed to a greater extent for landowners who spend more on reforestation, or those with higher incomes and hence higher income tax brackets and marginal tax rates.

The definition of a QTP is subjective and can lead to confusion. For simplicity, here it is assumed to be a particular loblolly pine plantation. These reforestation tax incentives offer some extremely important advantages when compared to cost-share programs; there is no sign-up period and no ranking system or waiting in line.

Mississippi State Income Tax Reforestation Tax Credit (MRTC)

The State of Mississippi offers nonindustrial private forest landowners (NIPF) the opportunity to use up to 50 percent of the costs incurred during a year associated with conducting approved reforestation activities to offset that tax year's income tax liability (Tanger 2020). The maximum credit amount that can be received annually is $10,000. To clarify though, the credit only offsets income tax liability. Thus, if a landowner's income tax liability is <$10,000 or any other amount of tax credit received, the remainder of the credit carries over to subsequent tax years until it is fully utilized. For example, if a landowner owes $2,000 in State income tax during the year in which the reforestation costs were incurred, and assuming the landowner qualified for $10,000 of tax credit, the $10,000 annual tax credit will only offset the $2,000 income tax liability, and the remaining $8,000 will carry over for use in subsequent tax years to offset State income tax liability, and so forth. Each tax year, beginning with the tax year in which reforestation costs were first incurred, a landowner can earn up to a $10,000 tax credit, and tax credit amounts incurred in each year can be carried forward to succeeding tax years to offset State income tax liability up to a lifetime total of $75,000 in tax credit.

As with the Federal income tax provisions, there are certain restrictions for landowners to qualify; a partial description follows. Firstly, only Mississippi taxpayers reforesting Mississippi land can claim the credit (e.g., Mississippi Department of Revenue Individual Income Tax Form 80-315). Secondly, a landowner must have an approved reforestation plan produced by a Registered Forester. Thirdly, in most cases, any costs of practices conducted on acres receiving Federal or State cost-share assistance funds for that same practice are not eligible for the tax credit. Fourthly, there are only certain practices approved by the Mississippi Forestry Commission (MFC), and each tax year the MFC establishes reasonable average costs for each practice (to avoid abuse). Practices include the costs of seedlings (different prices for bareroot versus container), costs of labor for the actual planting of seedlings, various site preparation activities, and a first-year herbaceous weed control treatment. Total reforestation costs eligible for the credit is 50 percent of the lesser of either 1) the actual costs for approved reforestation practices or 2) the average costs as established by the MFC for approved reforestation practices (to avoid abuse). Currently, for loblolly pine plantations, the targeted planting density per acre must be ≥450 seedlings, and there does not appear to be any direct restrictions on planting configuration (e.g., rectangularity).

Reducing Reforestation Costs Via Silviculture

Landowners should consider other methods to reduce reforestation costs such as reducing their planting density and establishing seedlings in varying degrees of rectangularity (as opposed to a square configuration). Planting fewer seedlings

per acre decreases the cost of purchasing the seedlings and the cost of placing the seedlings into the ground. Beyond that, the costs of any operation that depends on the number of passes on a property, such as bedding, subsoiling/ripping, banded herbaceous weed control treatments, will likely be decreased when planting density is reduced. Plus, fewer rows resulting from a decrease in planting density may result in lowering the amount of chemical applied to a property, which is beneficial both economically and environmentally.

For the same planting density per acre, rectangular plantings can reduce reforestation costs because once again the number of passes on the property should be less, and thus the associated benefits as described in the previous paragraph also apply. Greater degrees of rectangularity should result in a greater reduction in reforestation costs. However, one should consider reductions in yields resulting from extreme degrees of rectangularity. For loblolly pine, it has generally been found that rectangularity ratios of between row to within row spacing that are 3:1 or less result in decreased reforestation costs with minimal impacts on yields (Adams and Clason 2002, Amateis and others 2004, VanderSchaaf and South 2004).

Lower planting densities, within a reasonable range, can also be beneficial financially because yields of the more valuable chip-n-saw and sawlog classes can be obtained at an earlier age, and that across time, greater yields of these more valuable product classes can be obtained (Amateis and Burkhart 2012, Land and others 2004). Given poor stumpage prices in southwestern Mississippi (Measells 2022), and that pulpwood prices continue to decrease, landowners want to use a planting density that not only reduces the amount of pulpwood, but that also reduces the age at which chip-n-saw and sawlog production begins. Planting densities ranging from around 300 to 625 seedlings per acre, given the quality of seedling being planted today, should produce a quality stand. Planting densities at the lower end of this range will result in obtaining sawlogs at an earlier age and will likely result in more sawlog production across time. Plus, lower planting densities will likely eliminate the need for a pulpwood-dominated first thinning.

The objective of this analysis was to assess the impacts of these two income tax reduction opportunities, along with reducing planting density per acre, on loblolly pine financial returns. To determine how reduced reforestation costs from lower planting densities may impact financial returns, planting densities of 454, 519, and 605 seedlings per acre were examined for an 80-acre stand with a site index of 65 feet (base age 25). Varying degrees of rectangularity when planting seedlings were assumed, reducing reforestation costs. For Federal tax purposes, a landowner classified as an investor in the 22 percent Federal ordinary income tax bracket was assumed.

METHODS

Projections

Projections from an Excel-based (Microsoft, Redmond, WA) simulator entitled GulfLOB produced by the lead author, from which a copy can be obtained, were used for this analysis. The majority of the cutover plantations used in model development were established in the 1960s and 1970s, with a few in the 1980s, in Louisiana, southern Mississippi, and east Texas (Baldwin and Feduccia 1987). To account for the effects of improved genetics, better nursery practices, improved site preparation activities, better vegetation control, among other factors, a 2-year age-shift was used and an increase of 15 percent in the carrying capacity was assumed. Simulations were for three planting densities [i.e., 454 (16 feet by 6 feet), 519 (12 feet by 7 feet), and 605 (12 feet by 6 feet) trees per acre] and for land with a site index of 65 feet (base age 25). No fertilization treatments were considered. It has generally been found that rectangularity ratios of between row to within row spacing of 3:1 or less do not meaningfully impact loblolly pine yields (Adams and Clason 2002, Amateis and others 2004, VanderSchaaf and South 2004); thus, no direct adjustments in yield estimates were made for a particular planting density.

Silvicultural Activities and Associated Costs

GulfLOB does not allow for a direct response in growth and yield to various site preparation activities. However, it was assumed an aerial chemical treatment was applied in July and mechanical treatments were applied in November of tax year 0. Assumed initial costs were $84.58 and $131.44 per acre, respectively; these costs were based on Maggard (2021) for the year of 2020. Due to substantial increases in inflation, costs from 2020 found in Maggard (2021) were increased by 15 percent on an annual basis in both 2021 and 2022. Thus, the chemical and mechanical site preparation treatments were assumed to cost $111.86 and $173.83 per acre in 2023, respectively. Additionally, a banded first-year herbaceous weed control treatment was conducted in April (tax year 1), and due to assumed rectangularity in the planting configurations, a reduction factor in costs was assumed (VanderSchaaf and South 2004).

However, given inflation, which for simplicity was assumed to offset any reduction in banded herbicide application costs due to rectangular plantings (i.e., fewer passes resulting in less fuel, labor, maintenance), the herbaceous weed control cost reported in Maggard (2021) was maintained at a cost of $56.09. Since the 519 and 605 planting densities both used 12-foot row spacings, they were considered the standard and the entire $56.09 was applied. However, for the 454 planting density, because 16-foot row spacings reduce the number of passes on the property (hence reduce fuel, labor, and chemical), the first-year herbaceous weed control costs were reduced to $42.07 per acre for the 454 spacing. This reduction in cost was obtained based on the relative values of the number of passes on the property. The square root of 43,560 square feet is 208.71, and this value was then divided by 12 feet and 16 feet. Thus, for the 519 and 605 planting densities there were 17.39 passes and for the 454 planting density there were 13.04 passes. The relative value of 13.04/17.39 was multiplied by $56.09 to produce $42.07.

Costs by tax year and planting density are shown in table 1. Outright deduction and amortization values by tax year and planting density are shown in table 2. Table 3 contains calculations related to the MRTC. For the 2022 tax year, in relation to the MRTC, the MFC reimbursed landowners $54 per acre for the purchase of seedlings ($54 x 80 acres = $4,320), $60 per acre for the labor of planting ($60 x 80 acres = $4,800),

Table 1—The timing and costs per acre by planting density (seedlings per acre) of reforestation activities, and associated State and local timber severance and Federal and State income taxes

			Planting density		
Component	Operation	Tax year	454	519	605
Reforestation	Chemical site preparation	0	$111.86	$111.86	$111.86
	Mechanical site preparation	0	$173.83	$173.83	$173.83
	Seedlings (Elite mass-control pollinated)	0	$114.41	$130.79	$152.46
	Hand planting	1	$77.18	$88.23	$102.85
	First-year herbaceous weed control (banded)	1	$42.07	$56.09	$56.09
Severance taxes	Pulpwood ($0.30/cord)	26	$2.71	$2.94	$3.25
	Chip-n-saw ($0.12/ton)	26	$12.42	$13.82	$15.42
	Sawtimber ($0.12/ton)	26	$8.75	$8.23	$7.60
Income taxes	Federal (15% capital gains)	26	$381.98	$389.15	$397.00
	State (5% ordinary)	26	$127.33	$129.72	$132.33

Total costs of reforestation operations incurred during tax year 1 were discounted 1 year at a 5 percent interest rate. Total discounted reforestation costs for the 454, 519, and 605 planting densities per acre were $513.67, $553.93, and $589.52, respectively. Total severance taxes at age 26 for the 454, 519, and 605 planting densities per acre were $23.88, $24.98, and $26.27, respectively. Pulpwood included both whole-tree pulpwood and topwood.

Table 2—Per acre recovery (dollars) from the outright deduction in tax years 0 and 1 and amortization schedule by planting density

			Tax year										
YI	PD	Type	0	1	2	3	4	5	6	7	8	NDT	DT
0	454	D	27.50	–	–	–	–	–	–	–	–	27.50	27.50
	519		27.50	–	–	–	–	–	–	–	–	27.50	27.50
	605		27.50	–	–	–	–	–	–	–	–	27.50	27.50
1	454	D	–	26.23	–	–	–	–	–	–	–	26.23	24.98
	519		–	27.50	–	–	–	–	–	–	–	27.50	26.19
	605		–	27.50	–	–	–	–	–	–	–	27.50	26.19
0	454	A	4.32	8.65	8.65	8.65	8.65	8.65	8.65	4.32	–	60.54	51.28
	519		4.58	9.16	9.16	9.16	9.16	9.16	9.16	4.58	–	64.12	54.33
	605		4.92	9.84	9.84	9.84	9.84	9.84	9.84	4.92	–	68.88	58.37
1	454	A	–	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	519		–	0.30	0.61	0.61	0.61	0.61	0.61	0.61	0.30	4.25	3.43
	605		–	0.53	1.07	1.07	1.07	1.07	1.07	1.07	0.53	7.47	6.03

The amortization schedule is of costs in excess of the $10,000 annual outright deduction. Recovery values were calculated using 0.22 (landowner Federal income marginal tax rate) and then divided by 80 acres. For the 454 seedling per acre planting density during tax year 1, there are no amortization recoveries because the total reforestation costs incurred in tax year 1 did not exceed the maximum annual allowable outright deduction of $10,000. The discounted total recovery of the reforestation costs incurred in both tax year 0 and 1 and for both the outright deduction and amortized deduction for the planting densities (seedlings per acre) of 454, 519, and 605 are $103.76, $111.45, and $118.09, respectively.

YI = tax year in which reforestation costs were incurred; PD = planting density (seedlings per acre); Type = type of tax incentive, where D refers to a deduction and A refers to an amortization; NDT = nondiscounted total; and DT = discounted total using a 5-percent interest rate.

$100 per acre for chemical site preparation ($100 x 80 acres = $8,000), $190 per acre for mechanical site preparation ($190 x 80 acres = $15,200), and $75 per acre for postplanting site preparation ($75 x 80 acres = $6,000). For the purposes of the tax credit, postplanting site preparation is defined as a practice to reduce or control undesirable competition within the first growing season of an established crop of trees and is commonly referred to as a first-year herbaceous weed control treatment. In total, across the 80 acres, MFC averaged costs eligible for consideration of the tax credit were $38,320. Actual, nondiscounted, reforestation costs for the planting densities of 454, 519, and 605 seedlings per acre were $41,547.64, $44,863.84, and $47,767.20, respectively. Regardless of planting density, the MFC costs of approved reforestation activities (2022 tax year) were lower relative to the actual reforestation costs. Hence, the landowner is entitled to a tax credit of either 1) 50 percent of the MFC average costs for approved reforestation activities, a value of $19,160, or 2) $10,000—whichever is less. Thus, the landowner is entitled to a tax credit of $10,000.

Table 3—Mississippi Reforestation Tax Credit calculations

Tax year	Income tax liability offset	Credit remaining	Per acre	Discounted to Year 0
	~~ *dollars* ~~~~~~~~~~~~~~~~~~			
0	2,176.00	7,824.00	27.20	27.20
1	2,176.00	5,648.00	27.20	25.90
2	2,176.00	3,472.00	27.20	24.67
3	2,176.00	1,296.00	27.20	23.50
4	1,296.00	0.00	16.20	13.33
Total	10,000.00		125.00	114.60

The Mississippi Reforestation Tax Credit only offsets income tax liability, thus the table shows the recovery of the tax credit per tax year assuming the landowner incurs $2,176.00 annually in Mississippi income tax (ordinary tax rate of 5 percent). Per-acre values were calculated assuming a plantation of 80 acres. An interest rate of 5 percent was used for discounting.

Seedling and Planting Costs

Mass-control pollinated Elite (ArborGen, Inc., Ridgeville, SC) bareroot seedlings at a cost of $252 per thousand were planted by hand. It is assumed that seedlings were purchased in June of tax year 0. Seedling costs were based on observation of common prices for the 2022–2023 planting season. Hand planting per seedling was initially

assumed to cost $0.13 (Maggard 2021). Similar to site preparation costs, a 15-percent increase in both 2021 and 2022 was assumed, and hence final planting costs per seedling was $0.17. Seedling planting costs occurred in tax year 1.

Merchandizing Specifications, Stumpage Revenues, and Interest Rate

Upper stem diameter outside bark of 8 inches, 4 inches, and 2 inches were assumed to represent sawtimber, chip-n-saw, and pulpwood merchantable classes, respectively. Minimum diameters at breast height were 12 inches, 8 inches, and 4 inches for sawtimber, chip-n-saw, and pulpwood merchantable classes, respectively. The harvesting of upper-stem pulpwood, or topwood, was assumed. Stumpage values per green ton for pine pulpwood, chip-n-saw, and sawtimber were $3.27, $12.93, and $23.46, respectively, and were obtained from the 2nd quarter of 2022 for southwestern Mississippi (Measells 2022). An interest rate of 5 percent was used. No sources of revenue beyond pulpwood, chip-n-saw, and sawlog markets were assumed. Table 4 contains estimated yields per acre by product class (based on assumed merchandizing specifications) and planting density, and revenues per acre based on predicted yields and stumpage values.

Landowner Forest Property Income Tax Classification

It was assumed that each planting density generated a single, but distinct, QTP. It was assumed the landowner owned the property (hence no mortgage interest). Beyond that, it was assumed the landowner was classified as an investor for their forest ownership because the landowner had a profit motive for their property (hence they are not personal use or hobby), and they had a management plan demonstrating their profit motive, but they did not have the activities required to classify their forest ownership as a trade or business. Landowners with forests classified as business (both material participant and passive) generally have superior tax benefits because they have less restrictions to deduct the costs of ordinary and necessary practices associated with carrying on their business during the year in which they were incurred. However, in order to be classified as a business, a landowner must spend a meaningful amount of time (and thus cost) visiting and managing their property or attending educational activities. The landowner must demonstrate through documentation to the Internal Revenue Service that indeed they conduct practices justifying the business classification.

Capitalized Ad Valorem Property Taxes and Maintenance/Firelane Costs

In addition, an annual property tax (ad valorem) of $5 per acre per year was assumed, and annual property maintenance/firelane costs of $4 per acre per year was assumed. An ad valorem tax rate of $5 was based on revenues and acreages from 2018 across the State of Mississippi ($4 per acre per year) but increased slightly to bring forward to 2023. Since the landowner was considered an investor, and essentially operating expenses had been excluded from being deducted

Table 4—Yield and revenue per acre by product class and planting density (seedlings per acre) at age 26

	Pulpwood				
Planting density	Whole trees	Topwood	Chip-n-saw	Sawlog	Revenue
seedlings per acre	~~~~~~~~~~ *tons per acre* ~~~~~~~~~~				*dollars*
454	8.4	15.1	103.5	72.9	3,126.00
519	10.5	14.9	115.2	68.5	3,180.34
605	13.5	14.6	128.5	63.4	3,239.93

for tax years 2018 to 2025 when classified as an investor due to the Tax Cuts and Jobs Act of 2017, property maintenance/firelane costs ($4 per acre per year) were capitalized. Stated differently, they were added to an investor's timber tax basis. Beyond that, because the standard deduction was taken each year as opposed to itemizing deductions, all annual ad valorem property taxes ($5 per acre per year) were also capitalized. Thus, these annual costs were placed into the timber tax basis and were used to reduce the capital gains (timber harvest revenues minus timber tax basis minus timber severance taxes minus timber sale administrative costs) from the clearcut at age 26. At age 26, for all three planting densities, the value of the operating expenses (maintenance and firelanes) was $108 per acre ($4 for tax years 0 to 26) and the value of the carrying charges (ad valorem property taxes) was $135 per acre ($5 for tax years 0 to 26).

Timber Severance Tax Reducing the Amount of Capital Gain

Mississippi leverages a timber severance tax on timber harvested. This money is partially used to fund the Forest Resource Development Program cost-share (80 percent of tax) and parts of it are returned directly to the county government (20 percent) in which the timber sale occurred. The Mississippi tax code intends for the landowner (grower) to pay the severance tax; however, it is stated in the code that if the landowner does not pay these taxes, then the liability is passed to the purchasers and they need to pay them. Often the mill will pay severance taxes in Mississippi, and if not, then the logger will pay them. Currently, within Mississippi, the severance tax rates per ton (or cord for pulpwood) are fairly low and thus severance taxes are not significant on a per acre basis. Although purchasers of stumpage may operationally pay the severance tax, for illustrative purposes and this analysis, we assumed the landowner (grower) incurred the cost.

For each sawlog ton (and assumed chip-n-saw ton), a severance tax of $0.12 is charged and each 128-cubic foot cord of pulpwood is taxed $0.30 (Mississippi Code 27-25-1). Thus, because tons is utilized in this assessment, the reported pulpwood tonnage from the simulator was converted to a 128-cubic foot cord by dividing tons by 2.6. The value of 2.6 was obtained by dividing 5,200 pounds per cord (Mississippi Code 75-27-39) by 2,000 pounds per ton. This was equivalent to a severance tax of $0.1154 per ton for pulpwood ($0.30/2.6).

Nonetheless, for Federal and Mississippi State income tax purposes, if paid by the landowner (grower), severance taxes can be subtracted from timber revenues, thus reducing the amount of capital gain that is taxed. Table 5 contains total severance tax amounts and by product class for each planting density.

Assumed Investor Annual Taxable Income and Federal and State Capital Gain Tax Rates

For this analysis, the single landowner annual taxable income, prior to any timber related deductions or amortizations, was assumed to be $70,000. Thus, this landowner was within the Federal 22 percent income tax bracket for the 2022 tax year. As an investor, because the standing

Table 5—Severance taxes per acre associated with the clearcut at age 26 by planting density

	Pulpwood				
Planting density	Whole trees	Topwood	Chip-n-saw	Sawlog	Revenue
seedlings per acre	~~~~~~~~~~~~~~~~~~ *dollars* ~~~~~~~~~~~~~~~~~~				
454	0.97	1.74	12.42	8.75	23.88
519	1.22	1.72	13.82	8.23	24.98
605	1.56	1.69	15.42	7.60	26.27

Sawtimber and chip-n-saw product classes are taxed at $0.12 per ton while pulpwood is taxed at $0.30 per cord (Mississippi Code 27-25-1). To convert from cords to tons, $0.30 was divided by 2.6 producing a value of $0.1154 per ton.

timber would have been owned for more than 1 year before any harvesting operation, and the timber sale would be conducted under a lump-sum contract, the Federal long-term capital gains tax rate applies. It was assumed the landowner was within the 15 percent capital gains tax rate bracket. Within Mississippi, there was no special capital gain tax rate, and thus all income was taxed at the ordinary income tax rate of 5 percent.

It was assumed the landowner's plantation was 80 acres. The acreage impacts the benefits on a per acre basis obtained from the Federal and State income tax provisions. For example, if a landowner owns 100 or 1,000 acres, the $10,000 annual Federal Deduction maximum and the MRTC annual maximum amount of $10,000 still both apply. Table 6 contains capital gain amounts by planting density after subtracting from the per acre revenues the timber tax basis (annual carrying charges of ad valorem property taxes and operating costs of site and firelane maintenance), severance taxes, and timber sale administration costs. The table also contains income tax amounts based on the Federal capital gains tax rate of 15 percent (as opposed to the ordinary income tax rate of 22 percent) and the Mississippi State ordinary income tax rate of 5 percent. Remember the State of Mississippi does not have special capital gains tax rates. A parcel of 80 acres was also selected to reduce the complexity of the analysis due to factors such as the net investment income tax (3.8 percent) on timber income of high earners ($250,000 adjusted gross income married filing jointly or $200,000 adjusted gross income single taxpayer) classified as an investor.

Final Taxable Amounts

For the final clearcut operation, a 10-percent timber sale administration and consulting fee was charged by a consulting forester. This was a common percentage used by consulting foresters (Chhetri and others 2022, Self 2019) and it has been used in recent timber harvest tax assessments (Baral and others 2020, Li and others 2020). Thus, the Federal capital gain at age 26 (table 6) can be expressed as:

$$CG = Revenue - Timber\ Tax\ Basis - Severance\ Tax - Timber\ Sale\ Administration\ Costs \quad (1)$$

where

CG = capital gain amount per acre by planting density at age 26

Revenue = amount of revenue received per acre from pulpwood, chip-n-saw, and sawlog yields per acre and their associated stumpage values at age 26

Timber Tax Basis = ad valorem taxes of $135 plus maintenance costs of $108

Severance Tax = presented in table 5

Timber Sale Administration Costs = 10 percent of revenue

At age 26, the timber tax basis only contains the annual ad valorem property taxes ($135 per acre) and the operating expenses ($108 per acre) associated with maintaining the property and firelanes. All reforestation costs were recovered near the time of planting through the Reforestation Tax Deduction and Amortization

Table 6—Capital gains per acre by planting density

							Income tax	
Planting density	Revenue	Maint	Property	Admin	Severance	Capital gain	Federal	State
seedlings per acre	~~~~~~ *dollars* ~~~~~~							
454	3,126.00	108.00	135.00	312.60	23.88	2,546.52	381.98	127.33
519	3,180.34	108.00	135.00	318.03	24.98	2,594.32	389.15	129.72
605	3,239.93	108.00	135.00	323.99	26.27	2,646.67	397.00	132.33

Revenue is total revenue across all product classes at the clearcut age of 26 years (see table 4).

Maint = operating costs for site maintenance and firelane construction or maintenance ($4 annually from tax year 0 to 26); Property = carrying charges of ad valorem property taxes ($5 annually from tax year 0 to 26); Admin = 10 percent of the revenue per acre; Severance = severance taxes at age 26 (see table 5); Capital gain = revenue minus maint, property, admin, and severance (equation 1); Federal = the amount of income tax owed resulting from the capital gain, using the capital gains tax rate of 15 percent; and State = the amount of income tax owed to Mississippi based on the ordinary income tax rate of 5 percent.

provisions. Timber sale administration costs were $312.60, $318.03, and $323.99 per acre for the 454, 519, and 605 planting densities per acre, respectively.

One final note: it currently is not entirely clear for Mississippi State income tax purposes if a landowner can deduct and amortize reforestation costs in excess of the amount of tax credit received by the landowner from the MRTC (Tanger 2020). Thus, to some degree, recovery of reforestation costs could be increased further beyond what has been reported in this analysis.

RESULTS AND DISCUSSION

Impacts of Taxation on Financial Returns

Most forest financial assessments of southern forests are calculated before tax. This is most likely because of the difficulty in understanding all aspects of taxation and their associated calculations, but also because of the variability among State and local taxation. This analysis is specifically for Mississippi; however, the Federal taxation component of this paper should be applicable to landowners regardless of the State in which they live. Beyond that, results will differ based on IRS landowner classification, complicating analyses. Being classified as a business, particularly a material participant, would be advantageous in terms of tax deductions relative to an investor. For instance, the operating expenses of site maintenance and the maintaining of firelanes and ad valorem property tax carrying charges could be deducted in the year in which the costs were incurred. This is advantageous for at least four reasons. First, the deductions would offset income at the ordinary income tax rate of 22 percent rather than the capital gains tax rate of 15 percent such as at the time of harvest. Second, the negative impacts of discounting would be less. Third, the recovery revenues could be put into another investment to gain interest (e.g., the bank). Fourthly, timber income of high earning ($250,000 married filing jointly or $200,000 single taxpayer) business landowners is not subject to the net investment income tax (3.8 percent). However, a high earning landowner classified as an investor is subject to the net investment income tax (3.8 percent).

Given the assumptions of this analysis, for a particular planting density, when compared to a before-tax financial assessment, landowner net revenues were reduced when conducting an after-tax financial assessment (table 7). However, the optimal planting density of 454 seedlings per acre did not change when an after-tax assessment was conducted. Hence, the increased revenues associated with using a lower planting density, resulting from reduced reforestation costs and a greater amount of sawlog production at the time of harvest, was stronger than the ability of tax provisions, allowing landowners to recover greater amounts of reforestation costs for higher planting densities.

Table 7—After-tax assessment discounted revenues and costs

	Discounted revenues (5%)				Discounted costs (5%)						
PD	Rev	Ded	Amort	MRTC	RC	ST	Maint	Prop	Admin	Fed	State
Seedlings per acre	~~~~~~ *dollars* ~~~~~~											
454	879.16	52.49	51.28	114.60	513.67	6.72	61.50	76.88	87.92	107.43	35.81
519	894.44	53.69	57.76	114.60	553.93	7.03	61.50	76.88	89.44	109.44	36.48
605	911.20	53.69	64.40	114.60	589.52	7.39	61.50	76.88	91.12	111.65	37.22

Total discounted revenues for the planting densities of 454, 519, and 605 seedlings per acre were $1,097.52, $1,120.49, and $1,143.89, respectively, while the total discounted costs were $889.91, $934.70, and $975.28, respectively. After-tax net present value per acre for planting densities of 454, 519, and 605 seedlings per acre was $207.61, $185.79, and $168.61 per acre, respectively. Before-tax net present value per acre for planting densities of 454, 519, and 605 seedlings per acre was $365.49, $340.51, and $321.68 per acre, respectively (Rev minus RC). An interest rate of 5 percent was used.

PD = planting density (seedlings per acre); Rev = stumpage values received (table 4); Ded (table 2) = recovered costs from the Federal Reforestation Tax Deduction and Amortization (Amort) tax provisions (assuming an ordinary income tax rate of 22 percent); MRTC = recovered reforestation costs from the Mississippi Reforestation Tax Credit, assuming an annual state income tax liability of $2,176.00 (table 3); RC = reforestation costs of the chemical and mechanical site preparation operations, seedling purchase and planting costs, and a first-year herbaceous weed control treatment (table 1); ST = severance taxes (table 5); Maint = operating costs for site maintenance and firelane construction or maintenance ($4 annually from tax year 0 to 26); Prop = carrying charges of ad valorem property taxes ($5 annually from tax year 0 to 26); Admin = 10 percent of Rev per acre; Fed = Federal income taxes (15 percent capital gains tax rate, table 6); and State = income taxes (5-percent tax rate, table 6).

CONCLUSIONS

Regardless of whether a financial assessment was before or after tax, for the assumptions used here, the financially optimal planting density was 454 seedlings per acre. Currently, pulpwood markets in Mississippi as a whole are poor (Measells 2022). Beyond that, chip-n-saw markets, and even sawtimber markets, are relatively poor. When a landowner is classified as an investor, reducing reforestation costs by planting fewer seedlings and the use of rectangularity when planting, and associated increases in stumpage values resulting from earlier and greater production of the more valuable sawlog product class, seems to have a greater impact on loblolly pine plantation financial returns than reforestation tax provisions. When only analyzing revenues from traditional timber markets, this analysis suggests that often before-tax financial assessments will result in the same ranking of alternative planting densities as after-tax financial assessments for loblolly pine plantations.

LITERATURE CITED

Adams, J.C.; Clason, T.R. 2002. Loblolly pruning and growth characteristics at different planting spacings. In: Outcalt, K.W., ed. Proceedings of the eleventh biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-48. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 153–155. https://doi.org/10.2737/SRS-GTR-48.

Amateis, R.L.; Burkhart, H.E. 2012. Rotation-age results from a loblolly pine spacing trial. Southern Journal of Applied Forestry. 36: 11–18. https://doi.org/10.5849/sjaf.10-038.

Amateis R.L.; Radtke, P.J.; Hansen, G.D. 2004. The effect of spacing rectangularity on stem quality in loblolly pine plantations. Canadian Journal of Forest Research. 34: 498–501. https://doi.org/10.1139/x03-210.

Baldwin, V.C, Jr.; Feduccia, D.P. 1987. Loblolly pine growth and yield prediction for managed West Gulf plantations. Res. Pap. SO-236. New Orleans, LA: U.S. Department of Agriculture Forest Service, Southern Forest Experiment Station. 27 p. https://doi.org/10.2737/SO-RP-236.

Baral, S., Li, Y.; Mei, B. 2020. Financial effects of the 2017 Tax Cuts and Jobs Act on nonindustrial private forest landowners: a comparative study for 10 southern states of the United States. Journal of Forestry. 118: 584–597. https://doi.org/10.1093/jofore/fvaa032.

Chhetri, S.G., Parker, J.; Izlar, R.L.; Li, Y. 2022. Forest management practices and costs for family forest landowners in Georgia, USA. Forests. 13: 665. 18 p. https://doi.org/10.3390/f13050665.

Land, S.B., Jr.; Roberts, S.D.; Duzan, H.W., Jr. 2004. Genetic and spacing effects on loblolly pine plantation development through age 17. In: Connor, K.F., ed. Proceedings of the twelfth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 413–419. https://doi.org/10.2737/SRS-GTR-71.

Li, Y.; Cushing, T.L.; Frey, G.E. 2022. Tax tips for forest landowners: 2022 tax year. FS-1209. Washington, DC: U.S. Department of Agriculture Forest Service. 4 p. https://www.timbertax.org/publications/fs/taxtips/TaxTip2022.pdf.

Li, Y.; Frey, G.E.; Wang, L. 2020. Estimated effects of the Tax Cuts and Jobs Act of 2017 on private noncorporate forest landowners' timber income in the U.S. South. Forest Policy and Economics. 115: 102118. 5 p. https://doi.org/10.1016/j.forpol.2020.102118.

Maggard, A. 2021. Costs & trends of southern forestry practices 2020. FOR-2115. Auburn, AL: Alabama Cooperative Extension Service. 6 p. https://www.aces.edu/wp-content/uploads/2021/12/FOR-2115_CostsandTrendsofSouthernForestryPractices2020_121021L-G.pdf.

Measells, M. 2022. Mississippi timber price report, 2nd quarter 2022. Mississippi State, MS: Mississippi State University Extension Service. 4 p. https://extension.msstate.edu/sites/default/files/topic-pdfs/Timber%20Prices/2q-2022-price-report.pdf.

Self, B. 2019. Choosing a consulting forester. Publ. P2718. Mississippi State, MS: Mississippi State University Extension Service. 4 p. https://extension.msstate.edu/sites/default/files/publications/publications/p2718.pdf.

Tanger, S. 2020. Paying for a new forest without cost-share funding. Publ. 2420. Mississippi State, MS: Mississippi State University Extension Service. 8 p. http://extension.msstate.edu/sites/default/files/publications/publications/P2420_web.pdf.

VanderSchaaf, C.L.; South, D.B. 2004. Rectangular spacing: an economic benefit? In: Connor, K.F., ed. Proceedings of the twelfth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 437–440. https://doi.org/10.2737/SRS-GTR-71.

IMPACTS OF DIESEL AND INSURANCE COSTS ON OWNER-OPERATOR LOG TRUCK DRIVERS IN THE WESTERN GULF

Curtis L. VanderSchaaf

ABSTRACT

After felling, skidding, delimbing and often topping, and merchandizing of trees, logs must be loaded onto a truck for the hauling of that raw woody resource to a mill. This analysis focuses on owner-operator truck drivers, or self-employed drivers, and looks at the impacts of costs per gallon of diesel ($3, $4, $5, and $6 per gallon), different log truck annual insurance rates ($10,000, $15,000, and $22,000), along with basic assumptions about hauling conditions and maintenance costs, and varying rates of salary received per load ($0.13 to $0.23 per ton per loaded mile, by $0.02 increments) on the net annual revenue of an average log truck driver. For a common rate of $0.19 per ton per loaded mile, and two assumed log truck routes, diesel per gallon increases from $3 to $4, $5, and $6 reduced net annual revenues by up to 16.0, 31.9, and 47.9 percent, respectively. In all cases, reductions were greatest when annual insurance costs were $22,000.

INTRODUCTION

Forestry is a large business in the Southeastern United States and a vital component is regular harvesting to capture the growth, yield, and value of the trees. Forest harvesting, or logging, is extremely important to the fiber supply chain, allowing landowners to conduct management on their property and to receive financial compensation for their capital investment, while providing fiber to processing facilities that convert the raw woody resource into wood products.

After the felling, skidding, delimbing and often topping, and merchandizing of trees, the logs must be loaded onto a truck for hauling or transport of that raw woody resource to a mill. The hauling and transport component, or the trucking component, is vital to the forest harvesting process (Conrad 2018, 2021). Many loggers believe the costliest component of harvesting operations is trucking, and several studies have also shown this to be true (Conrad 2018, 2023; Conrad and others 2018; McConnell 2020). This is particularly true in recent years as truck insurance and diesel fuel costs have been rising at rapid rates. Additionally, the financial difficulties associated with trucking can increase with lower value wood products (e.g., hauling pulpwood versus hauling sawlogs) (Grebner and others 2005).

Mills commonly pay a haul rate that is calculated per ton per loaded mile. Typically, there is a minimum haul distance, often 40 miles (TimberMart-South 2022, VanderSchaaf and others 2023), and the log truck owner is paid this rate for deliveries at distances up to the minimum haul distance. The minimum haul distance varies depending on local market conditions; in the Gulf region of the Southern United States, at the current time, common minimum haul distances are 30, 40, or 50 miles.

The minimum haul distance is used to compensate the log truck owner for the disproportionate amount of time spent loading and unloading on short hauls. On longer hauls, the log truck owner receives the minimum haul rate multiplied by the minimum haul distance, plus an incremental haul rate multiplied by the incremental haul distance [equation (1)]. The incremental haul rate may be the same, higher than, or less than the minimum haul rate.

Curtis L. VanderSchaaf, Assistant Professor, Mississippi State University, Central Mississippi Research and Extension Center, Raymond, MS 39154

$$P = T_D \times HD_M \times HR_M + T_D \times HD_I \times HR_I \quad (1)$$

where

P = mill payment for timber transportation, dollars per ton per loaded mile

T_D = tons of payload delivered

HD_M = minimum haul distance (loaded miles)

HR_M = minimum haul rate (dollars per ton per loaded mile)

HD_I = incremental haul distance (loaded miles)

HR_I = incremental haul rate (dollars per ton per loaded mile)

This analysis is focused exclusively on owner-operator, or independent contract drivers, and hence drivers that are self-employed. In most cases, these drivers pay for the truck and trailer (and other necessary accessories such as straps or chains), insurance for the truck and trailer, licensing, maintenance, fuel, mandatory professional training, salary, fringe benefits such as health insurance, and retirement.

However, in some cases the owner-operator only owns the truck (or cab or tractor), and thus the contractor (the driver) uses the contractee's (i.e., logger's) trailer. I focused on the situation where the owner-operator owns both the truck and trailer. Additionally, I did not focus on log truck drivers who are employed directly by a logging company (i.e., a fleet truck), and hence are generally not directly financially responsible for the purchase, maintenance, licensing, and fuel of the truck and trailer. Both types of drivers (owner-operator and employed) are common in the Southeastern United States. For instance, Conrad (2023) reported that 36 percent of timber harvested across much of the Southeastern United States was hauled by owner-operator log truck drivers.

VanderSchaaf and others (2023) described several ways in which log truck drivers receive compensation. This analysis concentrated on those cases where a driver is paid using the per ton per loaded mile approach (their Approach 2). Basically, this type of compensation only applies when the trailer is loaded with raw woody unprocessed products and heading to the mill [e.g., loaded miles in equation (1)]; hence, the driver is not directly compensated for the return trip from the mill. With this approach, a driver receives revenue not only for the weight (tons), but also the distance (miles). Thus, more revenue is received for heavier loads transported greater distances.

This compensation approach has several advantages for the log truck driver. For instance, the number of daily loads is not a consideration; longer hauls do not penalize the driver. If a driver is paid exclusively by the number of daily loads, longer hauls will reduce payment to the driver unless the pay rate is adjusted accordingly. For the most part, the value of the timber (e.g., pulpwood versus sawtimber) doesn't impact driver compensation. Owner-operators typically receive the entire haul rate paid by the mill and must use the haul rate to cover fuel, maintenance, truck payments, in addition to a personal salary and fringe.

When using the dollars-per-ton-per-loaded-mile payment approach, the driver is paid for driving the minimum haul distance even if the actual distance is less. If the actual haul distance exceeds the minimum haul distance, then the driver is paid an additional incremental haul rate for the distance traveled beyond that minimum haul distance (Conrad 2021). For owner-operators, rates near the end of 2022 varied from around $0.16 to $0.22 per ton per loaded mile.

This analysis looked at the impacts of different costs of diesel per gallon ($3, $4, $5, and $6 per gallon), different log truck annual insurance rates ($10,000, $15,000, and $22,000), and two haul distances (40 and 60 miles, given a 50-mile minimum haul distance), along with basic assumptions about hauling conditions and maintenance costs, and varying rates of salary received per load ($0.13 to $0.23 per ton per loaded mile, by $0.02 increments) on the average net annual revenues of a log truck driver. Conrad (2023) reports that log truck insurance costs and fuel costs were the two most important hauling factors of logging businesses across the Southeastern United States.

METHODS

Hauling Conditions

Many factors beyond those examined (cost of diesel and insurance, haul distance, and pay rate) can impact net annual revenues of owner-operators. These factors include legal gross vehicle weight (GVW) and tare weight of the truck and trailer (the weight of the trailer when empty). Load height and even load length restrictions can occasionally impact payloads and log truck routing (e.g., underpass heights may limit the ability of a truck to pass under when loaded). Other factors include type of trailer (four-bolster versus pole trailer), whether payments are still being made on the truck and trailer, mechanic hourly rates, type of road condition and maintenance (unpaved woods roads, county or parish roads, four-lane State roads, interstates). Additional concerns are title and licensing fees, and number of weeks, days, and hours per day worked. Weather is another important factor in terms of travel speed, but also weather frequently limits access to logging sites; however, this analysis did not address the impacts of weather directly.

For this analysis, it was assumed the owner-operator owned both the truck and trailer outright and hence no payments were made for these vehicles. However, licensing and taxes were assumed to cost $5,550 annually. Further, it was assumed the driver had purchased a harvest permit and thus a constant maximum GVW of 88,000 pounds was assumed [the legal GVW for a truck and trailer in Mississippi beginning in July of 2023 (Section 63-5-33)]. I assumed a day cab truck and thus a weight of 19,000 pounds, and the use of a drop-center four-bolster trailer that weighed 10,000 pounds. Hence, the tare weight of the truck and trailer was 29,000 pounds and for simplicity I assumed each haul contained a payload of 59,000 pounds (29.5 tons). It was assumed no overloading or underloading at the logging site occurred.

Loading, Unloading, In Route Delays, and Daily Routine Service

The amount of time required to be loaded at the logging site (woods wait) and the amount of time required to be unloaded at the wood processing facility (mill wait) were other factors (Conrad 2023). The woods wait and mill wait were both assumed to be 30 minutes, thus 60 minutes in total per haul trip. Delays for various reasons are another consideration; 30 minutes in total per haul trip was assumed for both in-route delays and daily routine service. Thus, in total, 90 minutes were needed per haul trip for loading, unloading, delays, and daily service.

Log Truck Routes, Fuel Costs, Repair and Maintenance Costs, Insurance Costs, and Licensing and Taxes

Table 1 contains the amount of time spent on each category of road type; the time required depends on the average miles per hour (mph) and distance traveled per category of road type. Based on a combination of personal observation and Conrad (2023), the haul distances used in this analysis of 40 and 60 miles are reasonable for Mississippi. Miles per gallon differs by road condition. An approach described in Grebner and others (2005) was used to estimate annual repair and maintenance costs; however, costs and rates have been updated for the current economic situation and inflation. A constant rate per mile was assumed for four categories on the truck and trailer: brakes, tires, normal maintenance, and minor repairs (table 2). Relative rates of damage and need for repair were assigned to each category of road type. Then, a final estimate of repair and maintenance costs per haul trip was obtained by taking into account the number of miles a truck traveled in each category of road type (table 1) and the associated repair and maintenance costs per mile per trip (tables 2 and 3). Additional maintenance costs included such things as diesel exhaust fluid.

Table 1—Number of miles per road type, average travel speed (miles per hour), number of minutes spent traveling on each road type, and miles per gallon of diesel when fully loaded and unloaded

		Log truck route 1				Log truck route 2			
Load	Road type	Miles	Speed	Travel	Fuel	Miles	Speed	Travel	Fuel
Fully loaded	Forest	0.5	8.0	3.8	2.0	0.5	8.0	3.8	2.0
	Gravel county road	0.0	30.0	0.0	2.8	3.5	30.0	7.0	2.8
	Paved county road	0.0	40.0	0.0	3.6	1.0	40.0	1.5	3.6
	Two-lane highway	31.5	50.0	37.8	5.0	0.0	50.0	0.0	5.0
	Limited access highway	5.0	60.0	5.0	5.0	46.0	60.0	46.0	5.0
	Federal interstate	0.0	65.0	0.0	6.0	0.0	65.0	0.0	6.0
	Urban street	3.0	20.0	9.0	2.3	9.0	20.0	27.0	2.3
	Total	40.0	48.5	55.6	4.5	60.0	51.5	85.3	4.0
Unloaded	Forest	0.5	8.8	3.4	2.2	0.5	8.8	3.4	2.2
	Gravel county road	0.0	33.0	0.0	3.0	3.5	33.0	6.4	3.0
	Paved county road	2.0	44.0	2.7	3.6	1.0	44.0	1.4	3.6
	Two-lane highway	1.0	55.0	1.1	6.0	0.0	55.0	0.0	6.0
	Limited access highway	3.0	66.0	2.7	6.0	46.0	66.0	41.8	6.0
	Federal interstate	30.5	70.0	26.1	6.0	0.0	70.0	0.0	6.0
	Urban street	3.0	22.0	8.2	2.5	9.0	22.0	24.5	2.5
	Total	40.0	63.7	44.3	5.2	60.0	56.6	77.5	4.6

Two-lane highway = State or Federal two-lane highway; Limited access highway = four lanes, noninterstate; Federal interstate = 80,000 pounds gross vehicle weight restriction.

Table 2—Maintenance costs per mile by category and road type

Category	Per mile	Limited access	Forest road	Graveled county	Paved county	State/Fed two lane	Urban streets	Federal interstate
	Dollars	~~~~~~~~~~~~~~~~~~~~~~~~~ *Percentage* ~~~~~~~~~~~~~~~~~~~~~~~~~						
Brakes	0.06	100	200	150	150	125	150	100
Tires	0.17	100	200	150	175	100	100	100
Normal maintenance	0.02	100	200	175	150	125	125	125
Minor repairs	0.03	100	200	150	150	100	125	100

Limited access = limited access highway (four lanes, noninterstate); State/Fed two lanes = State or Federal two-lane highway; Federal interstate = Federal interstate (80,000 pounds gross vehicle weight restriction).

Table 3 contains various estimates of driver performance, total fuel cost, and maintenance costs for each truckload and annually for each of the two log truck routes. The total maintenance costs (when loaded and unloaded) given market conditions in 2022, were higher than those reported in Leslie and Murray (2022). However, their reported costs may not be overly indicative of maintenance costs for log trucks. Three rates of annual insurance costs were assumed ($10,000, $15,000, and $22,000). The $10,000 annual cost could be considered representative for drivers with 3 or more years of experience while the $22,000 annual cost could be considered representative of a new authority, or a new driver with limited or no experience. These costs are comparable to those reported in Conrad (2023) for trucks within fleets of logging businesses. However, often insurance premiums are higher for owner-operators. Thus, currently,

depending on the situation, annual tractor-trailer insurance costs for owner-operators may actually exceed $22,000. As mentioned, costs of $1,600, $3,500, and $450 were assumed annually for licensing, ad valorem taxes, and heavy use tax, respectively (Grebner and others 2005); these costs are likely conservative.

Hours Worked Annually and Minimum Haul Distance

It was assumed the driver worked 51 weeks per year, 5 days per week, and 10 hours per day. A minimum haul distance of 50 miles was assumed [HD_M in equation (1)], and two haul distances were examined: 40 (log truck route 1) and 60 (log truck route 2) miles. Thus, 40 miles was less than the minimum haul distance and 60 miles was greater than the minimum haul distance. For the haul distance of 60 miles, the incremental haul rate was $0.01 more than the minimum haul rate [HR_M in equation (1)]. Table 3 contains estimates of the number of haul trips per day and annually for each of the two log truck routes. Table 4 presents total costs per load and annually for each log truck route.

Salary Rate

For the 50-mile minimum haul distance, compensation to the log truck driver ranged from $0.13 to $0.23 per ton per loaded mile, by $0.02 increments. For the 60-mile route, each additional mile beyond the minimum haul distance of 50 miles received an additional cent, $0.01, per ton per loaded mile. Hence, the incremental haul rate ranged from $0.14 to $0.24 per ton per loaded mile, by $0.02 increments.

RESULTS AND DISCUSSION

Figures 1 and 2 provide relative comparisons of net annual revenues that could be received given assumptions utilized during this assessment. The

Table 3—Individual load and annual summary values by log truck route

	Per load		Annual	
Factor	Route 1	Route 2	Route 1	Route 2
Miles	40	60	32,240	36,320
Hours driving loaded	0.93	1.42	746	860
Hours driving unloaded	0.74	1.29	595	782
Average speed loaded (miles per hour)	48.48	51.48	–	–
Average speed unloaded (miles per hour)	63.66	56.63	–	–
Hours spent not driving	1.50	1.50	1,209	908
Hours spent driving	1.66	2.71	1,341	1,642
Total time per trip	3.16	4.21	2,550	2,550
Trips per day worked	3.16	2.37	806	605
Trips per week worked	15.80	11.87	–	–
Maintenance costs loaded	$12.10	$18.01	$9,750.44	$10,903.71
Maintenance costs empty	$12.01	$18.01	$9,675.88	$10,903.71
Diesel at $3 per gallon	$49.76	$83.49	$40,107.18	$50,538.50
Diesel at $4 per gallon	$66.35	$111.32	$53,476.23	$67,384.66
Diesel at $5 per gallon	$82.94	$139.15	$66,845.29	$84,230.83
Diesel at $6 per gallon	$99.52	$166.98	$80,214.35	$101,076.99
Percentage miles on woods roads	1.25	0.83	–	–

Log truck route 1 = 40-mile haul; log truck route 2 = 60-mile haul.

Table 4—Total cost per load and annually for each route (40 or 60 miles) by cost of diesel per gallon and annual insurance cost ($10,000, $15,000, and $22,000)

Rate	Route	Diesel cost per gallon	Annual insurance cost		
			$10,000	$15,000	$22,000
			~~ *Dollars* ~~~~~~~~~~~~~~~~~~~~~~		
Per load	**1 (40 miles)**	3	93.16	99.36	108.05
		4	109.74	115.95	124.63
		5	126.33	132.54	141.22
		6	142.92	149.12	157.81
	2 (60 miles)	3	145.20	153.46	165.02
		4	173.03	181.29	192.85
		5	200.86	209.12	220.68
		6	228.69	236.95	248.51
Annual	**1 (40 miles)**	3	75,083.49	80,083.49	87,083.49
		4	88,452.55	93,452.55	100,452.55
		5	101,821.61	106,821.61	113,821.61
		6	115,190.67	120,190.67	127,190.67
	2 (60 miles)	3	87,895.92	92,895.92	99,895.92
		4	104,742.08	109,742.08	116,742.08
		5	121,588.25	126,588.25	133,588.25
		6	138,434.41	143,434.41	150,434.41

net annual revenues assumed the truck (tractor or cab) and trailer were owned free and clear. These net revenues would be greatly reduced if the price of a new truck and trailer were included. The impact would depend on the manufacturer, whether a day cab or sleeper cab was purchased, financing, down payment, time since purchase, and tax depreciation provisions. Net annual revenues would also be reduced if the purchase price of used vehicles were included, but of course, to a lesser extent. Similarly, the costs for the used truck and trailer would depend

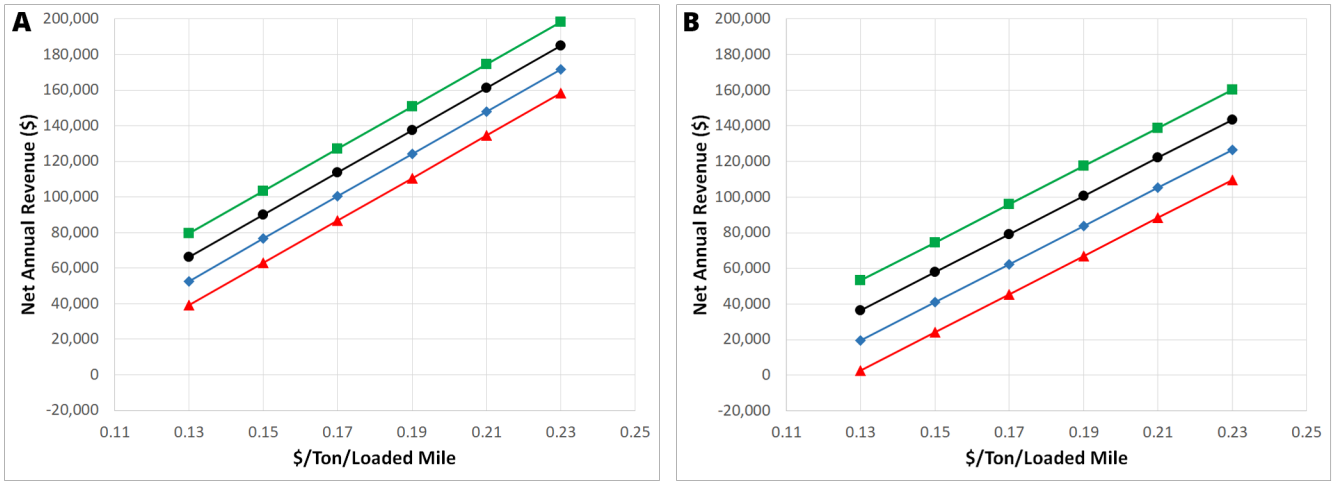

Figure 1—Log truck driver net annual revenue by dollars per ton per loaded mile revenue and cost of diesel per gallon [green line (squares) = $3; black line (circles) = $4; blue line (diamonds) = $5; and red line (triangles) = $6]. (A) is for log truck route 1 (40-mile haul, payment is based on the minimum haul distance of 50 miles) and (B) is for log truck route 2 (60-mile haul). Assumed annual insurance cost of $10,000.

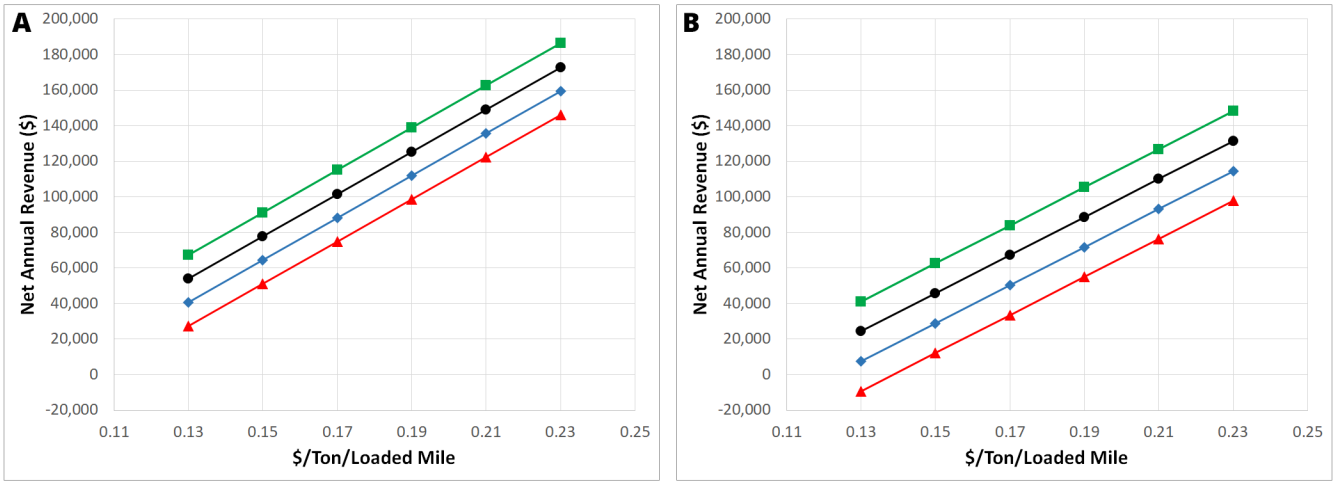

Figure 2. Log truck driver net annual revenue by dollars per ton per loaded mile revenue and cost of diesel per gallon [green line (squares) = $3; black line (circles) = $4; blue line (diamonds) = $5; and red line (triangles) = $6]. (A) is for log truck route 1 (40-mile haul, payment is based on the minimum haul distance of 50 miles) and (B) is for log truck route 2 (60-mile haul). Assumed annual insurance cost of $22,000.

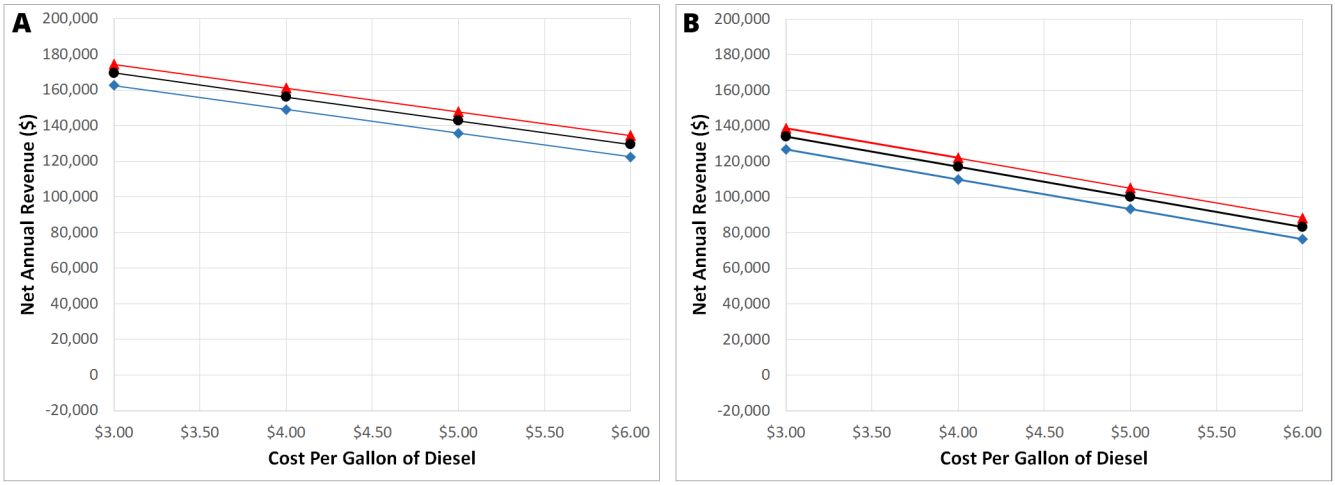

Figure 3— Log truck driver net annual revenue by cost of diesel per gallon for three rates of annual insurance cost [red line (triangles) = $10,000; black line (circles) = $15,000; and blue line (diamonds) = $22,000) for dollars per ton per loaded mile revenue = $0.21]. (A) is for log truck route 1 (40-mile haul, payment is based on the minimum haul distance of 50 miles) and (B) is for log truck route 2 (60-mile haul).

on the manufacturer, age, miles driven, condition of the vehicle, time since purchase, and any down payment.

When using the dollars-per-ton-per-loaded-mile compensation approach, an important consideration is the loading time and waiting time at the mill, or turn-times (Conrad 2021, 2023). Notice in equation (1) that there was no direct consideration for time. If a driver perpetually waits for extended times at the harvesting site or at the mill, net annual revenues can be reduced by meaningful amounts. For instance, for the 40-mile haul, if a driver waits an extra 30 minutes both at the harvesting site and the mill (one more hour in total per load), and they still only want to work 10 hours per day, essentially one trip is lost per day (2.40 rather than 3.16 loads per day as in table 3). The time per load is then increased from 3.16 to 4.16 hours. In order to reach 3.16 trips per day, the driver would need to work 13 hours per day. Activities that can reduce waiting times are very beneficial to owner-operator drivers when they are paid using the dollars-per-ton-per-loaded-mile approach; more driving time means more revenue.

For a common rate of $0.19 per ton per loaded mile, when assuming no direct payments for the vehicles, and for the 60-mile haul (log truck route 2), diesel per gallon increases from $3 per gallon to $4, $5, and $6 reduced net annual revenues by up to 16.0, 31.9, and 47.9 percent, respectively (figs. 1 and 2). For the 40-mile haul (log truck route 1),

diesel per gallon increases from $3 per gallon to $4, $5, and $6 reduced net annual revenues by up to 9.6, 19.3, and 28.9 percent, respectively. When using a rate of $0.21 per ton per loaded mile (figs. 1–3), when assuming no direct payments for the vehicles themselves, for the 60-mile haul, diesel per gallon increases from $3 per gallon to $4, $5, and $6 reduced salaries by up to 13.3, 26.6, and 39.8 percent, respectively. For the 40-mile haul, diesel per gallon increases from $3 per gallon to $4, $5, and $6 reduced salaries by up to 8.2, 16.4, and 24.7 percent, respectively. In all cases the greatest reductions were seen when annual insurance costs were $22,000. Higher insurance rates exacerbate the problem of higher diesel prices by increasing overall costs.

It should be realized that net annual revenue estimates must also pay for fringe benefits such as health and dental insurance, retirement, and social security payments. Hence, the net annual revenues could be reduced by 33 percent or more to account for such factors.

This assessment is an initial attempt to gain knowledge about optimal haul rates for log truck owner-operators in Mississippi given current market conditions. At the current time, common rates of dollars per ton per loaded mile are probably $0.19 and $0.20. An important caveat for future analyses is that more realistic reductions in loads per day should be considered. For instance, a driver can't drive 3.16 trips per day; this should be rounded down to 3 trips per day. When fuel costs increase, in particular, but other expenses as well, often logging businesses and mills will temporarily adjust their contract owner-operator haul rates. However, adjustment response times often vary across the businesses and mills, and hence this can be detrimental to owner-operator log truck drivers, perhaps leading them to or causing them to find employment elsewhere. Given the fairly strong reliance on owner-operator log truck drivers to haul wood in the Southeastern United States (Conrad 2023), temporary haul rate adjustments can be extremely important.

CONCLUSIONS

The cost of diesel can have substantial impacts on log truck driver net annual revenue, and the problem is exacerbated by increasing annual vehicle insurance costs as well as the impact of inflation on all aspects of operating a log truck. It is important that legislation is passed and that policies are implemented that maintain reasonable diesel costs. As diesel increases to $5 or $6 a gallon, revenues become extremely marginal and this will likely lead to the use of older vehicles, less routine maintenance, and potentially for the hauling of payloads in excess of what would be legal given the 88,000 GVW maximum. These situations are not only dangerous for the drivers themselves, but also for the general public. In addition to safety concerns, we should all be concerned about log truck driver net annual revenues because without the log truck component of the wood supply chain, landowners will likely have very limited forest management options and abilities to generate revenues from their capital investments. Further yet, the wood supply for mills that produce wood products demanded by society will be substantially decreased or even eliminated.

LITERATURE CITED

Conrad, J.L. IV. 2018. Costs and challenges of log truck transportation in Georgia, USA. Forests. 9: 650. 14 p. https://doi.org/10.3390/f9100650.

Conrad, J.L. IV. 2021. Evaluating profitability of individual timber deliveries in the US South. Forests. 12: 437. 14 p. https://do.org/10.3390/f12040437.

Conrad, J.L. IV. 2023. Log truck insurance premiums, claims, and safety practices among logging businesses in the US South. International Journal of Forest Engineering. 34: 204–215. https://doi.org/10.1080/14942119.2022.2124704

Conrad, J.L. IV; Greene, W.D.; Hiesl, P. 2018. The evolution of logging businesses in Georgia 1987–2017 and South Carolina 2012–2017. Forest Science. 64: 671–681. https://doi.org/10.1093/forsci/fxy020.

Grebner, D.L.; Grace, L.A.; Stuart, W.; Gilliland, D.P. 2005. A practical framework for evaluating hauling costs. International Journal of Forest Engineering. 16: 115–128. https://doi.org/10.1080/14942119.2005.10702520.

Leslie, A.; Murray, D. 2022. An analysis of the operational costs of trucking: 2022 update. Atlanta, GA: American Transportation Research Institute. 53 p. https://truckingresearch.org/wp-content/uploads/2022/08/ATRI-Operational-Cost-of-Trucking-2022.pdf. [Date accessed: October 30, 2023].

McConnell, T.E. 2020. Unit costs and trends within Louisiana's logging contract rate. Forest Products Journal. 70: 50–59. https://doi.org/10.13073/FPJ-D-19-00036.

TimberMart-South. 2022. Biomass, logging rates, & species detail 1st quarter 2022. Athens, GA: Norris Foundation.

VanderSchaaf, C.L.; Conrad, J.; Withers, W.; Livingston, D. 2023. How do log truck drivers receive payment? P3850. Mississippi State, MS: Mississippi State University Extension. 8 p.

Bragg, D.C.; Oswald, B.P.; Koerth, N.E., eds. 2024. Proceedings of the 22nd biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-274. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 358 p. https://doi.org/10.2737/SRS-GTR-274

The 22nd Biennial Southern Silvicultural Research Conference was held in Nacogdoches, TX, March 21–23, 2023. This conference provided a forum for silviculturists, researchers, and practitioners to exchange the latest research information on the ecology and management of southern forests. Of the 68 oral presentations and 46 posters presented during the conference, 36 papers and 16 extended abstracts were submitted for these proceedings. The papers cover 10 subject areas, which include growth and yield and modeling; fire; hardwoods; shortleaf pine; restoration and reclamation; loblolly pine; intermediate treatments; management and silvics; ecosystem services; and longleaf pine. Papers and abstracts from oral and poster presentations provide managers with the latest research from universities, government agencies, industry, and other natural resource-based institutions.

Keywords: Afforestation, biomass, old growth, hardwoods, disturbance, ecophysiology, fertilization, herbicide, LiDAR, growth and yield, fire, history, hybrids, invasive species, loblolly pine, longleaf pine, modeling, oak, plantations, shortleaf pine.

In accordance with Federal civil rights law and U.S. Department of Agriculture (USDA) civil rights regulations and policies, the USDA, its Agencies, offices, and employees, and institutions participating in or administering USDA programs are prohibited from discriminating based on race, color, national origin, religion, sex, gender identity (including gender expression), sexual orientation, disability, age, marital status, family/parental status, income derived from a public assistance program, political beliefs, or reprisal or retaliation for prior civil rights activity, in any program or activity conducted or funded by USDA (not all bases apply to all programs). Remedies and complaint filing deadlines vary by program or incident.

Persons with disabilities who require alternative means of communication for program information (e.g., Braille, large print, audiotape, American Sign Language, etc.) should contact the responsible Agency or USDA's TARGET Center at (202) 720-2600 (voice and TTY) or contact USDA through the Federal Relay Service at (800) 877-8339. Additionally, program information may be made available in languages other than English.

To file a program discrimination complaint, complete the USDA Program Discrimination Complaint Form, AD-3027, found online at http://www. ascr.usda.gov/complaint_filing_cust.html and at any USDA office or write a letter addressed to USDA and provide in the letter all of the information requested in the form. To request a copy of the complaint form, call (866) 632-9992. Submit your completed form or letter to USDA by: (1) mail: U.S. Department of Agriculture, Office of the Assistant Secretary for Civil Rights, 1400 Independence Avenue, SW, Washington, D.C. 20250-9410; (2) fax: (202) 690-7442; or (3) email: program.intake@usda.gov.

USDA is an equal opportunity provider, employer, and lender.